BIODEGRADATION AND ENVIRONMENTAL IMPACT OF LIPID-RICH WASTES UNDER AEROBIC COMPOSTING CONDITIONS

by

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ABSTRACT

The aim of this research was to evaluate the 'compostability' of organic wastes rich in lipids, such as canola oil and grease trap sludge, when added to yard trimmings or food wastes. This study was divided in three parts: composting process performance, environmental impact, and compost quality. In addition, a composting simulation model was developed in this study.

Aerobic biodegradation of yard trimmings and food waste loaded with lipidic compounds, up to 35% dry solids (ds) for the canola oil tests, and up to 10% ds for grease trap sludge, produced satisfactory results in terms of temperature profile, lipids and volatile solids reduction, wet mass consumption, and moisture content removal. For the high rate phase of composting, treatments with either canola oil or grease trap sludge added resulted in biodegradation rate values for volatile solids (k_{vs}) of 0.009-0.039 day^{-1}. In contrast, yard trimmings alone had a k_{vs} of 0.033 day^{-1} and food waste alone had a k_{vs} of 0.023 day^{-1}.

For treatments with yard trimmings as main substrate, the addition of 5% ds grease trap sludge had similar air emission as yard trimmings alone. The addition of 10% ds grease trap sludge to yard trimmings resulted in more nitrous oxide and carbon dioxide, less ammonia, and similar odor emissions, when compared with the emissions of yard trimmings alone. As for the phytotoxic potential, treatments with 10% ds grease trap sludge added (to food waste or yard trimmings) resulted in germination indexes similar to the treatment with distilled water alone, but their values were significantly smaller than the values for the treatments with 5% ds or no lipid added.

A 'macrokinetic' model was developed using a dynamic modeling approach (mass and energy balances with kinetic parameters) for the simulation of the composting process. The model allowed for the inclusion of lipid wastes as an energy amendment, while including provisions for temperature control through aeration. These produced satisfactory results in terms of thermal parameters simulation.

As a practical recommendation, yard trimmings composting with grease trap sludge added at 5% ds would result in enhanced thermal performance, improved rate and extent of biodegradation of solids and lipids, and greater overall reduction in wet mass and water content, when compared with the composting of yard trimmings alone.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

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DEDICATED

To my Dad
for being my guiding star and my inspiration

To my Mom
for being the source of my strength and inspiration, and for showing me through her example to take one thing at a time and to enjoy life and the present time

To all the children of war, girls and boys, my siblings
wishing in my heart that their numbers stop increasing and as a reminder that the future exists in us and that dreams do come true

CHAPTER 1
GENERAL INTRODUCTION

1.1 INTRODUCTION

Fats, Oils, and Grease (FOGs) residues, also known as lipid-rich wastes, are considered problematic substances in both liquid and solid waste treatment systems. In wastewater collection systems, FOGs tend to clog drainpipes and sewer lines, producing an odor nuisance and sewage back-ups into residential and commercial facilities (GVRD 2000). The presence of FOGs under anaerobic conditions often leads to the corrosion of sewer pipes (Becker et al. 1999, Stoll and Gupta 1997).

In wastewater treatment operations, FOGs are involved in the formation of scum (floating foams). Scum is considered a residual; therefore it needs further treatment and/or disposal. Due to the biodegradable nature of FOGs some countries, like France, have banned their disposal in sanitary landfills (Lefebvre et al. 1998).

In the Lower Mainland of B.C., as of July 2000, a new amendment to the Greater Vancouver Sewerage and Drainage District's By-Law No. 164 (GVSDD 2000) requires that all food establishments use grease interceptors. This amendment called "Code of Practice for Wastewater Management at Food Sector Establishments" is applicable to food processors, retailers, and food outlets. The purpose of this Code is to reduce grease discharges into the sewer. The grease trap sludge generated by the grease interceptors will be collected by trucks and will be discharged at the non-domestic Trucked Liquid Waste facility of the Iona Island Wastewater Treatment Plant (WWTP) at a cost of $\$61.62/m^3$. It is noticeable that the unit cost of treating this waste at the wastewater treatment plant has increased by more than 30% in one year.

An estimated 20,000 tons per year of lipid-rich wastes enter the four major wastewater treatment plants of the Greater Vancouver Regional District (GVRD). From that amount, more than 4,000 tons per year end up in the receiving waters, due to the low treatment efficiency in half of the wastewater treatment plants. For example, the FOGs' treatment efficiency at Iona Island WWTP is about 50% (GVRD 1999, 1998).

The main source of FOGs (as grease trap sludge) is the food industry, particularly fast food restaurants, slaughterhouses, rendering plants, oil mills, fish and poultry processors, milk and cheese industry, and meat processors. About 55% of all

food-processing firms of British Columbia are located in the Greater Vancouver area (BCMAFF 2000).

According to the City of Vancouver Wastewater, Storm Water and Watercourse By-Law (1999), and the Greater Vancouver Sewerage and Drainage District's By- Law (1990), disposal regulations do not allow FOG's concentration of more than 150 mg/L to be discharged in sewer lines. In April 1999, 50% of the companies that appeared in the GVSDD Non-Compliance List were there due to violations on their effluent permit limits for total oil and grease (GVRD 1998).

There are three factors that will translate into an increase in FOGs generation: firstly, the increase in population in the region, particularly the transient population due to tourism and its demand for goods and services; secondly, the growth in the seafood and poultry industries; and thirdly, an increase in the production of health-conscious food such as low fat and fat free products.

Stricter organic matter disposal regulations, increasing treatment cost at the wastewater treatment plant, and increasing recycling goals present a need for innovative treatment options for organic wastes, particularly for those not previously treated by aerobic biodegradation processes.

1.2 COMPOSTING AS AN ALTERNATIVE

Composting has proven to be a very successful bioprocess for the treatment of mineral oils (hydrocarbons) and residues from oil extraction processes (e.g. olive oil) (Filippi et al. 2002, Wan et al. 2002, Kirchmann and Ewnetu 1998, Cegarra et al. 1996). However, composting of food processing wastes rich in oil and grease has not been widely applied.

There are a number of advantages of using lipid-rich wastes in aerobic composting, such as improved heat production, faster composting process, and a pathogen-free final product (LaPara and Alleman 1997, Gariépy et al. 1989). The high-energy content of lipid residues, more than double the energy content of sugars and starch (Wiley 1957), can be used to generate heat during aerobic microbiological processes, thus making it easier to fulfil the time-temperature conditions to achieve the pathogen reduction regulatory requirements. Haug (1993) affirms that lipids, or fats, are readily degradable in composting systems.

There are only few studies about the aerobic degradation of organic wastes loaded with lipid-rich residues of vegetable and animal origin. Lefebvre et al. (1998) found that a lipidic mixture of about 80% fatty acids is easily degradable. In the literature review, values above 85% were found for FOGs biodegradation during aerobic digestion (Keenan and Sabelnikov 2000, Becker et al. 1999, Wakelin and Forster 1997,1998, Mahendraker and Viraraghavan 1994, Fernandes et al. 1988, Viel at al. 1987a,b, Wiley 1956). In contrast, biodegradation of FOGs under anaerobic processes has achieved a maximum of 50-65% (Gutierrez et al. 1999, Dinel et al. 1990).

The value of compost as fertilizer is based on its nutrient content. Nitrogen losses during composting can be as high as 50% of the total initial nitrogen (Witter and Lopez-Real 1988). The loss of nitrogen is related to the carbon to nitrogen balance in the composting mix. By supplementing the composting mix with an easily degradable source of carbon, an increased metabolic rate is expected; thus resulting in more nitrogen metabolized, and potentially more nitrogen retained in the final product.

The environmental impact of the addition of lipids in composting has not been reported in the literature, particularly, greenhouse gas emissions and odour generation. Since these emissions play a key role in the environmental friendliness and public acceptance of composting, it is desirable to quantify these gaseous emissions.

1.3 AIM OF THE STUDY

The aim of this study was to determine the feasibility of biodegrading organic wastes rich in lipids, under solid substrate aerobic composting conditions. Specifically, the kinetics and thermal parameters of the biodegradation process were studied. In addition, the environmental impact of this biodegradation process, in terms of greenhouse gases and ammonia emissions, and odor generation, were investigated. The quality of the compost produced, in terms of nitrogen content and phytotoxicity, was also evaluated.

1.4 SCIENTIFIC CONTRIBUTION

The contributions of this study are in three different areas:

1. Composting Science: The results of this research will fill some of the scientific gaps in the understanding of the degradation of lipid-rich wastes during aerobic composting, mainly via knowledge of the biodegradation kinetics and

thermal behaviour. In addition, the outcomes of the composting process, in terms of product quality, and environmental impact, are original contributions to the composting field. The data and information thus generated would be useful for the design of an efficient system for treating lipid-rich wastes using an environmentally friendly bioprocess that results in a value-added product.

2. Composting Technology: By examining the composting process' performance when using lipid-rich wastes, recommendations can be made to incorporate such waste into commercial composting operations. In particular, the effect of adding lipid-rich wastes on the compost product will provide useful information relevant to the marketing of the compost product. Such specific information will benefit the compost producers.

3. Waste Management Strategies: By way of finding an alternative treatment for lipid-rich wastes such as grease trap sludge, practical recommendations for the modification of current waste management practices could be made.

1.5 ORGANIZATION OF THIS WORK

The present study is divided into seven chapters. Chapter 1 is a general introduction to the problems associated with lipid-rich wastes, and proposes aerobic composting as an alternative. The general aim of the study and the organization of the work are also presented in this chapter. Chapter 2 provides an in-depth literature review of previous works on composting, with emphasis on lipid-rich wastes composting and biodegradation using other processes.

Chapter 3 refers to the actual experiments and their results, using lab scale in-vessel composting technology on lipid-rich wastes. Empirical kinetics and thermal performance indicators are presented in this chapter.

The study of the environmental impact of degrading grease trap sludge using aerobic composting is summarized in Chapter 4; emphasis is put on ammonia, greenhouse gases, and odor emissions. Chapter 5 shows the effects of adding lipid-wastes during composting on the compost product. Compost quality was evaluated in terms of nitrogen content, lipids and organic matter degradation, and phytotoxicity effects.

A mathematical model for the simulation of kinetic and thermal behaviours of lipids degradation during composting is presented in Chapter 6. Finally, Chapter 7

summarizes the main findings of this research. Recommendations are presented concerning practical application of these findings.

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CHAPTER 2
REVIEW OF LIPID-RICH WASTES BIODEGRADATION

2.1 DEFINITION OF COMPOSTING

Composting is one of the more ancient of the agricultural arts. However, systematic studies of composting science and technology only began about 60 years ago (Golueke 1972). Processing methods, product evaluation and quality are still areas of debate, as are the rules and regulations governing the production and marketing of compost products.

In general, composting can be defined as an aerobic bioconversion in which the dissimilation of certain complex organic molecules (organic matter) and the assimilation (humification and mineralization) of new compounds take place. This bioconversion is carried out by a wide array of microorganisms with different life conditions and demands (Gajdoš 1997). The final product of composting is called "compost" (Gotaas 1956).

Composting is also defined as an engineered process that can be steered and regulated, but that usually is not set up in a way so as to fully decompose the organic matter present (Stentiford 1993). The final product, compost, may be in different degrees of maturation or stabilization depending on its final use.

Composting not only stabilizes organic matter, it also reduces the volume of material, kills plant and animal pathogens, reduces the carbon concentration, increases concentrations of plant nutrients and destroys organic compounds considered as environmental hazards. In general, composting essentially converts materials that are not land applicable into materials that are safe for land application (Barker 1997), with the exception of certain contaminants, such as heavy metals.

The different uses of compost are based on its different physical, chemical, and biological characteristics. Compost can serve as fertilizer, soil amendment, horticultural and potting substrate, plant disease suppressant, mulch, and landfill cover. Also, compost can be used in biofilters for odour removal, and for site remediation.

Horticultural or agronomic uses are usually based on nutrient content (chemical composition). Compost usually has a nitrogen content of 0.5 - 3% on a dry mass basis (db). According to Barker (1997), compost with a nitrogen concentration over 1% db may be used as fertilizer.

In the last few decades there has been a shift in the rationale of the composting process, from a waste management strategy to a resource recovery process. Notably, several authors have reported that a relatively low level of decomposition of organic matter can be associated with a high degree of lipid degradation, thus resulting in more organic matter conserved and converted to useful substances (Lemus and Lau 2001, Joshua et al. 1994, Fernandes et al. 1988, Viel et al. 1987a,b).

2.2 COMPOSTING PROCESS

The composting process is often divided in two periods: (1) a high-rate phase, also called an active period, and (2) a curing period. The high rate phase period is characterized by an intense microbial activity that results in a rapid increase of temperature, followed by a temperature stabilization period. After the easily degradable compounds are consumed (sugar, starch, lipids), the composting mix temperature gradually drops since the rate of heat biologically produced is then less than the rate of heat lost. The active phase usually ends whenever the temperature has dropped to the ambient temperature.

The curing period follows the active phase of composting, and is characterized by a very slow transformation of the organic matter. Curing often results in the mineralization of nutrients and humification of the organic matter. Figure 2.1 shows a common time-temperature pattern for the composting process.

Composting processes vary in duration depending on the technology used. The most common technologies are windrow, aerated static pile, and in-vessel composting. Windrow technology is perhaps the simplest and most traditional of the composting technologies, in which the composting mixture is placed in long narrow piles called windrows. Windrows might be turned periodically to improve the contact of the composting mix with air. Windrow composting is usually a very slow process with a duration between several months to 1-2 years.

The main feature of Aerated Static Pile, or ASP composting, is the provision of forced aeration via a combination of pipes and blowers. Usually ASP piles are not turned, and the process usually lasts for a few months. The fastest (and most expensive) composting technology is called 'in-vessel', where the composting mix is

completely enclosed, thus allowing for the utmost in process control and monitoring. In-vessel composting might last from several days to a few weeks.

Composting being a bioprocess is affected by any parameter that affects the microbial population, such as temperature, pH, presence or absence of oxygen, heavy metal content, and moisture content. Rynk (1992) recommends the optimum conditions, summarized in Table 2.1, to achieve rapid composting.

Composting temperature increases due to the energy released during the breakdown of complex organic molecules. Temperature also affects the microbial populations, giving preference to mesophilic species whenever the temperature range is from 25 to 50°C, and to thermophiles at temperatures above 55°C. Ceiling temperatures during composting are in the range of 65 – 70°C. According to Gray and Biddlestone (1971), thermophilic fungi, actinomycetes, and most bacteria become inactive at temperatures above 65 - 70°C.

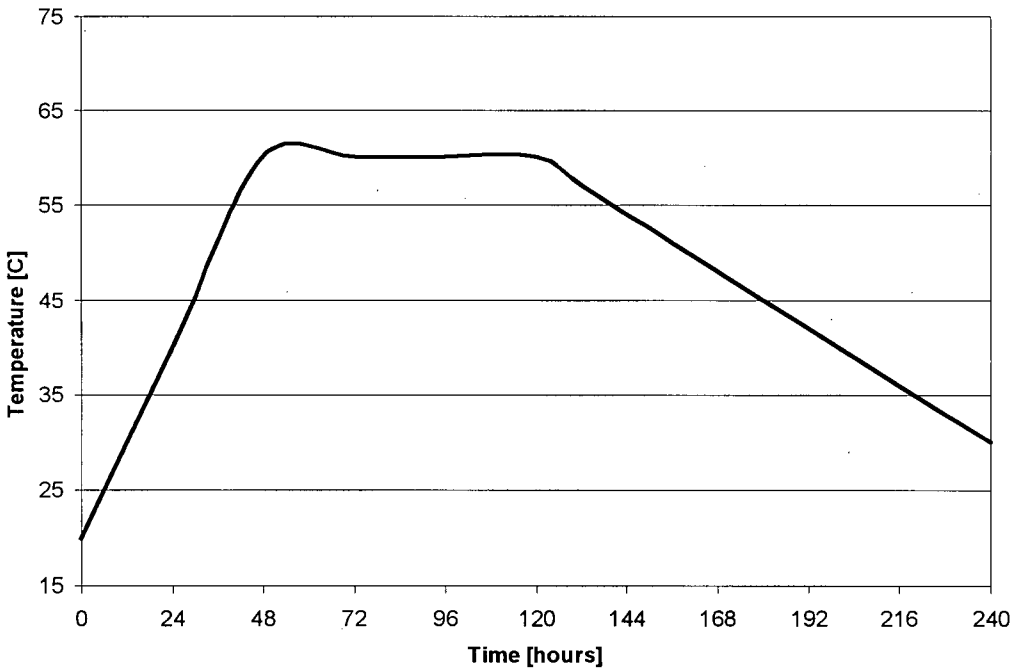

Figure 2.1. Typical time vs. temperature profile for in-vessel composting.

Table 2.1. Optimal conditions to achieve rapid composting (Modified from Rynk 1992).

Parameter	Optimal Range
Carbon-to-nitrogen ratio (C:N)	25 – 30
Moisture content	50 – 60 %
Oxygen concentration	> 5 %
pH	6.5 – 8
Temperature	55 – 60 °C

2.3 FATS, OIL, AND GREASE (FOGs) BIODEGRADATION

The main constituents of Fats, Oils, and Grease residues (FOGs) are animal fats and vegetable oils used in restaurants, and institutional/industrial operations. FOGs are essentially triglycerides consisting of straight-chain fatty acids attached, as esters, to glycerol. FOGs also comprise a combination of free fatty acids and glycerol whenever hydrolysis has taken place (See Figure 2.2).

Lipid residues include a broad variety of substances that share the common property of being soluble in various organic solvents (i.e. benzene, ethanol, chloroform, ether, etc.). These lipids include relatively simple compounds, consisting of long Carbon chains (even or odd) in the C_{16} - C_{32} range; such as fatty acids, n-alcohols, n-alkenes, sterols, terpenes, fats, waxes, and resins (Lefebvre et al. 1998, Wakelin and Forster 1997, Fernandes et al. 1988).

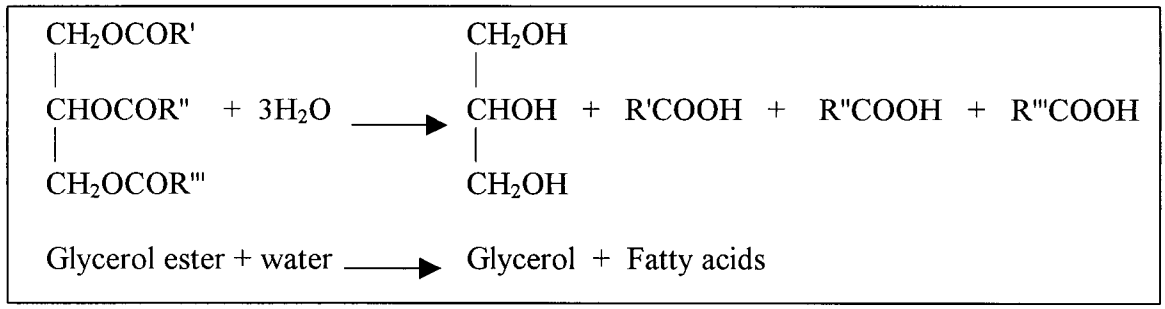

Figure 2.2. Fat hydrolysis reaction (Modified from McMurry 1992, Lawson 1985, Swern 1982). (R', R", and R'", refer to different fatty acids radicals)

Fats decomposition takes place in several steps, mainly fat hydrolysis and fatty acids oxidation. The use of lipases only results in fat hydrolysis, however the addition of microbial cultures results in fat hydrolysis and hydrolysis-by-products degradation.

According to Wakelin and Forster (1998), the attack of triglycerides by microorganisms is extracellular and takes place when ester bonds are hydrolyzed by lipolytic, hydrolytic enzymes (lipases).

Mulligan and Sheridan (1975) found that FOGs degradation takes place in two steps; first the emulsified FOGs are adsorbed to the biological floc, and later FOGs biological oxidation takes place. Among the advantages of FOGs biodegradation is the production of soluble fatty acids salts (soaps). These compounds are well-known emulsifying agents. The presence of emulsifying agents results in the solubilization of FOGs through micelle formation, thus improving the fluidity of these lipid residues. See Figure 2.3.

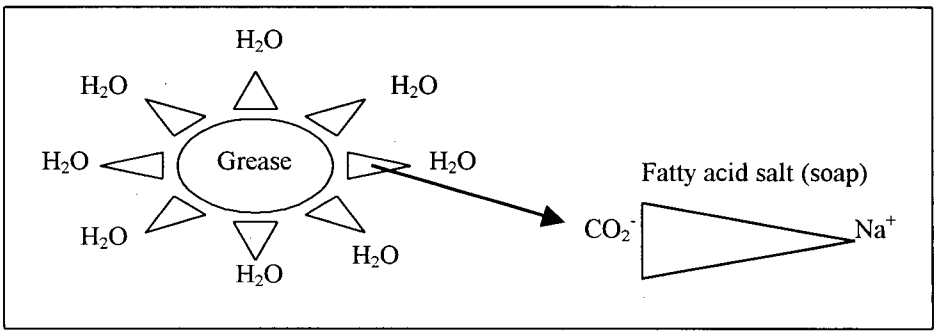

Figure 2.3. Solubilization of grease cluster by micelle formation (Modified from McMurry, 1992).

2.4 SOLID SUBSTRATE COMPOSTING FOR LIPIDS BIODEGRADATION

There are only few studies about the aerobic biodegradation of organic wastes loaded with lipid-rich wastes of vegetable and animal origin. Researchers have studied the feasibility of FOGs treatment of wool scouring wastewater, olive oil mill wastewater, food processing residuals, bakery wastes, and fast food restaurant wastewater.

Haug (1993) affirmed that lipids, or fats, are readily degradable in composting systems. Furthermore, Jakobsen (1994) stated that fats and oil resist biodegradation until they are hydrolyzed. Lefebvre et al. (1998) found that a lipidic mixture of about 80% fatty acids is easily degradable.

Lipids biodegradation in aerobic digestion has been high, with values above 85% (Keenan and Sabelnikov 2000, Becker et al. 1999, Wakelin and Forster 1997, 1998, Mahendraker and Viraraghavan 1994, Fernandes et al. 1988, Viel at al. 1987a,b, Wiley

1956). In contrast, biodegradation of lipids using anaerobic processes has achieved a maximum of 50-65% degradation (Gutierrez et al. 1999, Dinel et al. 1990).

Currently only a handful of composting facilities in the United States and Canada have reported on the testing or use of lipid-rich wastes (Kunzler and Roe 1995). One facility in Arkansas (Fisher 1997) uses flotation foams and grease trap wastes. In this facility the standard composting recipe (Carbon-to-Nitrogen Ratio: C:N = 30:1) is modified to achieve higher carbon levels and lower moisture content, and thus avoid either excess nitrogen or low oxygen conditions. Both of these latter conditions would produce foul odors. Another facility is located in Maui, Hawaii, and uses about 3-5.6% wet weight of commercial grease and cooking oils (Anonymous 1995).

A pilot research project was set up in Knoxville, Tennessee, to test the feasibility of co-composting grease trap sludge (GTS) with biosolids and yard trimmings. Mixes with GTS added easily fulfilled the temperature-time relation needed to achieve the pathogen reduction requirement. Odor from the GTS was noticeable in the raw materials storage area, but it decreased significantly when GTS was incorporated into the composting mixes, and through the composting process (Alpert et al. 2002, 2001).

Joshua et al. (1994) at La Trobe University, Australia, successfully composted grease trap sludge using an environmentally controlled composting system (a series of refrigerated shipping containers retrofitted for forced aeration). The use of supplemental nitrogen was not necessary because grease trap sludge had a nitrogen content of about 4% dry basis. The temperature of the composting mix (grease trap sludge, wheat straw, and wood shavings) peaked at 65°C and remained constant for the next 48 hours. After composting for 5-11 days the composting mix had no oxygen demand or reheating potential. The researchers concluded that a curing phase was not necessary.

Solid substrate aerobic composting was successfully used for FOGs degradation by a research group at the Ecole Nationale Supérieure Agronomique in Toulousse, France (Viel at al. 1987a,b, Fernandes et al. 1988). Fernandes et al. (1988) studied the feasibility of fatty wastes disposal through in-vessel composting. Solid substrate composting of lipid rich materials (flotation foams from wastewater treatment and slaughterhouse wastes) and other wastes (sawdust, corncobs) was carried out in a 100-liter stainless steel reactor. Their study found that process temperature (60-70°C) fulfilled pathogen reduction requirements. Key findings were that lipid breakdown was

not related to the initial fat content, and that a relatively low decomposition of organic matter could be associated with a high degree of lipids degradation.

Viel et al. (1987a) studied the aerobic composting of agricultural and industrial wastes. Lipid-rich wastes used were fats and slaughterhouse waste. The temperature peak was 75°C, and after composting for 7-9 days lipid concentration decreased from 9.3 to 1.4% dry basis (85% reduction). The lipid concentration after 120 days composting was 0.5% dry basis (95% reduction).

Viel et al. (1987b) researched the solid substrate composting of mixtures containing 8% wet weight of flotation foams at temperature-controlled conditions (60-76°C). The highest microbial activity took place at 60-70°C. Fat degradation was 85 %, and the energy released, mostly by fats, was estimated as 4180 kJ/kg dry matter per week (25 kJ/kg db.hr).

2.5 ADDITIONAL LIPIDS BIODEGRADATION SYSTEMS

Bilitewski et al. (1997) found that 5 days of liquid composting were comparable (in terms of results) to a total of 42 - 70 days (14 days in active phase plus 4 - 8 weeks in curing phase) of solid substrate composting. In addition, they found that a well-mixed liquid composting process, producing a hydrophobic phase emulsion, would favor lipase hydrolysis.

The main advantage of using liquid (or fluidized) composting was the short processing time due to increased reaction rate (high temperature). In addition, among other advantages, there was practically no odor release (in-vessel composting), and a high effectiveness of pathogen destruction and/or inactivation (Vik and Kirk 1993, USEPA 1990, Smith et al. 1975).

The following describes three examples of FOG degradation in liquid, aerobic treatments at bench or pilot scale. Wakelin and Forster (1998) tested the performance of a novel bioreactor (Weir tank reactor) for the treatment of fast food restaurant wastewater containing a low volume of grease. The Weir tank reactor achieved 84 - 96% FOG removal irrespective of the microbial inoculum, source of FOG, or water alkalinity. Their study suggested that a mixed culture, such as activated sludge, would be suitable for full-scale trials.

Becker et al. (1999) used a newly isolated microbial strain (*Bacillus thermoleovorans* IHI-91) for the biodegradation of olive oil and lipid-rich wool scouring

wastewater under aerobic, thermophilic conditions (65°C). Olive oil was almost completely removed (90%) in two hours. Fatty acids up to C_{18} were easily degraded, and the sterol fraction remained constant.

Keenan and Sabelnikov (2000) studied the feasibility of a biological augmentation approach to eliminate oil and grease from bakery wastewater. Reduction of oil and grease in the batch 4-L chemostat system was from 1.5 g/L to less than 0.03 g/L (98% reduction). For a comparison and summary of FOG biodegradation studies see Table 2.2.

2.6 COMPOSTING PROCESS PERFORMANCE CHARACTERIZATION

Composting process performance refers to the process efficiency to degrade fats, oil, and grease residues; the capability of self-heating to desirable temperatures without external heat addition; and the reductions in composting mix mass and volume, as well as water content.

2.6.1 Temperature Profile and Pathogen Reduction

In spite of their physical characteristics (i.e. slurry and oily), lipid-rich wastes have the advantage of being highly energetic residues. Lipids contain twice the energy of other organic components, like sugars and starch (Fernandes et al. 1988, Wiley 1957).

Thermophilic conditions are advantageous because favorable changes in most physical properties of FOG are temperature dependent. Specifically, thermophilic temperatures are usually above the lipids' melting points. Also, at high temperatures the reaction rates are faster, thus resulting in shorter residence times (Gariépy et al. 1989).

High temperatures and the presence of excessive amounts of water accelerate the hydrolysis of fats (Keenan and Sabelnikov 2000, Lawson 1985). Temperature affects organic wastes degradation because biological and chemical reactions, as well as solubilization and diffusion process, are temperature dependent.

An increase in temperature produces an increase in mass transfer due to reduced viscosity (in aqueous solution), a reduction in surface tension, and an increase in diffusion rate. Though higher temperatures increase the water solubility of solid and liquid substrates, the solubility of gases in both solid and liquid matrices (e.g. oxygen in solution) decreases (Sonnleitner 1983).

Table 2.2 Summary of previous studies on lipid-rich wastes biodegradation.

	Viel et al. 1987b	Fernandes et al. 1988	Joshua et al. 1994	Wakelin & Forster 1998	Becker et al. 1999	Keenan & Sabelnikov 2000
Substrate	Solid	Solid	Solid	Liquid	Liquid	Liquid
Feedstocks	Flotation foams, digested sewage sludge, sawdust	Flotation foams, slaughterhouse wastes, digested sewage sludge, sawdust	Grease trap sludge, straw, wood shavings	Fast-food restaurant wastewater	Olive oil and wool scouring wastewater	Bakery wastewater
Temperature	Controlled at 55-76°C	60-70°C (peak)	65°C (peak)	30 ± 2°C	65°C (peak)	--
Lipid degradation	85%	85-95% of initial dry matter	--	84-96%	90% olive oil, 20-30% wool scouring wastewater	98%
Process duration	6 - 8 days	8 - 10 days	5 - 11 days	33 days	2 hrs olive oil, 10-20 hrs wool scouring	12 hrs

-- Not reported.

Becker et al. (1997) stated that running a bioprocess at high temperatures has the following advantages: higher mass diffusion rates, increased solubility of lipids and other hydrophobic substances in water, and a reduced risk of microbial contamination.

Composting substrates can potentially carry human, animal, and plant pathogens (disease carrying organisms). Thermophilic composting has also the benefit of pathogen control (killing and/or inactivation) through heat inactivation.

The high-energy content of FOGs represents a clear advantage for processes where thermophilic degradation temperatures (above 50°C) are desirable to achieve pathogen reduction requirements. Becker et al. (1999) affirmed that thermophilic processes are hygienic operations.

Haug (1993) stated that in composting, the presence of undesirable biological agents (e.g. pathogens) is effectively controlled by maintaining elevated temperatures for a certain period of time. The combination of time-temperature conditions is a function of the composting technology used.

According to the B.C. Organic Matter Recycling Regulation (BCMWLAP 2002), the Composting Council of Canada (1998), and the U.S. Environmental Protection Agency 503 Regulation (1993), in-vessel composting should maintain a temperature equal or greater than 55°C for at least three consecutive days (72 hours) to ensure pathogen destruction. For Class A compost (other than biosolids compost), the vector attraction reduction requirement asks for an aerobic process of 14 days or longer with a temperature higher than 40°C, and an average temperature higher than 45°C (BCMWLAP 2002).

2.6.2 Lipids Biodegradation

Keenan and Sabelnikov (2000) and Becker et al. (1997) used microbial growth parameters to measure lipids degradation. However, according to Kramer (1971), an increase on biomass concentration may not produce an increase in lipid matter hydrolysis, because lipase production is not a function of cell growth or concentration.

Grego et al. (2000) studied the changes in the different apolar fractions (as extracted by hexane, dichloromethane, ethylacetate, methanol, and water) of grass clippings alone during composting. They found a reduction in all apolar fractions of more than 50%, and spectroscopic analysis indicated an increase in lipidic compounds (esters, acids, hydrocarbons) with longer linear aliphatic chains. Ouatmane et al. (2000)

found an increase in the aromatic to aliphatic structure ratio as aerobic decomposition proceeded.

2.6.3 Mass Changes during Composting

Soil organic matter includes plant and animal residues, the breakdown products of those residues, and a mix of compounds that are relatively decay resistant. Humus is the collective term given to the decay resistant fraction of organic matter (Brady 1990).

Composting refers to the stabilization process of the organic matter present in wastes. The value of compost as soil conditioner is based on its content of the highly stable, colloidal, soil organic fraction known as humus. The presence of humus is of great significance as storage of nutrients available to plants (Henis 1986).

Chemical oxygen demand (COD) is another measure of the degradability of organic matter. For liquid substrate systems, Keenan and Sabelnikov (2000) found a COD reduction of about 26% for a 98% decrease in oil and grease, although BOD was not significantly reduced during FOGs biodegradation. Becker et al. (1999) found a decrease in COD of 15-20% in a 10 - 20 hr composting process, and a correlation between COD, lipid concentration, and total suspended solids (TSS).

Decomposition of organic matter into a humus-like product is the main goal of composting. The different humic fractions of compost products have been studied using chemical partition-gravimetric methods, nuclear magnetic resonance (NMR), Fourier Transform Infrared Spectroscopy (FTIR), and differential scanning calorimetry (Ouatmane et al. 2000, Inbar et al. 1990, Saviozzi et al. 1988). Humic matter can be fractionated into about 10 different fractions (Stevenson 1965). Three of these fractions are of particular interest in composting: humin (HS), humic acid (HA), and the fulvic fraction (FF).

Ouatmane et al. (2000) found that humin and humic acid values increased with increasing composting time. This might be attributed either to the decrease in the fulvic fraction, to the progressive transformation of fulvic acids into humic acids, or to the biodegradation of readily decomposable compounds of the non-humic fraction.

Komilis and Ham (2000) tested a simple method to investigate gaseous emissions and solid decomposition during municipal solid waste composting. Dried and ground solids were sequentially analyzed for different organic compound groups. Hot water extraction was used to separate sugars, starch, and fatty acids, among others. A

mixture of toluene and methanol (2:1) was used to extract lipids, waxes, tannins, and part of protein and fulvic acids. Finally, sulfuric acid digestion and ashing were used to separate cellulose and hemicellulose, and lignin respectively.

The water content of the composting mix typically decreases over time, because any biologically produced water is usually less than the amount of water carried out in the exhaust gases. A large amount of water is lost as vapor during composting; thus composting also has a drying effect. A low moisture content (10 - 30%) is desirable for compost storage, transport, and packing.

2.7 COMPOSTING PRODUCT QUALITY MEASUREMENT

One of the major obstacles limiting composting as a resource recovery strategy is the lack of agreement and knowledge regarding end product (compost) quality. Compost quality is commonly defined as maturation and/or stabilization.

Stabilization, although commonly used as a synonym for maturation, refers to a product that has undergone complete bioxidation. Thus, a completely stabilized end product theoretically refers to a combination of carbon dioxide, water, and mineral ash (Haug 1993, Golueke 1972). Haug (1993) suggested a working definition of compost, in which compost is defined as a 'stabilized' product with humus-like properties. Organic residues might be in a 'pseudostable' stage when dried, though when rewetting takes place the biological activity will increase rapidly (Haug 1993).

Maturation is a functional definition. It can be defined as a relative degree of stabilization (i.e. the degree where compost will be mature enough to be used in a beneficial manner and to be safely stored). This beneficial use is a function of the particular application of the compost, with agronomic or plant usage being the most stringent among all the applications.

Compost for land application (agronomic use) needs to be "mature" enough in order to not exert oxygen demand, nor nutrient (especially nitrogen) demand from the soil. This oxygen and nitrogen demand will be exerted whenever the organic matter in the compost is still highly susceptible to biodegradation. In contrast, compost rich in stable organic matter, or humus, should not exert oxygen and/or nitrogen demand from the soil.

Mature compost should have a low salt content and a very low heavy metal content. Compost should also be free of weed seeds and plant pathogens, and should not have phytotoxic effects.

Nutrients in compost need to be present in a usable form for the plants (particularly nitrogen as nitrate instead of ammonia). Other good indicators of compost maturity are good physical properties, such as particle size distribution, texture, color, odor, and general appearance (Haug 1993). Additionally, the C:N ratio at the end of the processing time should be from 15 to 35. A curing period of at least 21 days is also required for stabilized compost (BCMWLAP 2002).

2.7.1 Volatile Solids Content

Organic matter is usually measured as the solids fraction oxidized at 550°C, called volatile solids (APHA 1995). The B.C. Organic Matter Recycling Regulation (BCMWLAP 2002) and the USEPA Regulation 503 (1993) require a value of more than 38% volatile solids reduction for biosolids composts that are going to be land applied.

During solid substrate composting of organic wastes loaded with flotation foams, Viel et al. (1987b) found that only 20% of organic matter, as volatile solids, were degraded. Fernandes et al. (1988) found organic matter reductions of 15 - 44% of initial dry weight, paired with lipids reduction of 85 - 95%. In comparison, the biodegradability of municipal solid waste has been reported as 35-38% (Atkinson et al. 1996).

2.7.2 Nitrogen Content and pH

The pH in composting operations changes from slightly acidic, to neutral or slightly basic at the end of the composting process. Since lipids would hydrolyze to form fatty acids, it is expected that the pH would drop to acidic conditions (Jakobsen 1994). Thus, the study and control of pH during lipids composting might be desirable.

Wiley (1956) found that slaked lime, $Ca(OH)_2$, addition during composting resulted in sharp increases in temperature to about 70°C in 1.5 days. Nonetheless, the use of alkaline compounds (e.g. lime) for pH control resulted in excessive nitrogen loss as ammonia (Jakobsen 1994).

Beccari et al. (1999) added lime to olive oil mill effluent to decrease inhibition in anaerobic digestion due to high concentrations of long-chain fatty acids. The decrease in inhibition was due to precipitation of the acids as relatively insoluble calcium salts.

The acids were later released back from the solid phase, thus increasing the methane yield.

Lefebvre et al. (1998) added potassium hydroxide, KOH, to domestic lipid residue (wastewater scum) in order to study the effect of saponification degree on the biodegradation of grease residues. After saponification the lipid residue was composted under pH controlled conditions. The result of this research indicated that saponified grease was biodegraded 3 to 4 times faster than raw grease.

The nutrient content of compost, particularly the nitrogen content determines the value of this product as fertilizer. Henis (1986) proposed that any material containing less than 1.2-1.6% nitrogen would cause nitrogen depletion when land applied. In contrast, products containing more than 1.6% nitrogen will increase the nitrogen availability in the soil.

Several authors (Mathur et al. 1993, Mason et al. 1992, USEPA 1990, Tyagi et al. 1990, Smith et al. 1975) found almost complete inhibition of nitrification at temperatures greater than 40°C. This is an advantage in aerobic composting because nitrification inhibition lowers the oxygen demand during the process. However, a disadvantage is that nitrogen oxidation (from ammonia to nitrate) is relatively poor. Nitrogen in the ammonia form might be lost easily when the temperature is in the thermophilic region and the pH of the composting mass is in the basic region (near or above 9) (Jakobsen 1994).

2.7.3 Lipids, Compost, and Soil

In the reviewed studies there were no evaluations of compost, as a product, when lipids were added to aerobic composting systems. Dinel et al. (1996a,b) studied the chemical changes in the lipid fractions of compost; specifically they tested the biodegradability/bioresistance of diethylether (DEE) extractable lipids and chloroform ($CHCl_3$) extractable lipids. Their results suggested that mature compost should have a ratio of <2.5 for DEE/$CHCl_3$ extractable lipids, and a ratio of >0.25 $CHCl_3$/Total lipid.

According to Dinel et al. (1990), lipid content appeared to be high in soils rich in humus, and inversely, low in soils poor in humus. The presence of lipids in soil has a positive effect on soil aggregation and aggregate stability, and a negative effect on water retention.

Furthermore, the addition of polar lipids improves structural stability, and the addition of nonpolar lipid compounds produces no improvement. In other words, the land application of untreated lipids would be of no benefit in terms of structural stability, but the application of hydrolyzed lipid compounds (i.e. fatty acids) would have a beneficial impact on soil structural stability. Structural stability is a determining factor in soil aeration, water infiltration and retention, all of which are important to plant growth (Dinel et al. 1990).

Plante and Voroney (1998) found that the addition of canola oil or oily food waste directly to soils was not responsible for improvements in soil structural properties; but the longer lasting binding abilities of microorganisms and their metabolism products were. Neuhauser et al. (1989) found negative effects with oily waste application directly into the soil, on the microarthropod and earthworm populations. The biomasses of these microorganisms did not recover to original levels for 2-3 years.

Soil lipids are usually 4-8% of the soil organic carbon, with higher values (up to 42%) for cultivated organic soils (Dinel et al. 1990). Chae and Lowe (1980) reported lipid content in British Columbian soils (Ah horizons) of 0.24 g/100 g oven-dry for delta-saline humic Gleysol, to 1.30 g/100g oven-dry for subalpine grass soil. Forest soils in B.C. (F and H horizons) had a lipid content of 1.61-5.33 g/100 g oven-dry soil. The addition of soil conditioners, rich in lipidic compounds, might be of benefit for B.C. soils.

2.8 ENVIRONMENTAL EMISSIONS

In this thesis research the air emissions studied were ammonia, greenhouse gases (carbon dioxide, methane, and nitrous oxide) and odour.

2.8.1 Ammonia Emissions

Most of the nitrogen losses during composting, particularly during the active phase, are a result of ammonia emissions (Martins and Dewes 1992). This ammonia loss has an impact on the agronomic value of the compost product, and it produces a harmful effect on the environment (Boucher et al. 1999). Moreover, Witter and Lopez-Real (1988) found that ammonia was the main odorant when composting mushroom substrate compost.

Ammonia losses account for 3 to 50% of total initial nitrogen, with concentrations of 30-1900 ppm in the composting exhaust gases. (Morand et al. 1999, Sommer and Dahl 1999, Mahimairaja et al. 1994, Martins and Dewes 1992, Witter and Lopez-Real 1988, Godden and Penninckx 1986). Witter and Lopez-Real (1988) showed that virtually all the ammonia losses occurred during the initial stage of composting (high-rate phase). In the high-rate phase of composting nitrogen-rich materials, like proteins, are transformed by biological and chemical reactions. This decomposition is accompanied by a high rate of ammonification (Bishop and Godfrey 1983).

Moller et al. (2000) reported that the gas phase ammonia concentration inside a compost heap ranged from 20 to 200 mg NH_3-N/m^3. According to Ekinci et al. (2000) ammonia losses decreased rapidly below pH 7, and increased rapidly when the pH value was greater than 8. This is supported by the chemistry of ammonia, since the ammonium ion is more prevalent whenever pH values are below 9, and conversely ammonia is more common at basic pH (Jakobsen 1994, Court et al. 1964).

2.8.2 Greenhouse Gas Production

Nitrous oxide (N_2O) and dinitrogen gas (N_2), have presumably accounted for as high as 40% of the total initial nitrogen losses during composting (Moller et al. 2000, Pel et al. 1997, Kuroda et al. 1996, Mahimairaja et al. 1994).

Nitrous oxide (N_2O) is a natural by-product of nitrification (aerobic microbial oxidation) and denitrification (anaerobic/anoxic microbial reduction). Nitrous oxide is an intermediate in the oxidation process from NH_4^+ to NO_2. Delwiche (1981) stated that under low oxygen supply conditions, nitrifying organisms might exclude competitive organisms by liberating a compound (N_2O) that is not readily available for further nitrification, thus avoiding the overuse of the limited oxygen supply.

He et al. (2000) reported that the N_2O generation during composting is largely due to denitrification, and that this gas production should be proportional to the available carbon. The production of N_2O is associated with anoxic and anaerobic microsites in the composting mix, thus biological denitrification has been found to occur even at oxygen levels as high as 15% (Hao et al. 2001, He et al. 2000, Hwang and Hanaki 2000).

The nitrogen losses as N_2O have been reported as 0.5% of total initial nitrogen for poultry manure and poplar bark compost (Morand et al. 1999). Hellebrand (1998) reported that when farm waste was used (bedding plus horse/poultry manure) the

nitrogen losses as N_2O were 2.2% of total initial nitrogen, and in the case of yard wastes (grass clippings and fallen leaves) the N_2O losses amounted 1.2% of total initial nitrogen.

For food waste composting, N_2O's peak value was 10 ppmv for the first week, with relatively high amounts of N_2O at the beginning of composting, proportional to the amount of food waste. Nitrous oxide emissions decreased to near atmospheric values after 2 days (He et al. 2000).

Carbon dioxide (CO_2) generation is an indication of microbial metabolism. Carbon dioxide is a result of the biological degradation of carbonaceous substrates, like sugars, starches, and lipids. Peak CO_2 values occur in the first few days of composting, and carbon losses, as CO_2, have been reported as 8 - 22% of total initial carbon (Morand et al. 1999, Hellman et al. 1997).

The presence of methane is an indicator of anoxic and anaerobic pockets in the composting matrix. Methane has been found during composting even when the oxygen content on the composting exhaust gases was no less than 15%. Methane peak values in the composting exhaust gases occurred in the active phase of composting with values of 3 ppmv for food waste, and around 500 ppmv for food waste with cattle manure added (He et al. 2000).

2.8.3 Odor Generation

Composting, being an aerobic biological oxidation of organic matter, involves the production of gaseous products. Fatty acids, particularly long chained, monocarboxylic acids, will be produced in the biodegradation of lipids. These compounds have been listed as potential significant odorants in composting processes (Haug 1993). None of the studies previously mentioned researched the impact of lipids biodegradation on odor generation.

During aerobic composting, many low molecular weights odorous compounds are produced. Among them, ammonia (NH_3), and acetic and other volatile fatty acids (VFAs). If the process tends to be anaerobic or anoxic, hydrogen sulfide (H_2S), volatile organic acids, mercaptans, and methyl sulfides, might also be produced (Haug 1993).

The amount of odorants released is closely related to a number of composting parameters. Among these parameters are the following: amount of material handled, flow of oxygen, aeration type (positive or negative), process temperature, geometric

composter design and age of compost pile, and the particular composting system used (Bidlingmaier 1993). High odorant emissions are expected during the high rate phase of composting (Day et al. 1998, Bidlingmaier 1993, Benedict et al. 1988).

Lau et al. (1996) found that the amount of ammonia emitted from in-vessel compost reactors had a positive correlation with the percentage of compost mass at a temperature greater than 60°C. Fraser and Lau (1998) found that a higher aeration rate would lead to lower odor concentrations, although higher mass air emission rates would be produced. Conversely, lower aeration rates would produce lower gas emissions and higher odorant concentrations.

Odor control involves odor measurement. Physical and chemical odor detection methods include gas chromatography and mass spectrometry (GC/MS). These methods are helpful with the detection of particular chemical compounds. However, a mix of compounds characterizes most composting odor emissions, and many of these are unknown or difficult to measure. Olfactometry using the human sense of smell is still the most useful, and directly related to receptors, way of measuring odors (Berglund et al. 1987).

Since the perception of odor by humans is subjective, lately the use of electronic noses has been preferred (Nicolas et al. 2000, Krzymien and Day 1997). The development of an electronic nose has been possible through the development of multiple chemical compound detectors. The electronic nose is a very promising technology for the measurement of complex odorant mixes.

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CHAPTER 3
BIODEGRADATION OF LIPID-RICH RESIDUES DURING COMPOSTING

3.1 ABSTRACT

Changes in food production, consumption patterns, and more restrictive regulations, result in larger quantities and a higher diversity of waste materials that need to be treated and disposed. Organic wastes rich in lipidic compounds are not typically composted. However, fats, oils, and grease (FOGs) residues have a high-energy content that should make them ideal candidates for aerobic composting.

The aim of this study was to evaluate the treatability of lipid-rich residues when composted under solid substrate aerobic conditions. Composting process performance was evaluated by measuring kinetic (rate of mass and lipids biodegradation), as well as thermal parameters (temperature profiles, energy produced).

A series of preliminary tests was performed using canola oil as a sample lipidic compound. A second series of tests used grease trap sludge (GTS) as a lipid-rich residue. Grease trap sludge, mainly from restaurants grease traps, was sampled from a vacuum collection truck.

Two different substrates were used: yard trimmings (mainly grass clippings), and synthetic food waste (dry dog food). Dry dog food was used to simulate food waste since its particle size is very uniform, and its composition is very similar to actual food waste. Hemlock wood shavings and wood chips were used as bulking agents.

Results indicated that during the high-rate phase of composting, canola oil concentration was reduced by 48-62% and 11-79%, for yard trimmings and synthetic food waste, respectively. Grease trap sludge was degraded by 39-51% for the yard trimmings experiments, and 10-27% for the synthetic food waste treatments during the high rate phase. In addition, the biodegradation of volatile solids varied from 6 to 24% for the canola oil treatments, and from 7 to 21% for the grease trap sludge ones. During curing, the lipids reductions were 11-48% and 22-50% for the treatments with either yard trimmings or food waste, with grease trap sludge added.

Addition of lipids seemed to have a very marked effect on the temperature profiles. With few exceptions, treatments with lipids added performed in the thermophilic

range. Treatments with lipid addition generated between 9.9 to 39.8 kJ/kg ds.hr in the case of yard trimmings, and 5.4 to 23.1 kJ/kg ds.hr for food waste. These values have the same magnitude as the energy release value of 25 kJ/kg ds.hr reported in the literature. Treatments with canola oil added resulted in more heat generated when compared with treatments with grease trap sludge added.

One of the unique contributions of this study was the measurement of the biodegradation rate coefficient (k) for lipid-rich wastes, as well as the 'k' values when these residues were added to composting mixes. The units of the biodegradation rate coefficient are kg BVS.(kg BVS.day)$^{-1}$, and are commonly referred to as 'day^{-1}'. The volatile solids biodegradation rate coefficient for the canola oil treatments was 0.009-0.039 day^{-1}, and the one for grease trap sludge treatments was 0.009-0.033 day^{-1}. For the active phase of composting, the average biodegradation rate coefficient for canola oil was calculated at 0.117 day^{-1}, and the one for grease trap sludge alone was calculated at 0.058 day^{-1}. Values for the biodegradation rate coefficient for composting mixes with lipid residues added, and for biodegradation of canola oil and grease trap sludge, have not been reported anywhere else in the literature.

As a practical recommendation, the addition of grease trap sludge up to 5% ds to yard trimmings composting mixes, would result in an enhanced temperature profile, an improved biodegradation rate and extent of volatile solids and lipids, and an overall larger reduction of wet mass and water content; when compared with the composting of yard trimmings alone.

3.2 INTRODUCTION

Fats, Oils, and Grease (FOGs) residues are considered problematic substances in both liquid and solid waste treatment systems. Some countries, like France, have banned their disposal in sanitary landfills due to the biodegradable nature of FOGs (Lefebvre et al. 1998).

The main constituents of Fats, Oils, and Grease residues (FOGs), or lipid-rich wastes, are animal fats and vegetable oils used in restaurants, institutions, and industrial operations. FOGs are essentially triglycerides consisting of straight-chain fatty acids attached, as esters, to glycerol. FOGs also comprise a combination of free fatty acids and glycerol whenever hydrolysis has taken place (Jakobsen 1994).

FOGs residues include a broad variety of substances that share the common property of being soluble in various organic solvents (e.g. hexane). These lipids include relatively simple compounds, with long carbon chains (even or odd) in the C_{16} - C_{32} range; such as fatty acids, n-alcohols, n-alkenes, sterols, terpenes, fats, waxes, and resins (Lefebvre et al. 1998, Wakelin and Forster 1997, Fernandes et al. 1988).

Composting has proven to be a very successful treatment process for mineral oil residues (hydrocarbons), and residues from oil extraction processes (e.g. olive oil) (Filippi et al. 2002, Wan et al. 2002, Kirchmann and Ewnetu 1998, Cegarra et al. 1996). However, composting of organic residuals, such as food wastes, rich in oil and grease is not a common practice.

Lipids contain twice the energy of other organic materials, like sugars and starches (Fernandes et al. 1988, Wiley 1957). This high-energy content represents a clear advantage for composting or other aerobic treatment processes where regulations require thermophilic temperatures to achieve effective pathogen reduction. High temperatures also induce faster reaction rates and hence shorter residence times (Gariépy et al. 1989, Pöpel and Ohnmacht 1972). Moreover, thermophilic conditions result in favourable changes in most physical properties of FOGs, such as melting point, diffusivity, and solubility (Becker et al. 1999, LaPara and Alleman 1997).

A pilot research project was set up in Knoxville, Tennessee, to test the feasibility of co-composting grease trap sludge (GTS) with biosolids and yard trimmings. Mixes with GTS added easily fulfilled the pathogen reduction requirement. Odor from the GTS was noticeable in the raw materials storage area, but it decreased significantly when GTS was incorporated to the composting mixes, and through the composting process (Alpert et al. 2002, 2001).

Viel et al. (1987a) studied the aerobic composting of agricultural and industrial wastes. The lipid-rich wastes used were fats and slaughterhouse waste. Fats degradation was 80% and the temperature peak was 75°C. After composting for 7-9 days the lipid concentration decreased from 9.3 to 1.4% dry basis (85% reduction), and the lipid concentration after composting for 120 days was 0.5% dry basis (95% reduction).

With temperature controlled conditions (55-76°C), Viel et al. (1987b), further researched the solid substrate composting of mixtures containing 8% wet weight of flotation foams. The highest microbial activity took place at 60-70°C. Fat degradation

was 85%, and the energy released, mostly by fats, was estimated as 4180 kJ/kg dry matter per week (25 kJ/kg db.hr).

Fernandes et al. (1988) studied the feasibility of fatty wastes disposal through in-vessel composting. Solid substrate composting of lipid rich materials (flotation foams from wastewater treatment and slaughterhouse wastes), and other wastes (sawdust and corncobs) was carried out in a 100-L stainless steel reactor. This study found that process temperature (60-70°C) fulfilled the pathogen reduction requirement, and that a relatively low decomposition of organic matter could be associated with a high degree of lipid degradation.

Joshua et al. (1994) successfully composted grease trap sludge using an environmentally controlled composting (ECC) system. The temperature of the composting mix (grease trap sludge, wheat straw, and wood shavings) peaked at 65°C, and remained constant for the next 48 hours. After composting for 5-11 days, the composting mix had no oxygen demand or reheating potential. The researchers concluded that a curing phase was not necessary.

More rigorous organic matter disposal regulations and increasing recycling goals present a need to find treatment options for organic wastes, particularly the ones not previously treated by aerobic biodegradation. The aim of this study was to evaluate the treatability of organic wastes loaded with lipid-rich residues under aerobic composting conditions. In particular this research's purpose was to study composting process performance in terms of temperature profiles, fulfillment of the pathogen reduction requirement, lipids compostability, and organic matter biodegradation. Furthermore, the effects of different inoculum types, as well as concentration were examined.

3.3 MATERIALS AND METHODS

3.3.1 Experimental Lab Set Up

Bench-scale composting trials were conducted at the Waste Management Pilot Plant of the Chemical and Biological Engineering Department, at the University of British Columbia. The details on composting reactors configuration, analytical procedures, as well as feedstock preparation and recipe formulation are described in the following sections.

3.3.1.1 Experimental configuration

For the high-rate composting phase, the bioreactors used were 6-L stainless steel Dewar flasks (Cole Parmer Instruments Company, Vernon Hills, IL). The composting vessel was placed inside an insulated (adiabatic) box in order to simulate typical in-vessel composting process without agitation, or the core region of a compost pile.

In this research for simulation of the in-vessel composting process, the insulation of the composting reactor was very important since the small quantity of materials used (1.5-2.5 kg) did not constitute a critical mass, for a noticeable temperature rise unless the composting reactor was well insulated (since composting mixes have insulating properties, Mears et al. 1975). For this purpose, the composting reactor was placed inside an insulated box made out of R-5 Styrofoam board, and the space in between the reactor and the box was filled with R-28 roof insulation (Owens Corning, Toledo, OH). Furthermore, the Dewar flask was wrapped with a custom made Reflectix® sleeve (heating tank insulation, Reflectix Inc., Markville, IN).

Temperatures inside the composter and ambient temperature were monitored hourly using copper-constantan thermocouples. The Dewar flasks had a double-wall, with a vacuum in between, that gave them thermos-like characteristics, so that heat losses were minimized.

The temperature data was collected using a data acquisition board (Advantech PCL-711S) and a PC. Fulfillment of the pathogen reduction requirement was assessed from the composting temperature profiles. Figures 3.1 and 3.2 show the schematics of the experimental lab set up used for the high-rate phase study. Three identical Dewar flasks were available for use in this study. The different experimental treatments were randomly assigned to any of the 3 reactors used.

For the curing period, the composting reactors were 4-L plastic buckets, with no provision for forced aeration, mixing, or continuous temperature monitoring. Furthermore, the curing reactors were not covered and were open to the ambient air.

3.3.1.2 Process control

Composting process temperature control was attained by using Labtech Control™ Software (V.9.02). The aeration strategy used, which affects the temperature profile, was the industrial standard (Rutger's Method), where aeration is intermittent (at 33% duty cycle, specifically the pumps were 'on' 1 minute, and 'off' 2 minutes, in a 3

minutes aeration cycle) below the temperature control set point (e.g. 70°C), and continuous above the temperature set point.

Aeration was provided using two diaphragm (aquarium) pumps (one large, Model 'Optima', and one small, Model 'Elite 802', Rolf C. Hagen Inc., Montreal, QC). Airflow was manually adjusted via the large pump speed dial or by controlling the tightness of the screw-type clamps on the air tubing until the desired airflow reading was indicated in the air flowmeter (Gilmont No.13, Gilmont Instruments, Barrington, IL). The pump set up and flowmeter were previously calibrated by Fraser (1997). Airflow reading was checked daily and the airflow was manually adjusted if necessary. For more detail on the experimental configuration see Figure 3.3.

Airflow was set at 0.72 lpm/kg of initial dry solids, unless otherwise noted, as suggested by Rynk (1992). Air entered the composting vessel from the bottom and was exhausted from the top area (upflow direction). A metallic mesh at the bottom of the reactor provided support for the composting mass, and an aquarium air diffuser located below the mesh distributed the incoming air.

Oxygen content in the exhaust gases was monitored, for the grease trap sludge treatments only, by using an oxygen probe and a controller (Model 1630, accuracy ± 0.1%, Engineered Systems and Designs Inc., Newark, DE). Oxygen was monitored daily during one aeration cycle (air pump 'on' for 1 minute, and 'off' for 2 minutes). The minimum oxygen concentration for the 3-minute aeration cycle was recorded manually once a day.

Relative humidity of the air in the lab where the composting set up was placed, as well as the relative humidity of the exhaust gases from the composting reactors was monitored daily (Model 4085 Traceable Hygrometer/Thermometer/Dew Point Meter, accuracy ± 1.5%, Control Company, Friendswood, TX). The relative humidity of the lab was fairly constant at 24 ± 1%, while the relative humidity of the exhaust gases from the composting reactors was always at saturation (100%).

The composting's high-rate phase was deemed to be finished whenever the composting mix temperature had dropped back to ambient level. For most of the experimental treatments this occurred within 168 hours (7 days). Thus, the duration of the high-rate phase for all treatments was chosen to be 168 hours, and data in the form of kinetic and thermal parameters were collected during this period.

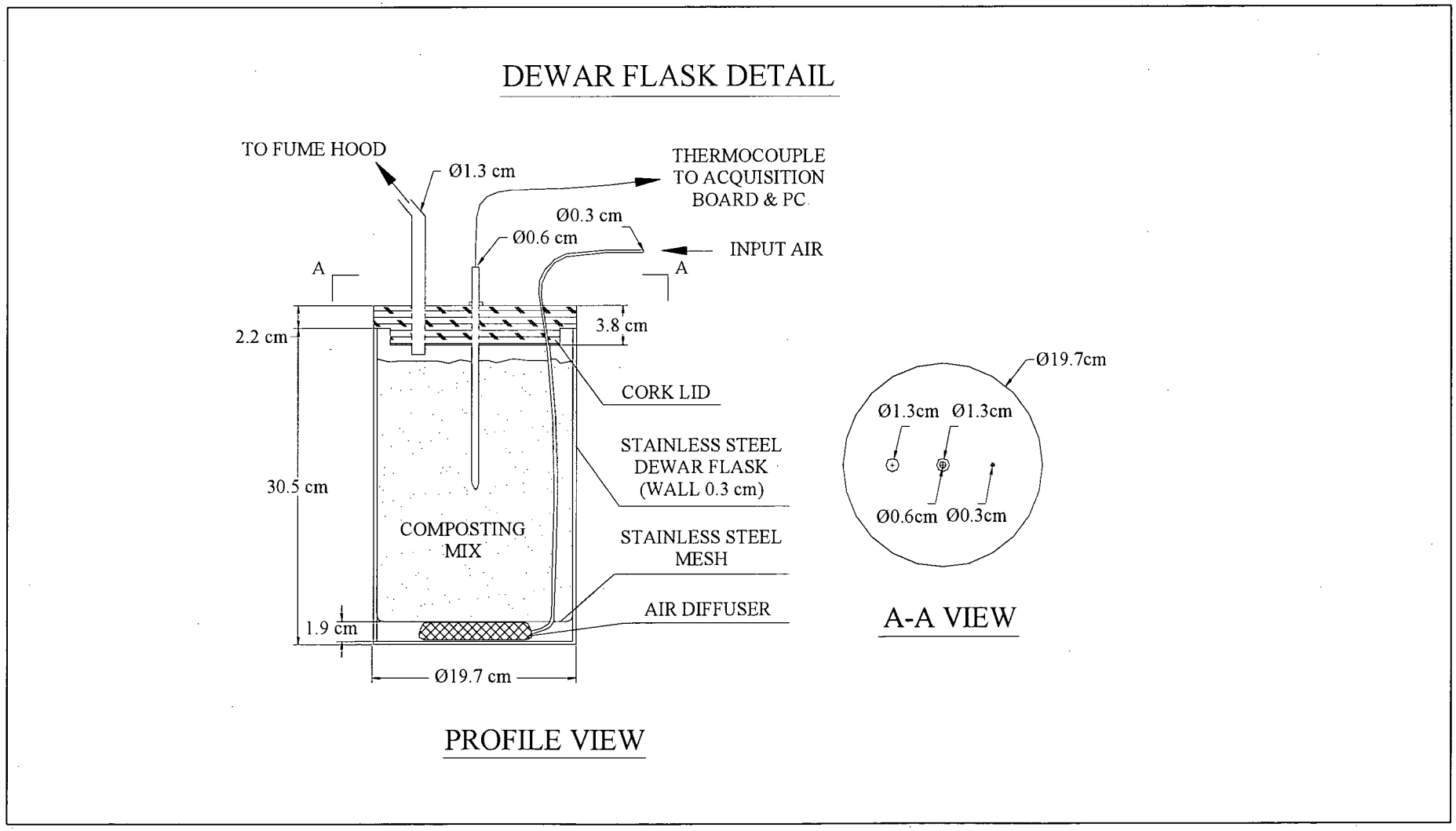

Figure 3.1 Dewar flask and cork lid details.

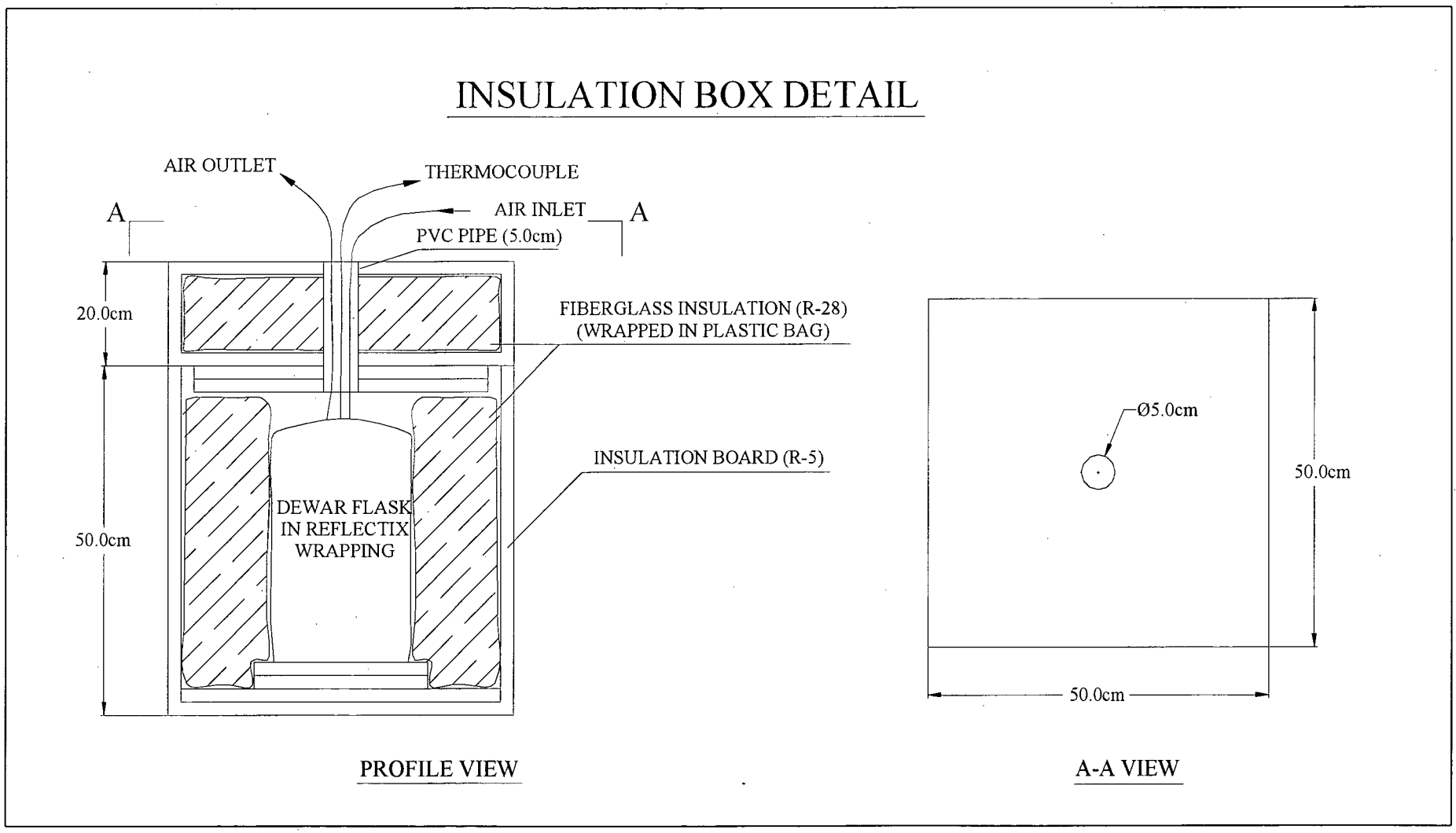

Figure 3.2 Adiabatic box details.

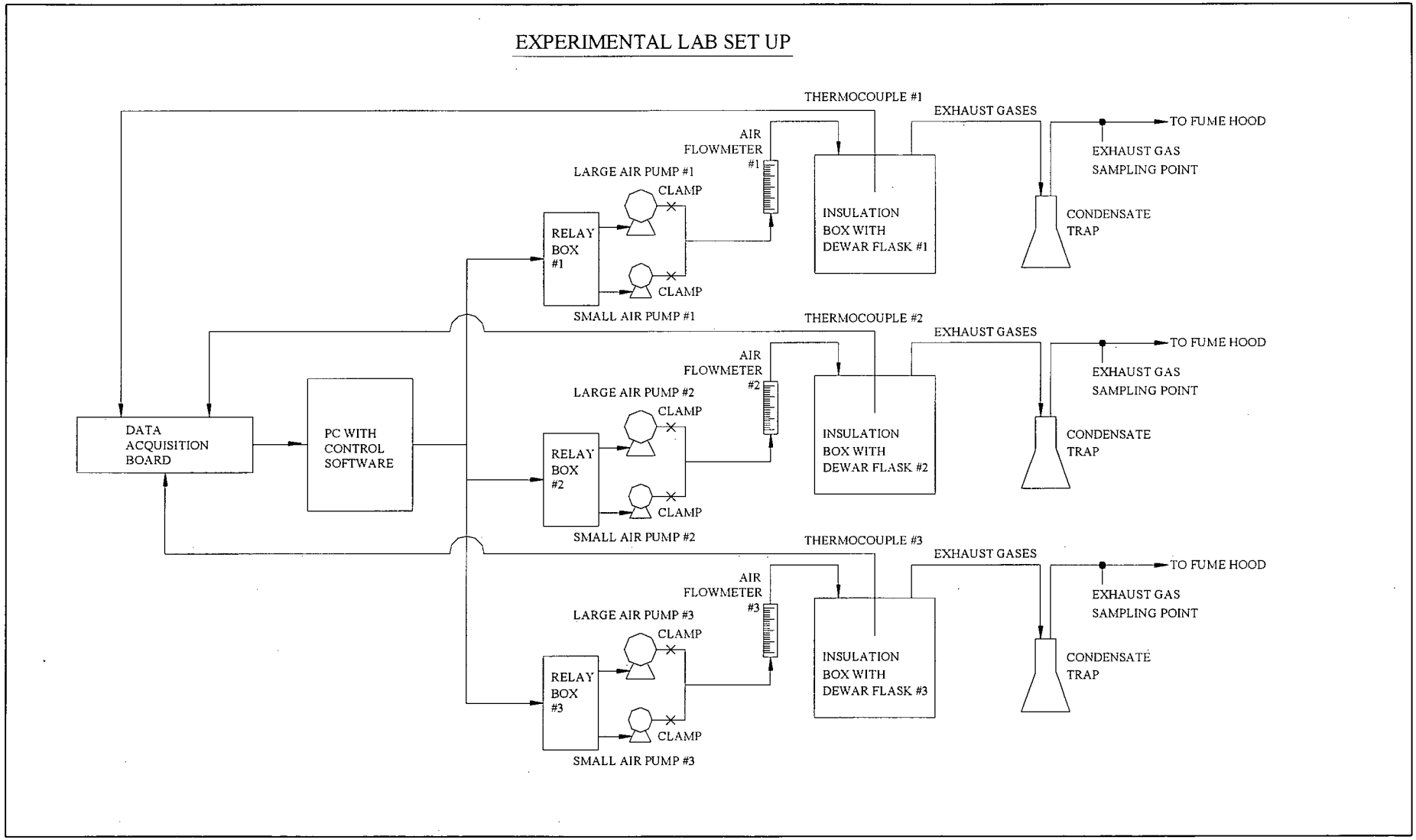

Figure 3.3 Experimental lab configuration and airflow direction.

After 168 hours from the start of the composting experiments, the composting material inside the Dewar bioreactors was mixed and sampled for physico-chemical analyses. For the treatments with grease trap sludge, the immature compost was then transferred to the plastic reactors, for curing over a period of 126 days.

3.3.1.3 Analytical measurements

Total composting mass was measured gravimetrically (Balance OHAUS I-10, Ohaus Corporation, Florham Park, NJ) before and after composting. Moisture content was measured by gravimetric analysis and oven drying at 101°C for 18-24 hours, according to APHA 2540B (APHA 1995). Volatile solids were measured by gravimetric analysis and ash content (ignition at 550°C for 2 hours, test accuracy ± 6.5%, APHA 2540E, APHA 1995).

Carbon content was calculated using the 'New Zealand' formula (Equation 1, as quoted by Haug 1993), originally proposed in 1951 and derived for organic wastes, which is still commonly used in the field of composting for estimating the percentage of carbon in the composting mixes (McCartney and Chen 2001, Tomati et al. 2001, Haug 1993, Rynk 1992, Schulze 1958). According to Haug (1993), the use of Equation 1 for composting mixes gives carbon content results with 2-10% accuracy.

Carbon concentration was also measured, for the grease trap sludge treatments only, by using a CN Carlo Erba NA-1500 Analyzer (accuracy ± 0.3%). On average, the carbon concentration thus measured differed from that calculated using Equation 1 by 10% (See Appendix B for more detail).

$$\%\ Carbon = \frac{100 - \%\ Ash}{1.8} = \frac{\%\ Volatile\ Solids}{1.8} \qquad (1)$$

Total nitrogen content in the composting materials and the cured compost was measured using a Total Nitrogen analyzer (either a LECO FP228 Nitrogen Determinator, Leco Corp., St. Joseph, MI, accuracy ± 2.0% of the measured N value; or a CN Carlo Erba NA-1500 Analyzer, accuracy ± 1.6%). Moisture content, volatile solids,

and total nitrogen concentration were measured on 3 subsamples (n=3) of the materials tested.

For the lipids concentration analysis, approximately 20 g of sample were acidified (with concentrated hydrochloric acid) and chemically dried over sodium sulfate. After drying for 30 min, the lipids were extracted by n-hexane as the organic solvent using a Soxhlet extraction apparatus. The extraction procedure took 4 hours with solvent recirculation at 20 cycles per hour. Thereafter, extraction the n-hexane was distilled to recover the solvent, and the residue was desiccated and weighted. Lipid testing was performed in 2 replicates (n=2) per sample. The materials extracted in this analysis are commonly called Hexane Extractable Materials (HEM), and might include relatively non-volatile hydrocarbons, vegetable oils, animal fats, waxes, soap, greases, and related matter (test accuracy for restaurant wastes is ± 8%, USEPA 1998). The amount extracted by the solvent, before and after composting, was compared in order to determine changes in the lipids concentration due to aerobic biodegradation.

Bulk density of the composting mixes was measured by the mass-per-volume technique, following Rynk's (1992) procedure, whereby a container of known volume was filled up with the composting mix, special care being taken not to overcompress the mixture.

3.3.1.4 Feedstock characterization

Two different substrates were used: Yard trimmings (mainly grass clippings), and synthetic food waste (dry dog food according to VanderGheynst et al. 1997). The dry dog food used was No Name®, Special Dinner-Packet Club (Sunfresh Limited, Toronto, ON), with 'as is' composition, according to the label, of 21 % protein, 7% fat, 5% fibre, 7.5% ash, and 12% moisture.

Generic brand canola oil (No Name®, Canola Oil, Sunfresh Limited, Toronto, ON), or grease trap sludge, was added as the lipid-rich fraction. Grease trap sludge (GTS) was sampled from a vacuum collection truck that gathered sludge mainly from restaurants grease traps.

The GTS sample was collected as a composite of the contents of all the sections of the truck's tank, meaning that the sample was approximately 33% from the top, 33% from the middle, and 33% from the bottom of the tank. This was accomplished by directing the truck discharge hose to the sample container (during similar time intervals)

at the beginning, middle, and end of the discharge procedure. The discharge of the different tank areas was achieved by lifting the tank to different heights. Figure 3.4 shows a vacuum collection truck with the tank lifted, and Figure 3.5 shows the different phases of a GTS sample.

Lipid measurement for the grease trap sludge was a very challenging task since GTS was mostly water. Thus, the lipid measurement protocol using vacuum drying (AOAC Official Method 906.12, AOAC 2000) required modifications (details are shown in Appendix C). The grease trap sludge had a measured lipid content of 31 ± 26% wb, a moisture content of 61 ± 10% wb, and a nitrogen content of 0.3 ± 0.1% ds; these parameters values were similar to the values reported by Plante and Voroney (1998).

Hemlock wood shavings and wood chips were used as bulking agents. Urea (Urea 46-0-0, Evergro Products, Delta, BC) was added to adjust the carbon-to-nitrogen ratio (C:N) for the yard trimmings treatments only. Either chicken litter (retail quality, Keefer's Greenhouses Inc., Richmond, BC), or activated sludge (from the UBC Civil Engineering Dept. pilot wastewater treatment plant), was used as inoculum.

Table 3.1 indicates the characteristics of the composting feedstocks, including moisture content, carbon and nitrogen content, and lipid content as hexane extractable materials (HEM).

3.3.1.5 Composting recipe formulation and feedstock preparation

In order to manage the composting process properly it is necessary to set the appropriate initial conditions and key process parameters to the optimum range (Richard et al. 2002). Optimum composting parameters were derived using a 'Composting Recipe Formulation Worksheet' (Excel 2000® software, Microsoft, Redmond, WA), and summarized in Table 2.1 (Chp. 2, pp.11). A sample of the composting recipe calculation worksheet may be found on Appendix D.

The composting recipe formulation took into account the properties of the different feedstocks (moisture content, carbon and nitrogen content, and lipid content) to achieve the composting mix characteristics. The proportions of each feedstock were changed manually until the desired moisture content, initial C:N ratio, and lipid concentration of the composting mix were achieved.

Figure 3.4 Vacuum collection truck discharging at the Iona Wastewater Treatment Plant.

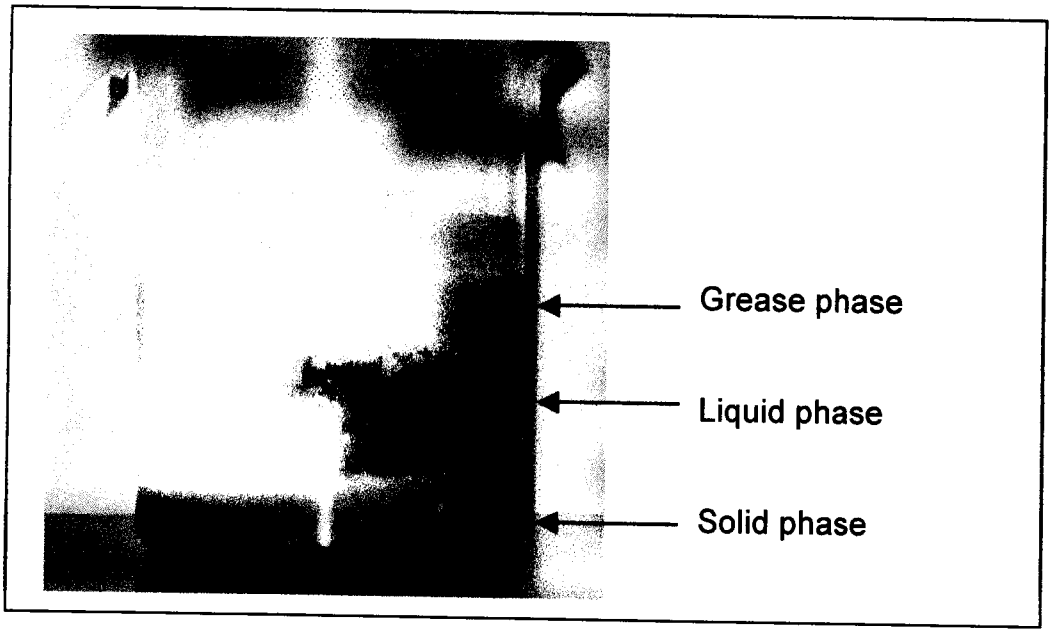

Figure 3.5 Grease trap sludge (GTS) sample.

Table 3.1 Characterization of individual feedstocks used for the lipid-rich composting trials.

Feedstocks	%mc	%vs	%C	%N	%HEM*	Comments
Yard trimmings	72.8 (1.94)	90.0 (0.28)	49.9 (0.15)	4.7 (0.06)	0.7 (0.15)	Mainly grass clippings, collected from UBC lands or from a residential source, during late Spring or early Summer.
Synthetic food waste‡	7.6 (0.18)	92.2 (0.16)	51.2 (0.09)	3.0 (0.24)	5.4 (0.2)	Dry dog food. No Name® Brand, Special Dinner Packet Club.
Canola oil	0.0	99.9 (0.01)	76.9	0.1 (0.02)	100.0	No Name® Brand, Canola Oil.
Grease trap sludge†	61.1 (10.0)	98.6 (0.2)	54.8 (0.1)	0.3 (0.1)	35.0 (26.0)	Composite sample from a collection truck. Trucked Liquid Waste (TLW) unit at Iona Wastewater Treatment Plant (IWWTP), Vancouver, B.C.
Woodchips	11.8 (0.17)	95.1 (0.51)	52.9 (0.28)	0.4 (0.06)	0.9 (0.05)	Bark in the shape of slivers and nuggets (hog fuel). Approximately 2-3" on largest side. Obtained from a wood mill.
Wood shavings	7.7 (0.25)	99.4 (0.01)	55.2 (0.01)	0.4 (0.12)	1.2 (0.05)	Clean hemlock wood shavings from the UBC barns, Animal Sciences Department (South campus).
Chicken litter	53.0 (3.38)	77.5 (0.66)	43.1 (0.36)	3.1 (0.13)	0.4 (0.15)	Mix of weathered chicken manure and bedding (wood shavings). Keefer's Greenhouses Inc.
Activated sludge	95.0 (0.19)	60.5 (0.2)	33.6 (0.1)	5.6 (0.25)	--	From the Civil Engineering Pilot Plant trailer (South campus).
Urea	1.2 (0.28)	99.8 (0.02)	20.0	46.7	--	Urea 46-0-0. Evergro Specialty Fertilizers.

All units are dry basis (d.b.), except for %mc and %HEM, which are wet basis (w.b.).
Figures in parenthesis are standard deviations. Parameters were measured on 3 subsamples , except for %HEM measured on 2 replicates.
*%HEM: Hexane extractable materials. Includes vegetable oils, animal fats, waxes, soaps and related matter.
† Each parameter was measured on 7 subsamples (n=7).
‡ Sulphur content 0.24 ± 0.03% d.b.

Except for the treatments where the initial moisture content and initial carbon-to-nitrogen ratio were manipulated variables to study, optimum initial moisture content, and C:N ratio were targeted at 55-60% and 25, respectively, in accordance with Haug (1993) and Rynk (1992).

Grease trap sludge, chicken litter, and yard trimmings were stored at 4°C. These materials were allowed to reach ambient temperature by taking them out of the cooler room the night before composting mix preparation. Grease trap sludge, when thawed, was mixed for a period of 2 hours before sampling. The amount of GTS required for recipe formulation was sampled during mixing to ensure sample homogeneity; special attention was placed on taking GTS subsamples to be as representative as possible.

After the proportions of each feedstock were calculated, the composting mix preparation proceeded as follows. For the yard trimmings treatments, a 'bed' of dry materials was formed by putting the yard trimmings, wood shavings, woodchips, and chicken litter at the bottom of the mixing container; then the canola oil or grease trap sludge was distributed uniformly on top of the dry materials 'bed'.

The composting mix was then mixed manually for about 20 minutes. Special care was taken to break any grease lumps.

Urea was dissolved in tap water prior to its addition to the composting mix, where necessary. The tap water was added to achieve the desired water content of the composting mix. The procedure for adding the activated sludge was similar to the one for urea. For the synthetic food waste treatments, the dry dog food was first reconstituted by adding the tap water needed to achieve the desired moisture content, and letting the dog food pellets absorb the water for approximately 30 minutes. The moist pellets were then 'pureed' manually. Thereafter the mixing procedure was similar to the one for the yard trimmings treatments.

3.3.2 Experimental Design

The experimental treatments using canola oil were divided into 3 experimental sets (#1, #2a, and #2b). Each set of experiment lasted for 168 hours when the composting process was going through the high-rate phase; results in terms of kinetic parameters and thermal parameters were analyzed over this 168-hour period. Actual lipid residue tests used grease trap sludge as the source of lipids, and were divided in 2 experimental sets (#3 and #4).

The following sections are organized according to the lipid substrate used. Seven manipulated variables were chosen based on the parameters suggested in the literature: substrate type, initial moisture content, initial Carbon-to-Nitrogen (C:N) ratio, initial lipid concentration, inoculum type, inoculum concentration, and aeration rate. There were seven response variables, also chosen based on literature review, as shown in Table 3.2.

Tables 3.3 and 3.4 summarize the composting recipes in terms of the composition of raw materials and basic physical properties of the composting mixture, respectively, used for all the experimental treatments. Table 3.5 gives the actual characteristics of the composting mixes used. All the canola oil treatments (except for YC1 and FC1) were performed in duplicate; and the ones with grease trap sludge were performed in triplicate. More details of the experimental treatments are given in Section 3.3.4.

3.3.3 Calculations from Experimental Data

The main response variables in this study were divided into two groups: 1) thermal parameters: temperature profile, fulfillment of the pathogen destruction requirement, heat generation, and contribution of lipids to total heat output; and 2) kinetic parameters: extent and rate of biodegradation of lipids, extent and rate of biodegradation of volatile solids, and changes in total mass and water content.

3.3.3.1 Thermal parameters

Temperature profiles were generated from the hourly composting temperature data, along with ambient temperature data. Table 3.6 displays the information to be derived from the temperature vs. time plots. The time to reach the peak temperature, t_p, refers to the total time from the start of the composting process (t=0) until the temperature peak is achieved, which includes the time lag from inoculation to the moment when temperature begins to increase above ambient.

The guideline to determine acceptable pathogen reduction (or "PFRP, process to further reduce pathogens") for in-vessel composting is 72 continuous hours (3 consecutive days) at 55°C or above (BCMWLAP 2002, USEPA 1993). For this reason, it was important to estimate the 'time at temperature equal or greater than 55°C, t_{55}' parameter.

Table 3.2 Summary of manipulated and response variables.

	Variables		Levels			
Manipulated:						
1.	Substrate type		Yard trimmings		Food waste	
2.	Initial moisture content (% wb)		40	55	60	
3.	Initial C:N ratio		20	40	60	
4.	Initial lipid concentration (% ds)		5	10	25	35
5.	Inoculum type		Chicken litter		Activated sludge	
6.	Inoculum concentration (% wb)		1		5	
7.	Aeration rate (lpm/kg ds)		0.72		1.44	
Response:						
1.	Thermal parameters:	Temperature profile				
2.		Fulfillment of the pathogen reduction requirement				
3.		Heat generation				
4.		Contribution of oil to total heat output				
5.	Kinetic parameters:	Lipids biodegradation extent (β_{lipids}) and biodegradation rate coefficient (k_{lipids})				
6.		Volatile solids biodegradation extent (β_{vs}) and biodegradation rate coefficient (k_{vs})				
7.		Reduction in total mass and moisture content				

Table 3.3 Composting mix composition, in (% ww)† except for lipid concentration. Lipid source – canola oil.

Experiment	Treatment	Lipid concentration (% ds)	Substrate	Lipid added	Bulking agent	Inoculum
Set 1	YC1	35	34	15	38	5
Substrate – yard trimmings	YC2	35	25	13	27	4
	YC3	25	23	14	21	3
	Control‡	1*	25	0	27	4
Set 2a	FC1	10	43	2	5	1
Substrate – food waste	FC2	10	10	4	35	1
	FC3	10	10	4	35	1
	FC4	10	18	4	28	1
	Control 1‡	3*	18	0	29	1
Set 2b	FC4	10	18	4	28	1
Substrate – food waste	Control 1‡	3*	18	0	29	1
	FC5	10	14	4	29	5
	Control 2‡	3*	15	0	30	5
	FC6	10	20	4	27	1
	FC7	10	19	4	27	5

† Remaining percentage is tap water and 1-2% ww urea (yard trimmings treatments only).
‡ Control: Treatment with no lipid added.
* Naturally occurring lipids from the raw materials used.

Table 3.4 Composting mix composition, in (% ww)† except for lipid concentration. Lipid source – grease trap sludge.

Experiment	Treatment	Lipid concentration (% ds)	Substrate	Lipid added	Bulking agent
Set 3	YG1	5	28	13	31
Substrate – yard trimmings	YG2	10	24	24	28
	Control‡	1*	33	0	36
Set 4	FG1	5	31	9	17
Substrate – food waste	FG2	10	26	25	20
	Control‡	3*	31	0	17

† Remaining percentage is tap water, 3% ww inoculum, and 1% ww urea (yard trimmings only).
‡ Control: Treatment with no lipid added. * Naturally occurring lipids from the raw materials used.

Table 3.5 Actual characteristics of composting mixtures (n=3, except for lipids with n=2).

Experiment	Treatment	Lipid concentration (% ds)		Moisture Content (%)		Carbon-to-nitrogen ratio, C:N		Bulk density, ρ_b (kg m^{-3})	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD
Set 1	YC1	34.3	1.2	40.1	1.1	25.3	0.2	nm	nm
	YC2	35.1	0.9	58.9	0.9	25.3	0.3	614	7
	YC3	24.9	1.2	55.0	0.7	25.3	0.3	214	9
	Control	1.2	0.2	62.1	0.5	25.3	0.2	167	49
Set 2a	FC1	10.9	1.1	55.0	0.2	20.0	0.2	509	9
	FC2	10.3	0.5	54.9	0.3	59.9	0.4	223	7
	FC3	10.3	0.5	55.1	0.2	59.9	0.4	223	7
	FC4	9.8	0.4	55.1	0.1	40.0	0.3	261	5
	Control 1	2.8	0.3	57.0	0.1	37.2	0.2	243	5
Set 2b	FC4	9.8	0.4	55.1	0.1	40.0	0.3	261	5
	Control 1	2.8	0.3	57.0	0.1	37.2	0.2	243	5
	FC5	9.6	0.6	55.0	0.9	40.1	0.2	253	2
	Control 2	2.7	0.5	57.2	1.3	37.3	0.2	250	3
	FC6	10.6	0.3	55.0	0.7	39.6	0.3	305	5
	FC7	10.6	0.3	54.8	0.1	39.7	0.2	305	6
Set 3	YG1	4.8	0.4	57.6	3.0	26.0	0.6	278	7
	YG2	9.8	1.1	59.6	4.1	26.2	0.6	248	9
	Control	1.1	0.4	57.5	3.4	25.9	0.5	112	12
Set 4	FG1	5.2	0.5	51.9	4.5	24.9	0.5	416	8
	FG2	10.3	0.3	52.2	4.2	27.9	0.3	385	5
	Control	2.7	0.4	55.0	0.1	24.7	0.3	400	15

nm: not measured.

Heat generation for the composting process was calculated based on the mass of solids and lipids consumed in the process, and the heat of combustions for volatile solids and lipids as reported by Tchobanoglous et al. (1993), with values of 15.1, 13.9, and 38.3 MJ.(kg ds)$^{-1}$, for yard waste, food waste, and fats, respectively. It was assumed that the heat of combustion for 'fats' was applicable to grease trap sludge lipids. In addition, the reported heat of combustion for canola oil (38.5 MJ.(kg ds)$^{-1}$, Munchen 1989) is very similar to the value reported by Tchobanoglous et al. (1993) for 'fats' with a value of 38.3 MJ.(kg ds)$^{-1}$. Tables 3.7 and 3.8 show the inputs and outputs of these calculations, using equations 3 to 18. Sample calculations can be found in Appendix E.

Table 3.6 Thermal parameters to be derived from the composting temperature profiles.

Parameter	Symbol	Units
Peak temperature	T_p	°C
Time to reach peak temperature	t_p	hr
Average temperature, over 168 hours of high-rate phase composting	T_{avg}	°C
Time at temperature equal or greater than 55°C	t_{55}	hr

Table 3.7 Inputs to composting kinetic and thermal calculations.

Parameter	Symbol	Units
Composting period	t	hr
Composting mix temperature	T	°C
Ambient temperature	T_{amb}	°C
Composting mass	M_{total}	kg wb
Moisture content	mc	% wb
Volatile solids content	vs	% ds
Oil content	oil	% ds
Heats of combustion	Q_c	MJ/kg ds

Table 3.8 Outputs from composting kinetic and thermal calculations.

Parameter	Symbol	Units
Biodegradability of volatile solids	β_{vs}	% ds
Biodegradability of lipids	β_{lipids}	% lipids
Biodegradation rate coefficient of volatile solids	k_{vs}	hr^{-1}
Biodegradation rate coefficient of lipids	k_{lipids}	hr^{-1}
Total heat produced	Q_p	kJ/kg.hr
Theoretical heat produced by oil	Q_{oil}	kJ

3.3.3.2 Kinetic parameters

The extent of biodegradation, also known as biodegradability 'β', refers to the amount of solids that is potentially biodegradable during the entire composting process. In other words, it is the amount of the original mass that is converted to gas or assimilated into the biomass.

The biodegradation rate coefficient refers to the reaction rate constant 'k' of a 1st order biodegradation equation (See Equation 2, where 'C' refers to concentration). According to Haug (1993) the assumption of first order kinetics is applicable to a number of biological oxidation processes, including composting. Bari et al. (2000) and Bari and Koenig (2000) found that the degradation during composting could be quantitatively predicted using a first order reaction model.

The use of first order reaction model is in agreement with the findings of Ndegwa et al. (2000), Hamoda et al. (1998), Boni and Musmeci (1998), Kaiser (1996), Keener et al. (1996), and Marugg et al. (1993). Thus, it was assumed that such first order kinetics prevailed in our experiments and hence Equations 7 and 8 were used to calculate 'k'.

$$First\ order\ biodegradation: \frac{dC}{dt} = -k * C \qquad (2)$$

The biodegradation rate coefficient 'k' corresponds to the amount of mass degraded relative to the total amount of biodegradable mass per unit time. The units of 'k' are mass biodegraded over total biodegradable mass per time, or simply 1/time (e.g. hr^{-1}). Equations 7 to 10 present the details of the kinetic parameters calculations.

Equations for thermal and kinetics parameters calculations

$$M_s = s * M_{total} = (1 - mc) * M_{total} \qquad (3)$$

$$M_w = mc * M_{total} \qquad (4)$$

$$M_o = oil * M_s \quad (5)$$

$$M_{vs} = vs * M_s \quad (6)$$

where 'M_s' is mass of solids (kg ds), 'M_w' is mass of water (kg water), 'M_o' is mass of oil (kg oil), 's' is solids content (decimal); and 'M_{vs}' is mass of volatile solids, in (kg ds).

Biodegradation rate coefficient for lipids [h^{-1}]:

$$k_{lipids} = \frac{ln\left(\frac{M_{oil,\ t=168\ hr}}{M_{oil,\ t=0\ hr}}\right)}{168\ hr} \quad (7)$$

Biodegradation rate coefficient for volatile solids [h^{-1}]:

$$k_{vs} = \frac{ln\left(\frac{M_{vs,\ t=168\ hr}}{M_{vs,\ t=0\ hr}}\right)}{168\ hr} \quad (8)$$

Biodegradability of lipids [%]:

$$\beta_{lipids} = \left(\frac{M_{oil,\ t=0} - M_{oil,\ t=168}}{M_{oil,\ t=0}}\right) * 100 \quad (9)$$

Biodegradability of volatile solids [%]:

$$\beta_{vs} = \left(\frac{M_{vs,\ t=0} - M_{vs,\ t=168}}{M_{vs,\ t=0}}\right) * 100 \quad (10)$$

Change in mass total [kg ww]:

$$M_{total,\ change} = \left(\frac{M_{total,\ t=0} - M_{total,\ t=168}}{M_{total,\ t=0}}\right) * 100 \quad (11)$$

Change in mass of water [kg water]:

$$M_{water,\,change} = \left(\frac{M_{water,\,t=0} - M_{water,\,t=168}}{M_{water,\,t=0}} \right) * 100 \quad (12)$$

Amount of lipids degraded [kg lipids]:

$$M_{oil,\,degraded} = M_{oil,\,t=0} - M_{oil,\,t=168} \quad (13)$$

Amount of volatile solids degraded [kg vs]: (14)

$$M_{vs,\,degraded} = M_{vs,\,t=0} - M_{vs,\,t=168}$$

Amount of 'non oil' volatile solids degraded [kg vs]:

$$M_{non\ oil\ vs,\,degraded} = M_{vs,\,degraded} - M_{oil,\,degraded} \quad (15)$$

Total heat produced [MJ]:

$$Q_p = M_{non\ oil\ vs,\,degraded} * Q_c + M_{oil,\,degraded} * Q_{co} \quad (16)$$

Heat from oil [MJ]

$$Q_{oil} = M_{oil,\,degraded} * Q_{co} \quad (17)$$

%Oil contribution to total heat generation

$$\%Oil_{contribution} = \left(\frac{Q_{oil}}{Q_p} \right) * 100 \quad (18)$$

3.3.4 Details of Experimental Treatments

This section describes the objective and the particulars of each experimental set.

3.3.4.1 Experimental set #1

The aim of experimental set #1 was to find out if the optimum range of moisture content for composting needed to be modified by taking into account the additional

liquid phase (oil) as suggested by Miller (1993). Fisher (1997) also recommended changing the moisture content of the composting mix when using materials like grease trap sludge, in order to have an optimum wet and dry material balance.

Canola oil at two concentration levels (25% ds as reported to be optimum by Fernandes et al. 1988, and 35% ds which represents a relatively large amount of oil) were used in the experiment. Inoculum was chicken litter at 3-5% ww. Initial C:N ratio was set at the recommended optimum value (25, Rynk 1992); it should be noted that urea was added (1-2% ww) to achieve the desired C:N ratio. Details of the formulation for treatments YC1, YC2, YC3, and Control, using yard trimmings as substrate, are shown in Table 3.9.

Table 3.9 Experimental set #1 – manipulated variables.

Treatment	Moisture content (% wb)	Oil content (% ds)	Liquid (water and oil) content (% wb)
YC1	40	35	61
YC2	60	35	74
YC3	55	25	66
Control	60	1*	60

* Naturally occurring lipids from the raw materials used.

3.3.4.2 Experimental set #2a

The objective of experimental set #2a was to test the effect of the initial carbon-to-nitrogen ratio (C:N) of the composting mix, and two aeration rates, on the composting process performance. The substrate chosen was synthetic food waste, rather than natural food waste, following VanderGheynst et al. (1997) suggestion; since the former has a consistent composition and a uniform particle size. For experimental set #2a and #2b, no urea was required to balance the C:N value.

The standard aeration rate used (for treatments FC1 and FC2) was 0.72 lpm.kg dry solids as suggested by Rynk (1992). For treatments FC3 and FC4 the standard aeration rate was doubled, since the degradation of lipids was expected to consume more oxygen (1.4 g oxygen/g food waste or yard trimmings, as compared with 2.8-2.9 g oxygen/g lipid, see Appendix F).

The composting recipe for treatments FC1 to FC4 had an initial moisture content of 55%, and 1% ww chicken manure as inoculum. Canola oil was added to all treatments in these experimental sets at a level of 10% ds, with the exception of the control treatments. Details about the manipulated variables for this experimental set are shown in Table 3.10.

Table 3.10 Experimental set #2a – manipulated variables.

Treatment	C:N	Aeration rate (lpm.kg dry matter)
FC1	20	0.72
FC2	60	0.72
FC3	60	1.44
FC4	40	1.44
Control 1	40	1.44

3.3.4.3 Experimental set #2b

The aim of experimental set #2b was to test the influence of the inoculum type and concentration on the composting process performance. Inoculum concentrations were 1 and 5 % ww, and inoculum types were chicken litter and activated sludge. Table 3.11 shows the details of the experimental treatments.

Again, canola oil was added to all treatments in this experimental set at a level of 10% ds, with exception of the control treatments that already had a lipid content of about 3% ds mainly due to the lipids present in the dry dog food. All treatments had an initial C:N around 40, and moisture content of approximately 55%; aeration rate was set at 1.44 lpm.kg dry matter.

3.3.4.4 Experimental sets #3 and #4

The objective of experimental sets #3 and #4 was to determine the effect of adding grease trap sludge (GTS) as a lipid-rich residue, in turn, to two different substrates - yard trimmings and synthetic food waste (dry dog food). There were 3 treatments involving different lipid concentrations in each set of these tests (See Table 3.12). Each treatment was run in triplicate since the actual lipid residue used (grease

trap sludge) was very heterogeneous (3 phase residue) in comparison to canola oil. All the experimental treatments had initial moisture contents of 52-60%, a C:N of 25-28, chicken litter as inoculum at 3% ww, and an aeration rate of 0.72 lpm.kg dry matter of composting mix. Lipid concentration for the control treatments corresponds to the amount of lipid naturally occurring in either yard trimmings or food waste.

Table 3.11 Experimental set #2b – inoculum type and concentration.

Treatment	Inoculum type	Inoculum concentration (%ww)
FC4	Chicken litter	1
FC5	Chicken litter	5
FC6	Activated sludge	1
FC7	Activated sludge	5
Control 1	Chicken litter	1
Control 2	Chicken litter	5

Table 3.12 Experimental sets #3 and #4. Yard or food waste, with grease trap sludge.

Treatment	Substrate	Lipid concentration (% ds)
YG1	Yard trimmings	5
YG2	Yard trimmings	10
Control	Yard trimmings	1*
FG1	Food waste	5
FG2	Food waste	10
Control	Food waste	3*

*Naturally occurring lipids present in the raw materials used.

3.4 RESULTS AND DISCUSSION

The results and discussion for all the experimental treatments are discussed by experimental sets, and by the group of response variables (thermal or kinetic).

3.4.1 Thermal Parameters

3.4.1.1 Experimental set #1

Figure 3.6 shows the temperature profiles for treatments YC1, YC2, YC3, and the Control. Thermal performance parameters are shown in Table 3.13. Comparing YC1, YC2, and YC3 at three moisture contents, 40%, 60%, and 55% respectively. YC2 with the highest oil content of 35% and moisture content of 60% achieved the highest thermophilic temperature in the shortest time period, and was followed by YC3 with moisture content of 55% and oil content of 25%. Lipids in the composting mix appeared to have a marked effect on the temperature profile.

The addition of oil at 25% ds (YC3) resulted in a time of 58 hours to reach T_p, compared with 29 hours for the treatment with 35% ds oil (YC2). In contrast, YC1 could only maintain a mesophilic temperature plateau of 40°C until hour 168, suggesting that the high oil content of 35% could provide a sustained energy level for the process, but the lower moisture content of 40% became a limiting factor for the biological process.

The control treatment, though having a moisture content of 60%, did not attain thermophilic temperature; the low oil content at less than 1% was likely the limiting factor. Thus, for composting of lipid-rich materials, both the moisture content and the oil content need to be at the appropriate levels.

Although the addition of oil at 35% ds might seem advantageous from a heat release (and drying) perspective, treatment YC2 was the only one to produce leachate as a non-desirable by-product, which potentially requires further treatment. In some preliminary trials it was observed that oil additions of more than 40% resulted in large amounts of oily leachate.

Table 3.13 Experimental set #1 – thermal performance (n=2).

Treatment	Moisture content (%)	Oil Content (% ds)	Liquid content (%)	T_p (°C)	t_p (hr)	t_{55} (hr)
YC1	40	35	61	40*	71	0
YC2	60	35	74	65	29	65
YC3	55	25	66	63	58	62
Control	60	1	60	47	55	0

*YC1 took 47 hours to reach 39°C.

None of the treatments in this set fulfilled the pathogen reduction requirement (of 72 hours at T ≥ 55°C). Nevertheless, treatments YC2 and YC3 almost fulfilled the pathogen reduction requirement, with a shortage of only 7 and 10 hours, respectively.

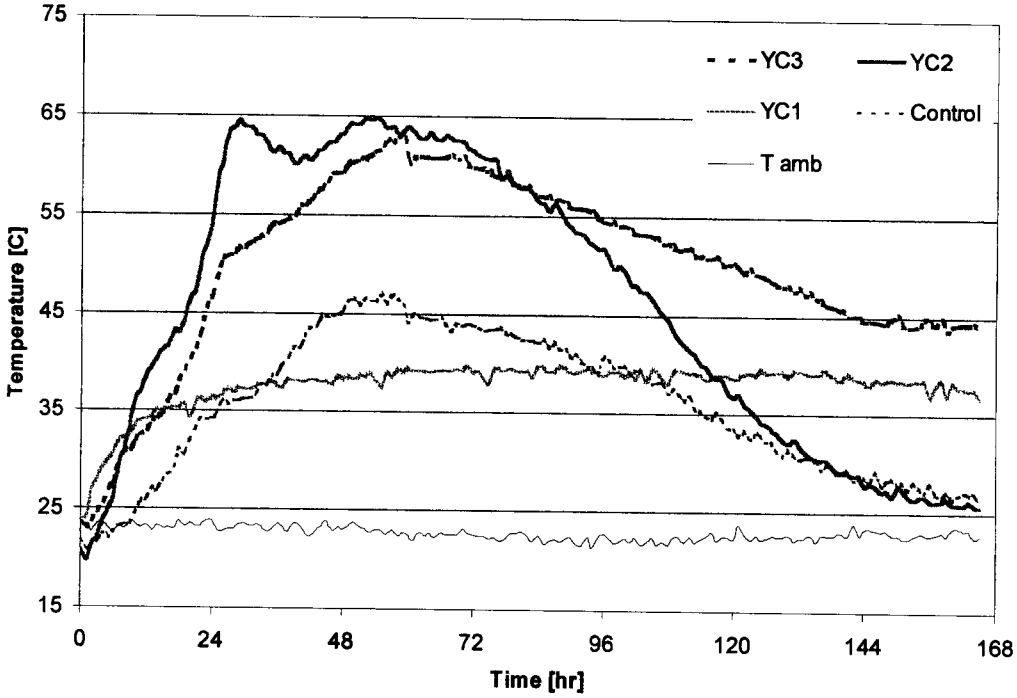

Figure 3.6 Temperature profiles for yard trimmings and canola oil treatments. Experimental set # 1. 60% MC (YC2): red; 55% MC (YC3): green; 40% MC (YC1): purple; without oil (Control): blue; room temperature: black.

3.4.1.2 Experimental set #2a

Figure 3.7 shows the temperature profiles for the 2 replicates for treatment FC2; which demonstrate that the behaviour of the temperature profiles was very similar among replicates. Temperature profiles for the treatments in set #2a are shown in Figures 3.8 (FC1 and FC2) and 3.9 (FC3 and FC4). All treatments were seen to perform in the thermophilic range, with peak temperatures ranging from 64 to 71°C (Table 3.14).

Time lag for the compost temperature to start to rise was approximately 24 hours for all four treatments (no lag times were observed in set #1). It should be noted that FC1 had a food content (43% w.b.) much higher than FC2 and FC3 (at 10%), and FC4 (at 18%). FC1 also had the lowest initial C:N value (20). Unexpectedly, this high-energy

composting mixture (FC1) took the longest time (5 days) to reach the maximum temperature of 65°C compared to other treatments with higher C:N values (40-60), although its temperature was within 90% of the peak value in less than 48 hrs. The thermal performance of FC1 also differed from other tests in that its temperature was maintained above 55°C for 130 hours and showed no sign of decline when the test was cut off at the 168th hour.

FC2 and FC3 had similar temperature profiles; hence, the doubling of aeration rate from 0.72 to 1.44 L.(min.kg initial dry matter)$^{-1}$ (for a C:N of 60) had virtually no effect on their thermal performance (a difference of only 13% in the areas under the temperature curve (3745 and 3315 °C.hr, for FC2 and FC3 respectively). The standard aeration rate would therefore be preferred, as energy consumption due to blowers can significantly affect the economics of composting (Keener et al. 1997b).

Table 3.14 Experimental set #2a – thermal performance (n=2).

Treatment	C:N	Aeration rate, (lpm.kg ds)	T_p (°C)	t_p (hr)	t_{55} (hr)
FC1	20	0.72	65	125*	130
FC2	60	0.72	66	54	66
FC3	60	1.44	64	55	60
FC4	40	1.44	71	63**	93
Control 1	40	1.44	71	39	23 + >55†

* FC1 took 43 hr to reach 60°C.
** FC4 took 45 hr to reach 69°C.
† Control was still at 55°C at the 168th hour.

Treatments FC1 and FC4 easily fulfilled the pathogen reduction requirement (t_{55} ≥ 72 hr), while FC2 and FC3 almost fulfilled the requirement with shortages of 6 and 12 hours respectively. The smaller difference in the time at temperature greater or equal than 55°C among treatments FC2 and FC3, and the larger 't_{55}' values for FC1 as compared with the values for FC3 and FC4; support the argument that the standard aeration rate should be preferred.

3.4.1.3 Experimental set #2b

As illustrated in the temperature profiles (Fig 3.10 and Fig. 3.11), all treatments in this experimental set performed in the thermophilic range. There were no noticeable differences between treatments (FC4, FC6) and treatments (FC5, FC7), implying that inoculum type (chicken litter versus activated sludge) or inoculum concentration (1% ww versus 5% ww) were not major factors affecting the thermal performance of food waste composting mixture with the addition of canola oil at 10% ds (Table 3.15). Control 2 had a temperature profile very similar to FC5, and it was consistently "ahead" of FC5 by 12-24 hours after the initial lag phase. However, it was evident that the temperature profile of Control #1 had two temperature peaks during the 168-hours composting period, and this phenomenon occurred without agitation of the compost materials in the reactor. The reason is unknown, as the only difference between FC4 and Control 1 is 7% more oil content for the former treatment. For this experimental set (#2b), the controls (without lipid added) generally did not meet the pathogen reduction requirement, in contrast to treatments with canola oil added.

Table 3.15 Experimental set #2b – thermal performance (n=2).

Treatment	Inoculum type	Inoculum concentration (% ww)	T_p (°C)	t_p (hr)	t_{55} (hr)
FC4	Chicken litter	1	71	63*	93
FC5	Chicken litter	5	70	59	81
FC6	Activated sludge	1	71	73**	100
FC7	Activated sludge	5	69	63	111
Control 1‡	Chicken litter	1	71	39	23 + >55†
Control 2‡	Chicken litter	5	67	43	55 + 19

* FC4 took 45 hr to reach 69°C. ** FC6 took 66 hr to reach 70°C.
† Control 1 was still at 55°C at the 168th hour. ‡ Control 1 had the similar composition as FC4, and Control 2 similar to FC5. None of the Control treatments had lipid residue added.

3.4.1.4 Experimental set #3

The control treatment performed in the mesophilic range with temperature peaking at 49°C. In contrast, yard trimmings treatments with lipid added (YG1 and YG2)

performed in the thermophilic range with maximum temperatures between 61-67°C (See Table 3.16 and Figure 3.12).

The presence of lipid-rich materials resulted in longer t_p (time to T_p) for the yard trimmings treatments, with values of 64 hours for YG2 and 49 hr for YG1; it also led to longer lag times (time from inoculation, t=0, to the time of first temperature rise above ambient) with values of 24 hours for YG2 and 12 hours for YG1, whereas the control did not exhibit any lag time.

After 168 hours, temperature was seen to have decreased to ambient levels for the control treatment and YG1; however it was still in the thermophilic range for treatment YG2. Such a significant difference in the thermal performance of the two bioreactors could be attributed to YG2 having a higher lipid content of 10% ds compared to YG1 at 5% ds.

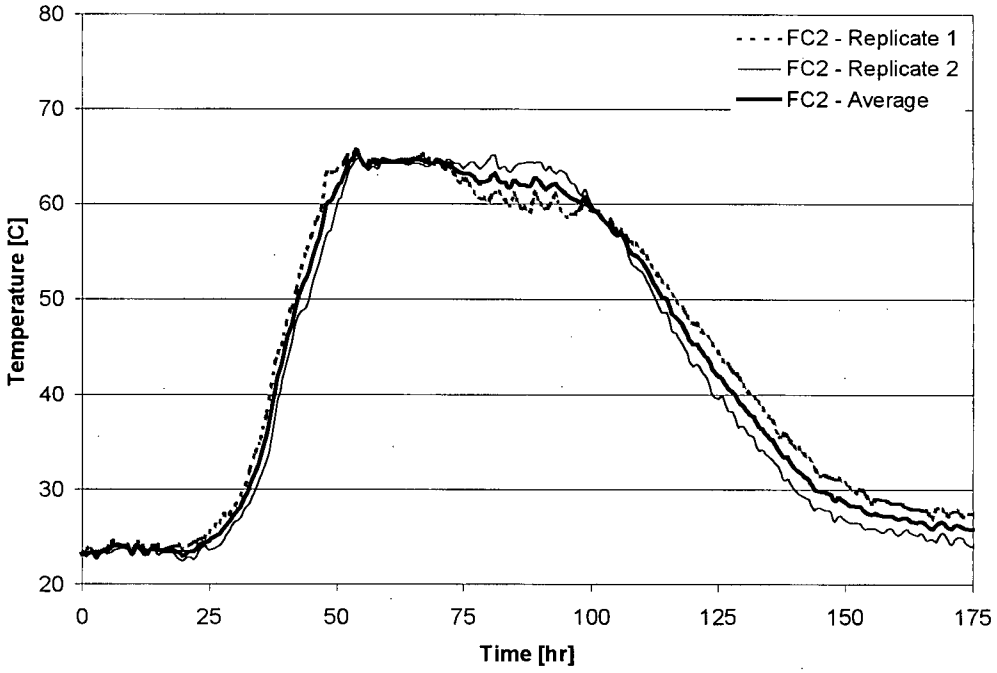

Figure 3.7 Temperature profiles (2 replicates and average) for treatment FC2 (Set #2a).

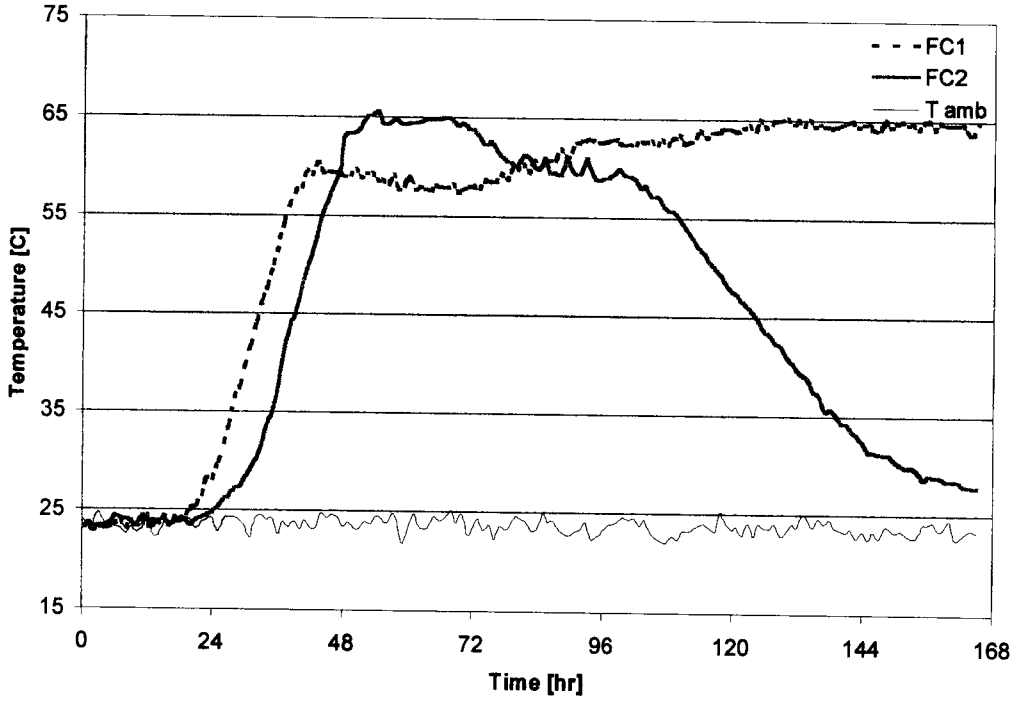

Figure 3.8 Temperature profiles for food waste and canola oil treatments. Experimental set 2a. C:N = 60 (FC2): red; C:N = 20 (FC1): green; room temperature: black. Aeration was 100% standard airflow.

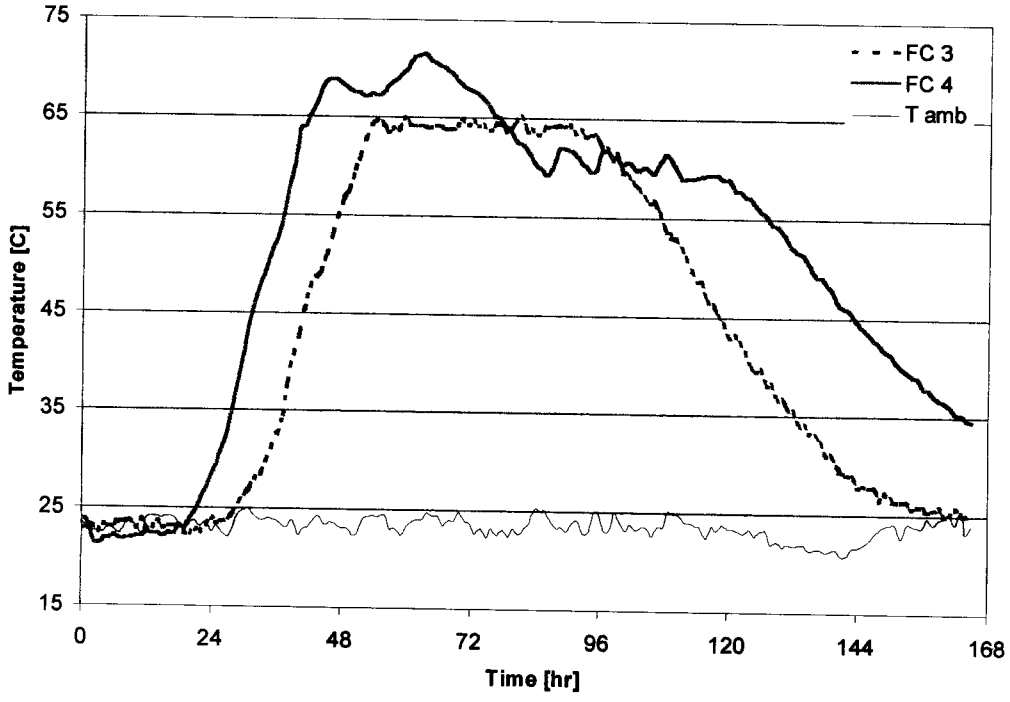

Figure 3.9 Temperature profiles for food waste and canola oil treatments. Experimental set 2a. C:N = 40 (FC4): red; C:N = 60 (FC3): green; room temperature: black. Aeration was 200% standard airflow.

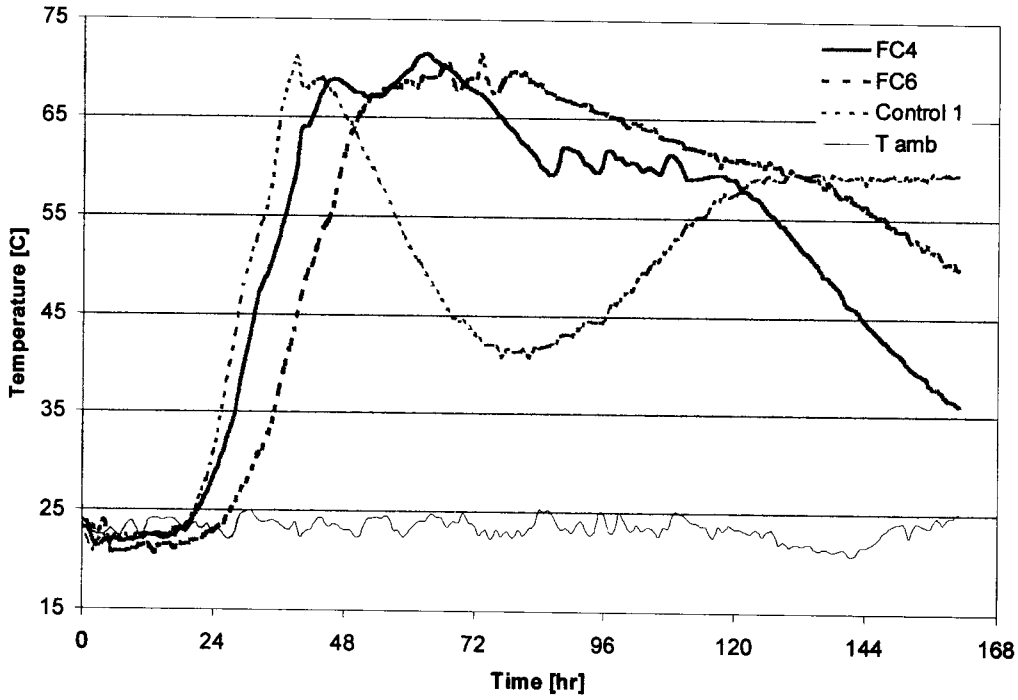

Figure 3.10 Temperature profiles for food waste and canola oil treatments. Experimental set 2b. Inoculum 1% ww. Chicken manure (FC4): red; Activated Sludge (FC6): green; Control 1: blue; room temperature: black. 10% ds canola oil added.

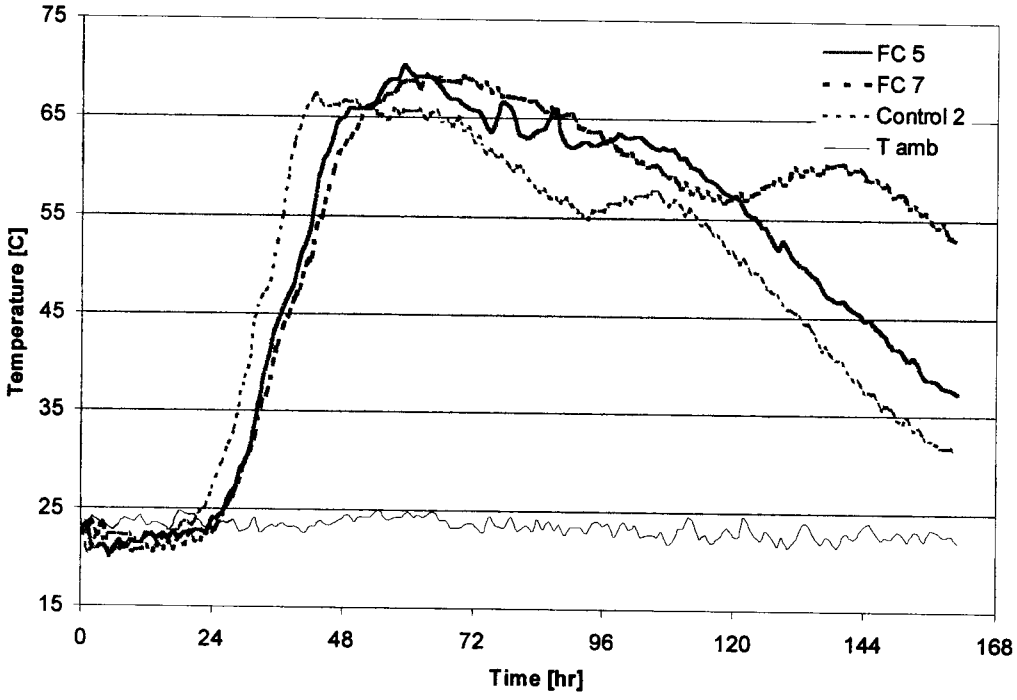

Figure 3.11 Temperature profile for food waste and canola oil treatments. Experimental set 2b. Inoculum 5% ww. Chicken manure (FC5): red; Activated sludge (FC7): green; Control 2: blue; room temperature: black. 10% ds canola oil added.

Table 3.16 Experimental set #3 – thermal performance (n=3).

Treatment	Lipid concentration (% ds)	T_p (°C)	t_p (hr)	t_{55} (hr)
YG1	5	61	49	21 + 25
YG2	10	66	64	112
Control	1	49	43	0

For pathogen reduction, only the yard trimmings treatment with 10% GTS (YG2), fulfilled the requirement. The YG1 treatment did not fulfill the requirement because the temperature profile presented a 'double bump' where composting temperature crossed the 55°C mark four times, hence not fulfilling the 72 'continuous' hours required by the Regulation.

Figure 3.13 shows the oxygen concentration in the exhaust gases for these treatments. There it can be seen that as time passed for YG2 the oxygen concentration first decreased up to about 72 hours, and then increased. This is consistent with the temperature profile for YG2, which peaks around 72 hr. For this we could conclude that the rise in the temperature and drop in oxygen concentration were indicative of microbial activity. However, while there were similar patterns for treatments YG1 and Control, the changes in oxygen were not very marked for YG1.

The increased microbial activity was also evident in the carbon dioxide profiles of the exhaust gases (Figure 4.5, pp. 104), where the YG2 treatment showed the highest carbon dioxide concentration (5.9% CO_2 at 48 hours) with decreasing values as time passed. Treatment YG1 presented the same trend as YG2, with peak carbon dioxide values between 48 and 72 hours. In contrast, the control treatment had the lowest carbon dioxide production with a peak value of 2.5% at hour 48.

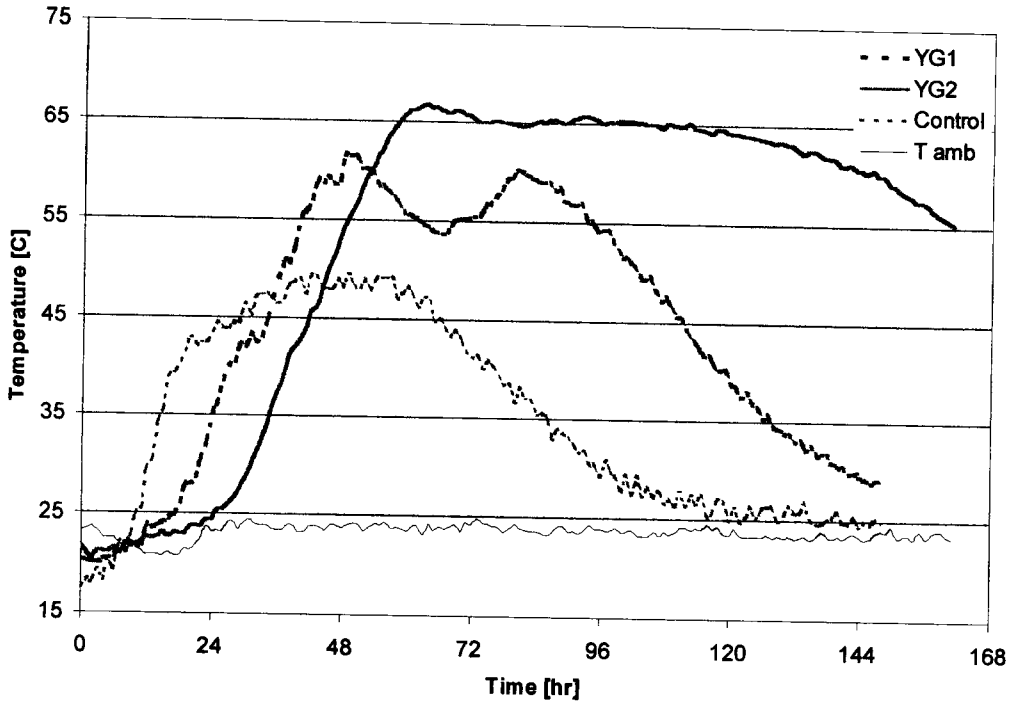

Figure 3.12 Temperature profiles for yard trimmings and grease trap sludge treatments. Experimental set 3. 10% GTS (YG2): red; 5% GTS (YG1): green; Control: blue; room temperature: black.

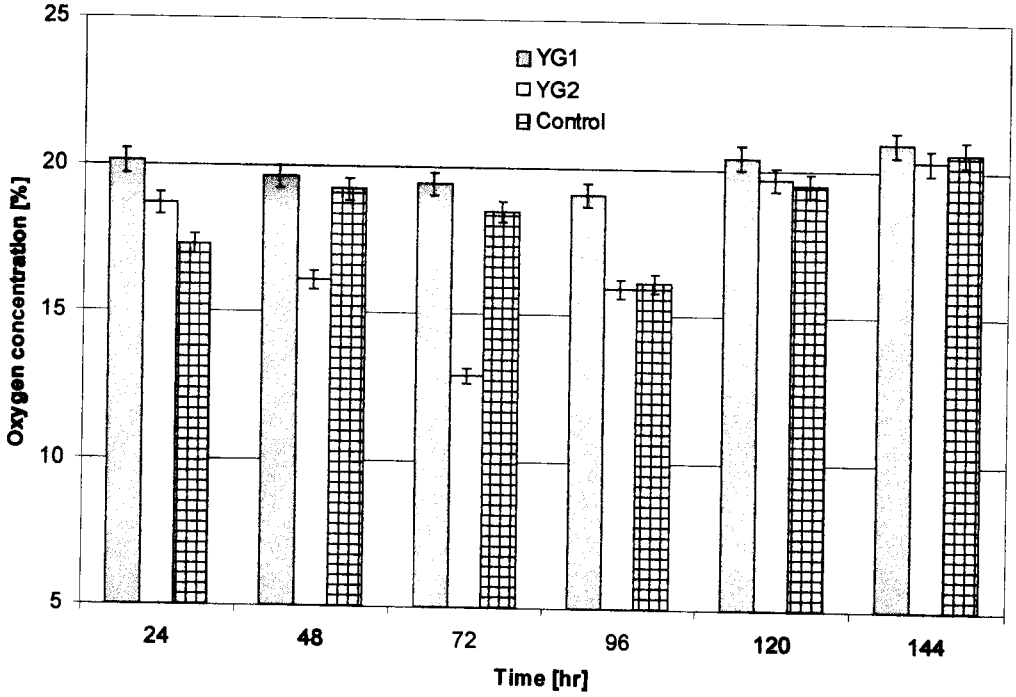

Figure 3.13 Oxygen concentration in the exhaust gases. Yard trimmings and grease trap sludge treatments. YG1: Left column; YG2: Center column; Control: Right column.

3.4.1.5 Experimental set #4

This experimental set (#4) had the same purpose as experimental set #3, except for the substrate being food waste with grease trap sludge added. Figure 3.15 shows the temperature profiles, and Table 3.17 presents the thermal parameters data for the treatment Control. Also, Figure 3.16 shows two replicate temperature profiles for the Control treatment; which had very similar temperature behaviours.

Table 3.17 Experimental set #4 – thermal performance (n=3).

Treatment	Lipid concentration (% ds)	T_p (°C)	t_p (hr)	t_{55} (hr)
FG1	5	(59)†	(168)†	(14+42)†
FG2	10	(54)†	(168)†	0†
Control	1	69	70*	148

† The numbers in brackets indicate that FG1 and FG2 were still in the active phase (increasing temperature) at the end of the 168-hour period. * Control took 38 hrs to reach 66°C.

In spite of having a high lipid content of 10% ds, no temperature peak or plateau was observed in the temperature profile of FG2 during the week-long composting period; this phenomenon is unique among most other temperature profiles in all the experimental sets. Moreover, the control treatment even outperformed the treatments with lipid waste added (FG1 and FG2).

Reduced porosity due to the presence of significant amount of lipids, considering the fact that dry dog food already had a higher lipid content than yard trimmings, could be the reason for these observations versus the opposite observations in yard trimmings tests with GTS. The bulk densities of treatments FG1, FG2, and Control, were 416, 385, and 400 kg/m^3 respectively (Table 3.5, pp.51), which were generally larger than the bulk densities of all other treatments (between 112 and 305 kg/m^3, except for FC1 and YC2).

Figure 3.15 shows that for FG2 the oxygen concentrations in the exhaust gas were always high and close to ambient (21.1 % O_2) indicating a low level of microbial oxidation. Note, however, that the oxygen concentration did continually decrease as time passed indicating that a greater level of microbial oxidation was occurring in line with the ever-increasing temperature (Fig. 3.14) noted for FG2. FG1's behavior was similar to FG2.

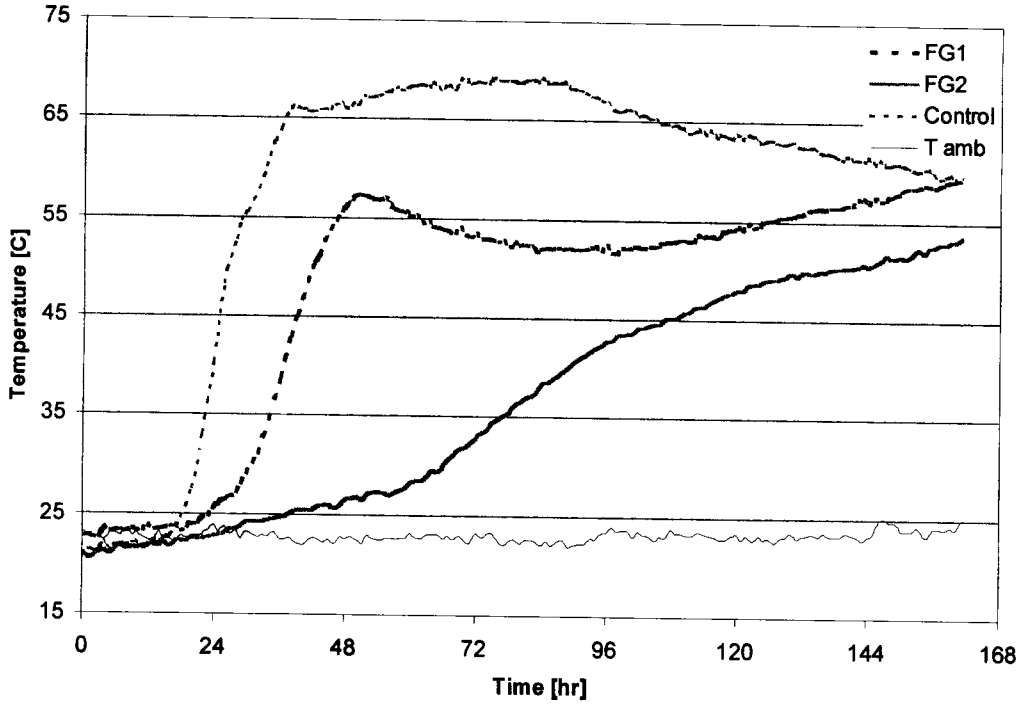

Figure 3.14 Temperature profiles for food waste and grease trap sludge treatments. Experimental set 4. 10% GTS (FG2): red; 5% GTS (FG1): green; Control: blue; room temperature: black.

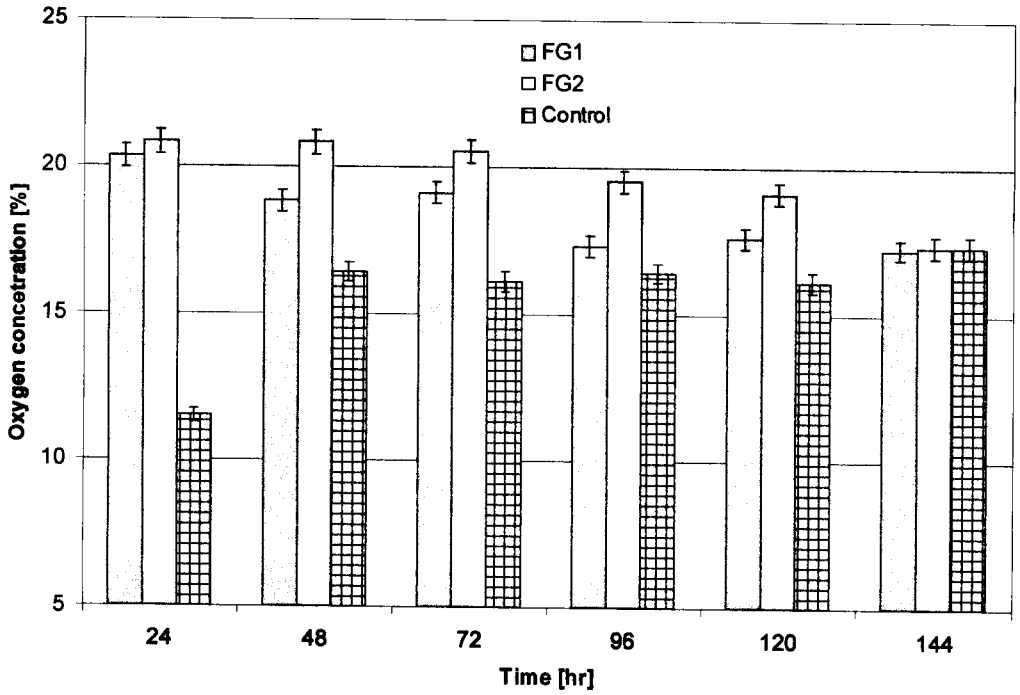

Figure 3.15 Oxygen concentration in the exhaust gases. Food waste and grease trap sludge treatments. FG1: Left column; FG2: Center column; Control: Right column.

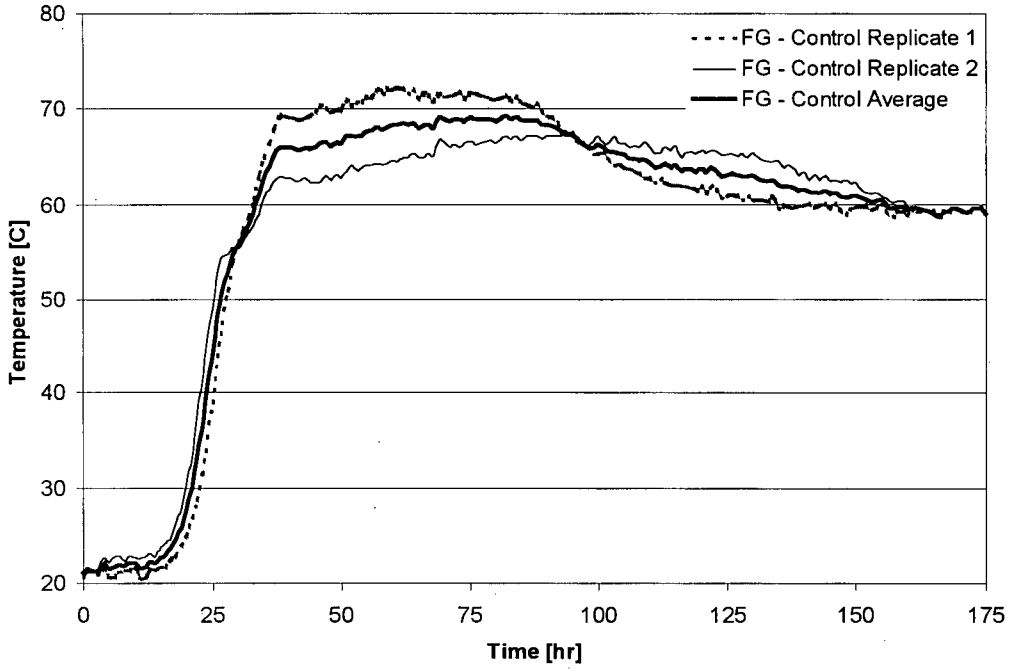

Figure 3.16 Temperature profiles (2 replicates and average) for treatment Control (Set #4).

For the Control, the oxygen concentration was always lower indicating that it tended to have the greatest level of microbial oxidation. In the case of food waste, the addition of grease trap sludge at 5 and 10% ds concentration appears to have been inhibitory to the microbial activity, as reflected in the temperature and oxygen profiles.

3.4.1.6 Total heat production and oil contribution to heat production

The total heat produced, and the percentage oil contribution to the total heat produced were calculated using Equations 13 to 18. A summary of the results for all treatments (high-rate phase of composting) is presented in Table 3.18, and Figures 3.17 and 3.18.

In general, treatments with lipids added (either canola oil or grease trap sludge) generated more heat than the control treatments, with the exception of treatments FG1 and FG2 that did not perform as expected, as explained in Section 3.4.1.5, and treatment YC1 where the microbial activity might have been restrained due to the low initial moisture content (40%) of this composting mix.

Treatments with lipid addition had mean total heat generations between 9.9 to 39.8 kJ/kg ds.hr in the case of yard trimmings, and 5.4 to 23.1 kJ/kg ds.hr for food waste. These values have a similar magnitude to the energy release value of 25 kJ/kg ds.hr reported in the literature (for flotation foams, digested sewage sludge, and sawdust as reported by Viel et al. 1987b). Treatments with canola oil added resulted in more heat generated when compared with treatments with grease trap sludge added.

The difference in total heat produced was more readily noticeable between experimental set #1 (particularly YC2 and YC3), and experimental set #3 (YG1 and YG2). The difference might be due to the larger lipid content present in set # 1, 25-35% ds, in comparison with 5-10% ds used in set # 3. In addition, the difference in total heat produced among treatments in set #1 and #3 might be due to the different type of lipids added (canola oil vs. grease trap sludge). Becker et al. (1999) using olive oil and wool scouring wastewater reported that, fatty acids up to C_{18} were easily degradable within a couple of hours. Moreover, the same authors stated that the sterol fraction of lipid wastes was very recalcitrant or difficult to biodegrade.

Similarly, the treatments with food waste and grease trap sludge added (set #4) resulted in smaller amounts of total heat output when compared with the food waste and canola oil treatments (sets # 2a and #2b).

The difference between the total heat produced for the yard trimmings treatments is linked to the contribution of oil to the total heat output. The treatments with canola oil added, had an oil contribution to heat value of 93-96% (set #1, YC3 and YC2), as compared with 38-59% for the grease trap sludge treatments (set #3).

The oil contribution to total heat output was similar for all the treatments with food wastes (except for treatment FC1), with values ranging from 41 to 72% for the treatments with canola oil, and 45 to 48% for the ones with grease trap sludge.

There was no correlation found ($R^2 = 0.06$) between the amount of volatile solids degraded and the total heat produced. However, there was a weak correlation between the percentage of lipids consumed and the total heat produced (heat produced = $0.25*\beta_{lipids}$, $R^2 = 0.57$). The correlation between the total heat produced and the overall lipid degradation was slightly stronger when only the treatments with lipids added were tested (except YC1, FG1 and FG2), resulting in a R^2 of 0.82 (heat produced = $0.32*\beta_{lipids}$) (See Appendix G).

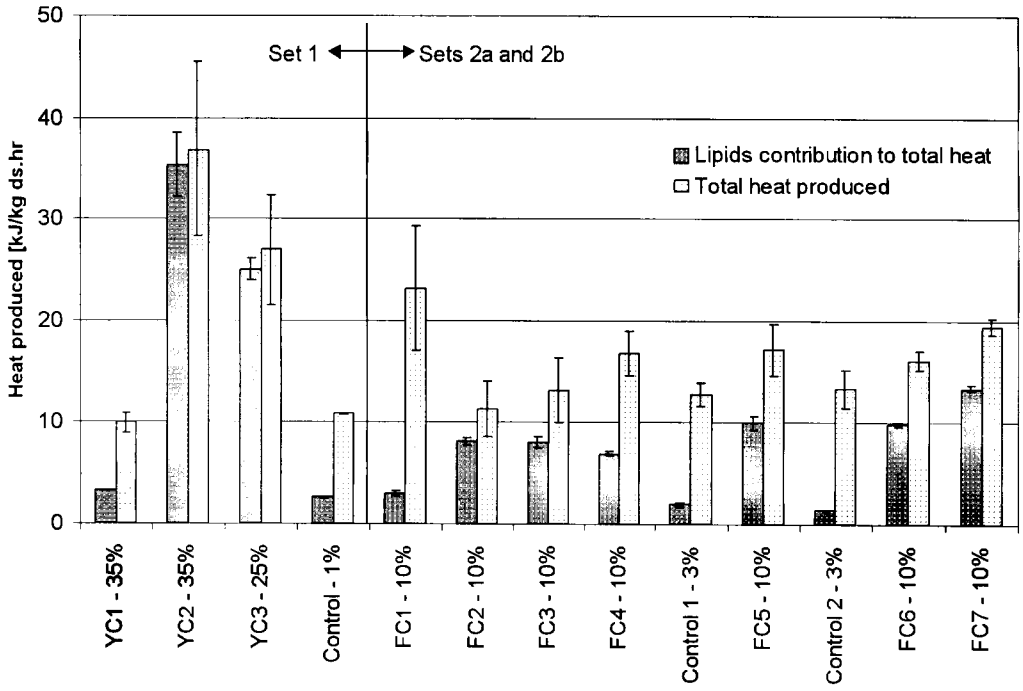

Figure 3.17 Contribution of oil to the total heat produced. Canola oil treatments.

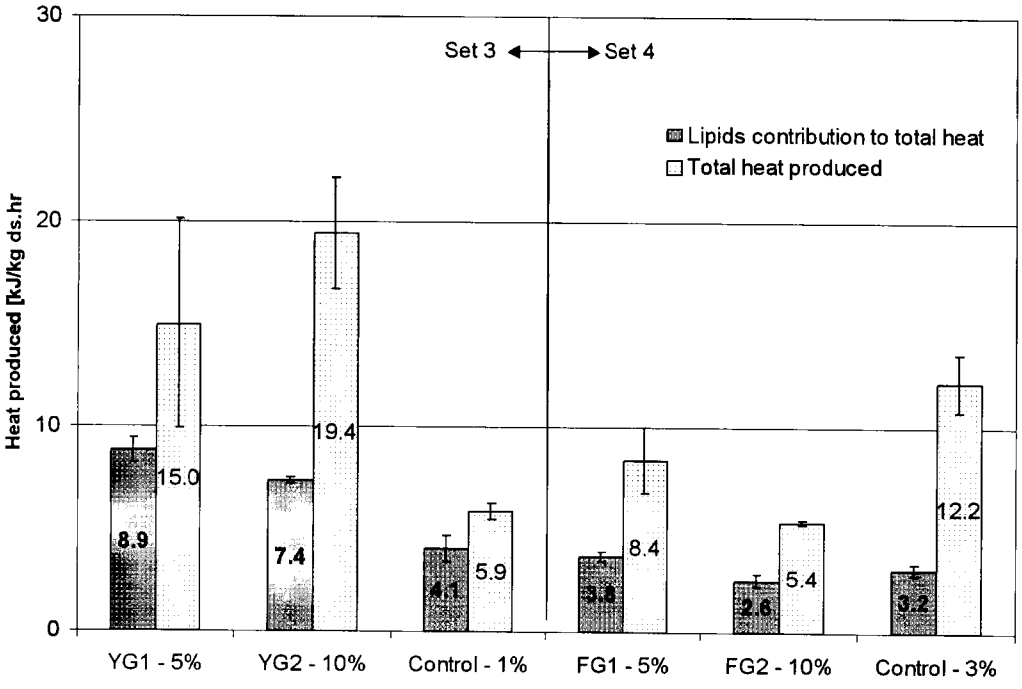

Figure 3.18 Contribution of oil to the total heat produced. Grease trap sludge treatments.

Table 3.18 Summary of heat production and oil contribution for all treatments (n=2, except Sets 3 & 4 with n=3).

Set #	Treatment and lipid content	Total heat produced (kJ/kg ds.hr)		Oil contribution to total heat produced (%)		β vs‡ (%)		β lipids‡ (%)	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD
Set 1	YC1 - 35%	9.9	1.0	33	3	6	2	61	3
	YC2 - 35%	39.8	5.4	96	9	22	4	48	24
	YC3 - 25%	26.9	8.6	93	4	15	3	62	2
	Control - 1%	10.8	0.1	24	3	26	18	75*	8
Sets 2a and 2b	FC1 - 10%	23.1	6.1	13	8	22	4	11	6
	FC2 - 10%	11.3	2.7	72	4	12	2	57	4
	FC3 - 10%	13.1	3.2	61	7	15	2	47	5
	FC4 - 10%	16.8	2.2	41	3	18	4	65	2
	Control 1 - 3%	12.8	1.1	15	8	22	4	61*	2
	FC5 - 10%	17.1	2.5	58	7	16	2	73	3
	Control 2 - 3%	13.3	1.9	10	7	23	5	39*	5
	FC6 - 10%	16.1	0.9	61	2	18	2	70	8
	FC7 - 10%	19.4	0.8	69	2	24	3	79	4
Set 3	YG1 - 5%	15.0	5.1	59	7	21	5	51	4
	YG2 - 10%	19.4	2.7	38	2	9	3	39	5
	Control - 1%	5.9	0.4	70	15	13	2	96*	19
Set 4	FG1 - 5%	8.4	1.6	45	6	7	2	27	8
	FG2 - 10%	5.4	0.1	48	12	7	2	10	8
	Control - 3%	12.2	1.4	26	9	13	5	29*	4

* Refers to the biodegradability of lipids naturally occurring in the feedstock materials.
‡ β: Biodegradability measured as percentage reduction of initial mass.

The smaller contribution of oil to total heat generated for the GTS treatments also confirms that grease trap sludge was already partially degraded. Even for the relatively short composting period (168 hours), the values of oil contribution averaging 56% are near to the 66% oil contribution found by Viel et al. (1987a,b).

3.4.2 Kinetic Parameters

3.4.2.1 Lipids and volatile solids biodegradation

Haug (1993) suggests that the values for the rate of conversion (k) and overall degradation (β) for lipids should be larger than the ones for volatile solids. Figures 3.19 and 3.20, and Table 3.19 summarize our findings for these parameters.

The biodegradation rate coefficients for the canola oil treatments were 0.009 - 0.039 day^{-1}, and for grease trap sludge treatments they were 0.009 - 0.033 day^{-1}. For the active phase of composting, the biodegradation rate coefficient for canola oil alone was calculated at 0.012 - 0.222 day^{-1} (average 0.117 day^{-1}), and for grease trap sludge alone at 0.015 - 0.101 day^{-1} (average 0.058 day^{-1}).

Values for the biodegradation rate coefficients for composting mixes with lipid residues added, and for the biodegradation of canola oil and grease trap sludge, are not known to have been reported anywhere else in the literature.

The biodegradation rate coefficients calculated for yard trimmings alone (0.021 - 0.045 day^{-1}) were smaller than the range of values (0.050 - 0.090 day^{-1}) reported by Keener et al. (1997a) for yard waste alone, and were similar to the values (0.030 - 0.061 day^{-1}) reported for mixtures of yard waste with chicken manure or biosolids. The biodegradation rates for synthetic food waste alone were calculated at 0.019 - 0.026 day^{-1}, which are similar to the Municipal Solid Waste value of 0.018 day^{-1} reported by Keener et al. (1997a). Bari et al. (2000) found a biodegradation rate of 0.024 day^{-1} for a mixture of food waste with paper and sawdust.

The overall degradation of volatile solids varied from 6 to 24% for the canola oil treatments, and from 7 to 21% for those with grease trap sludge. Although reported for different substrates, it may be useful to note that the volatile solids reduction values reported in the literature (Fernandes et al. 1988, Viel et al. 1987a,b), and found in this study for lipid-rich wastes composting are smaller than the 38% volatile solids reduction required by the BCMWLAP (2002) for Class A or B compost, and the USEPA (1993) requirements.

Lipids overall degradation (β_{lipid}) varied from 10 to 79% for the treatments with lipids added. There was no correlation found between the amount of lipids biodegraded and the initial lipid concentration (See Appendix G). This result is in agreement with the findings of Fernandes et al. (1988) who found that lipids breakdown was not related to the initial fat content.

From Table 3.19 for the canola oil and yard trimmings (set #1) experiments, it can be seen that the control treatment gave the highest values for 'k' both on a volatile solids and lipid removal basis. Similarly, the values of 'β', the biodegradability parameter, for both volatile solids and lipids were greatest for the control treatment. Thus, the addition of canola oil slowed the rate of conversion of yard trimmings and lipids, and reduced the amount of both degraded during the high-rate composting phase.

The addition of lipids presented an advantage in terms of temperature profiles, in view of the fact that the Control treatment showed the lowest peak temperature, when compared with the treatments with canola oil added. However, the lower values for the biodegradability and biodegradation rate coefficient for the treatments with yard trimmings and canola oil added is an indication that the presence of lipids might have an inhibitory effect in terms of organic mass degradation.

In experimental sets #2a and #2b, when comparing treatments with the same composition (having a similar composting recipe with the only difference in the lipid content, FC4 and Control 1, and FC5 and Control 2) the rate of conversion of volatile solids (k_{vs}) was larger for the control treatments, while the rate of conversion of lipids (k_{lipids}) was smaller for the controls. The same trend is observed for the overall (β) degradation values, with higher values for solids and smaller for lipids, for the controls (1 and 2) treatments, when compared with FC4 and FC5, respectively.

Treatments with activated sludge as inoculum (FC6 with 1% wb, and FC7 with 5% wb) had relatively high values for the overall lipid degradation, and the lipid degradation rate coefficient. Particularly, treatment FC7 showed the highest values for k_{vs} and β_{solids}, indicating that the addition of activated sludge as inoculum (at 5% wb) might help counteract the detrimental effect of the addition of lipids to composting mixes, in terms of the rate and extent of volatile solids' conversion. In addition, treatment FC7 showed the highest overall degradation of lipids in the experimental sets #2a and #2b.

For the treatments with yard trimmings and grease trap sludge added (set # 3), the treatment with 5% ds GTS added (YG1) led to a higher biodegradation rate coefficient and an extent of biodegradation for volatile solids larger than the control treatment. However, the opposite phenomenon was observed in the case of lipids (Note that the 96% lipid degradation does not represent a good basis for comparison, since

the small lipid content of the Control treatment might have been affected by measurements errors).

In contrast, all the kinetic parameter values for treatment YG2 (10% ds GTS) were smaller than the ones for the control treatment. This indicates that an increase in grease trap sludge added (above 5% ds) resulted in deteriorating composting performance. This trend is similar for the experiments with food waste and grease trap sludge (Set # 4).

The declining performance for the treatments with grease trap sludge added might be explained by the low pH value of the grease trap sludge (pH = 4) that resulted in composting mixes with low initial pH values (See Figures 5.7 and 5.8, pp. 135).

Table 3.20 shows a summary of the biodegradation extent and rate coefficient, for volatile solids and lipids, and for the curing period (grease trap sludge treatments only). Again, with yard trimmings as substrate (set #3), treatment YG1 surpassed the performance of the control in terms of biodegradation extent and rate of volatile solids and lipids. However, treatment YG2, as in the high-rate phase, presented values smaller than the control and YG1 treatments, indicating that the detrimental impact of the addition of grease trap sludge lasts for the entire composting period encompassing the high-rate phase and the curing phase.

With food waste as substrate (experimental set # 4) the values of 'k' and 'β' for volatile solids and lipids were larger for the treatments with grease trap sludge added than the ones for the control, suggesting that the composting process was still highly active after 168 hours (high-rate phase, see Fig. 3.14, pp. 69).

3.4.2.2 Change in total mass and moisture content

Table 3.21 shows a summary of the losses in total mass and water content losses for all the treatments for the high-rate phase of composting. Total mass reductions ranged from 3.1 to 45.0 % ww of initial wet mass. The majority of the values are of the order of the 30% reduction of wet mass commonly found in composting processes (Haug 1993).

Treatments with canola oil added lost total mass between 3.1 and 45.0 % wb, for treatments with canola oil added, whereas a smaller range (9.8 to 25.3% wb) was found for the treatments with grease trap sludge. Again, treatment YG1 had the largest losses in wet mass and water content (when compared with the Control of experimental set

#3). For experimental set #4, the control surpassed the performance of treatments FG1 and FG2, which is in accordance with the behavior of the grease trap sludge treatments explained in 3.4.1.5.

The water content reduction for the yard trimmings and canola oil treatments was 33.2-36.1% (except for FC1), while the food waste treatments with canola oil added lost between 42.8 to 67.4%. This higher amount of water loss for the food waste treatments might be explained by the larger aeration rate used (double the standard aeration rate). Overall, less total mass and water was lost for the treatments with grease trap sludge added, versus the canola oil treatments.

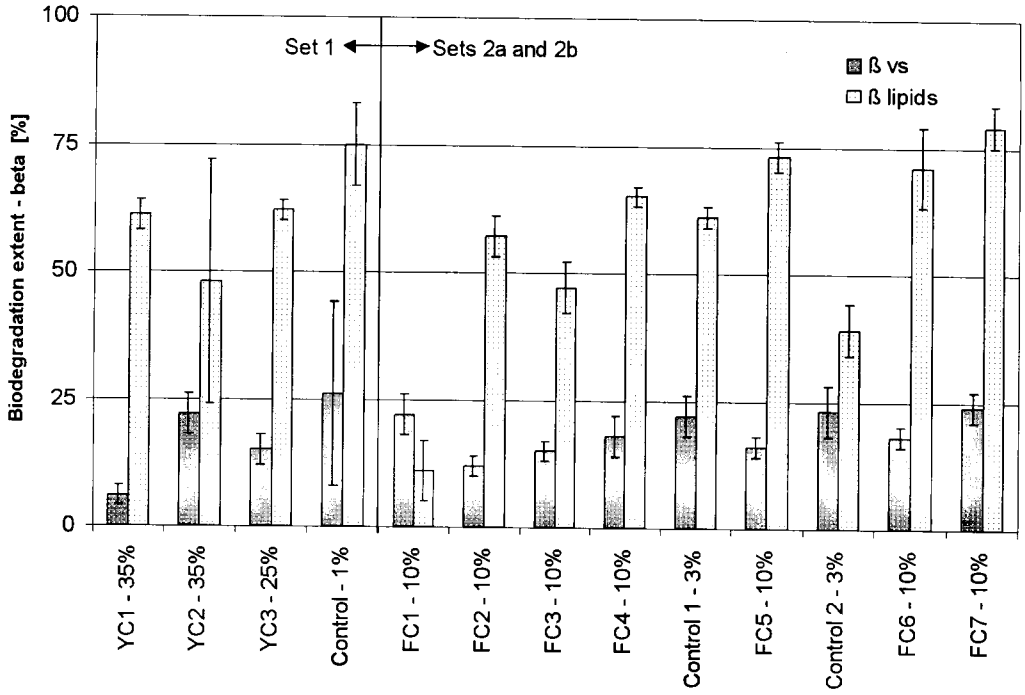

Figure 3.19 Biodegradation extent for volatile solids and lipids. Canola oil treatments.

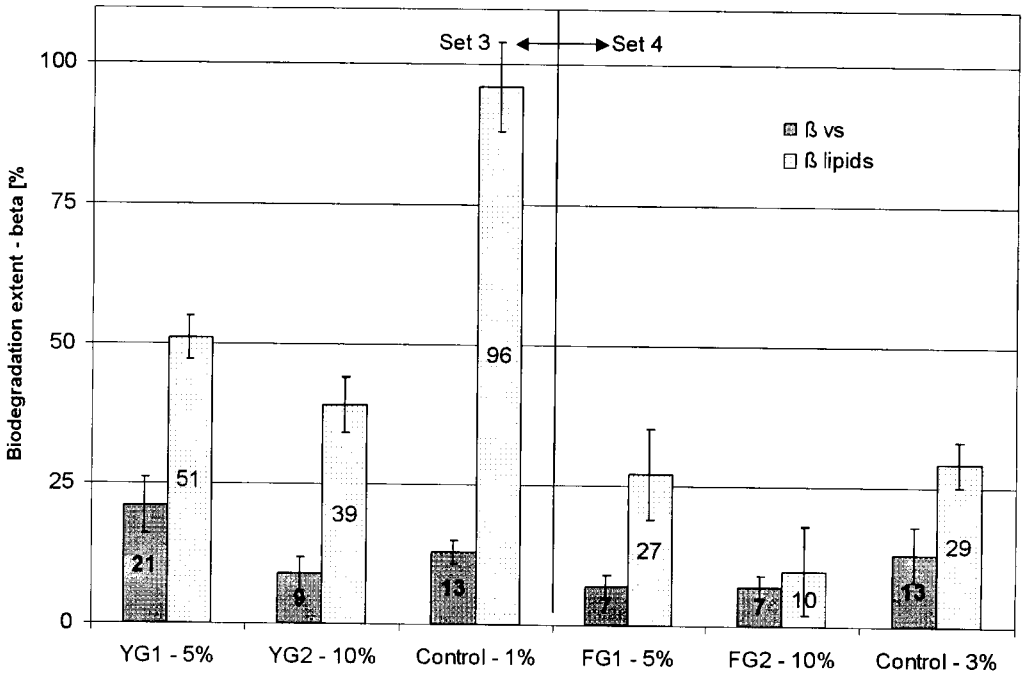

Figure 3.20 Biodegradation extent for volatile solids and lipids. Grease trap sludge treatments.

Table 3.19 Summary of biodegradability and biodegradation rate coefficient for canola oil and grease trap sludge treatments. *High rate phase* only. (n=3).

Set #	Treatment - oil % ds	k vs† (day^{-1})		k lipid† (day^{-1})		β vs‡ (%)		β lipid‡ (%)	
		Mean	SD	Mean	SD	Mean	SD	Mean	SD
Set 1	YC1 - 35%	0.009	0.003	0.135	0.007	6	2	61	3
	YC2 - 35%	0.036	0.012	0.106	0.014	22	4	48	24
	YC3 - 25%	0.024	0.007	0.137	0.024	15	3	62	2
	Control - 1%	0.045	0.036	0.205	0.045	26	18	75*	8
Sets 2a	FC1 - 10%	0.024	0.007	0.012	0.006	22	4	11	6
and 2b	FC2 - 10%	0.013	0.003	0.083	0.008	12	2	57	4
	FC3 - 10%	0.016	0.004	0.063	0.007	15	2	47	5
	FC4 - 10%	0.019	0.004	0.105	0.003	18	4	65	2
	Control 1 - 3%	0.024	0.006	0.093	0.003	22	4	61*	2
	FC5 - 10%	0.018	0.005	0.130	0.007	16	2	73	3
	Control 2 - 3%	0.026	0.008	0.049	0.005	23	5	39*	5
	FC6 - 10%	0.029	0.007	0.173	0.007	18	2	70	8
	FC7 - 10%	0.039	0.007	0.222	0.009	24	3	79	4
Set 3	YG1 - 5%	0.033	0.007	0.101	0.032	21	5	51	4
	YG2 - 10%	0.014	0.003	0.071	0.010	9	3	39	5
	Control - 1%	0.021	0.007	0.455	0.080	13	2	96*	19
Set 4	FG1 - 5%	0.010	0.003	0.045	0.023	7	2	27	8
	FG2 - 10%	0.009	0.003	0.015	0.009	7	2	10	8
	Control - 3%	0.019	0.005	0.049	0.021	13	5	29*	4

* Refers to the reduction of lipids naturally occurring in the feedstocks.
† k: Biodegradation rate coefficient.
‡ β: Biodegradability measured as percentage reduction of initial mass.

Table 3.20 Summary of biodegradability and biodegradation rate coefficient for grease trap sludge treatments. *Curing phase only*. (n=3).

Set #	Treatment	k vs† (day^{-1})	k lipid† (day^{-1})	β vs‡ (%)	β lipid‡ (%)
		Mean	Mean	Mean	Mean
Set 3	YG1 - 5%	$0.39 * 10^{-3}$	$8.37 * 10^{-3}$	9	48
	YG2 - 10%	$0.05 * 10^{-3}$	$0.90 * 10^{-3}$	1	11
	Control - 1%	$0.18 * 10^{-3}$	$7.39 * 10^{-3}$	4	4*
Set 4	FG1 - 5%	$1.39 * 10^{-3}$	$1.95 * 10^{-3}$	16	22
	FG2 - 10%	$1.24 * 10^{-3}$	$5.58 * 10^{-3}$	14	50
	Control - 3%	$0.87 * 10^{-3}$	$1.83 * 10^{-3}$	10	21*

* Refers to the reduction of lipids naturally occurring in the feedstocks.
† k: Biodegradation rate coefficient. ‡ β: Biodegradability measured (% of initial mass).

Table 3.21 Summary of total mass and water content changes for all treatments. (n=3).

Set #	Treatment - oil % ds	Total mass reduction (% ww of initial mass)		Water content reduction (% ww of initial water content)	
		Mean	SD	Mean	SD
Set 1	YC1 - 35%	3.1	0.5	1.7	0.3
	YC2 - 35%	28.3	8.0	33.2	17.1
	YC3 - 25%	26.8	9.4	36.1	15.1
	Control - 1%	7.8	1.6	10.2	3.9
Sets 2a and 2b	FC1 - 10%	45.0	4.5	67.4	2.7
	FC2 - 10%	37.9	2.9	60.0	5.2
	FC3 - 10%	31.7	4.3	46.2†	4.8
	FC4 - 10%	39.8	3.3	59.9†	4.9
	Control 1 - 3%	41.4	1.9	58.4†	3.6
	FC5 - 10%	33.1	4.3	49.9†	5.1
	Control 2 - 3%	36.8	1.5	60.0†	3.3
	FC6 - 10%	34.1	4.1	48.6†	0.7
	FC7 - 10%	32.5	3.3	42.8†	3.7
Set 3	YG1 - 5%	25.3	4.2	29.3	4.8
	YG2 - 10%	14.2	1.3	20.3	4.5
	Control - 1%	18.4	4.1	24.5	4.5
Set 4	FG1 - 5%	18.0	3.3	26.7	5.1
	FG2 - 10%	9.8	4.1	31.6	1.2
	Control - 3%	24.5	1.4	36.2	4.5

* Refers to the reduction of lipids naturally occurring in the feedstocks.
† these treatments had double the standard aeration rate (at 1.44 lpm per kg ds).

3.5 CONCLUSIONS

Following are the main conclusions derived from the information presented in this Chapter:

a. Addition of lipids seemed to have a very marked effect on the temperature profiles. Treatments with lipids added performed in the thermophilic range (except for the treatment with yard trimmings and 35% canola oil with 40% initial moisture content).

b. Addition of canola oil to yard trimmings composting at 35% ds or more resulted in the production of an oily leachate, which is undesirable due to its requirement of collection and treatment.

c. Even though the treatments with lipids added resulted in higher temperatures, they also presented a marked lag time (time from inoculation to the first rise in temperature), except from treatments with yard trimmings and canola oil, and yard trimmings alone, which showed no lag time.

d. In terms of aeration rate, doubling the standard aeration rate to 1.44 lpm/kg ds had virtually no effect on thermal performance; hence the standard aeration rate (0.72 lpm/kg ds) should be preferred due to the increased impact of higher aeration rates on composting costs (blower and operating costs).

e. The addition of grease trap sludge (up to 10% ds) seemed to be inhibitory of the composting process in terms of temperature performance and oxygen status.

f. With the exception of the treatments of food waste with grease trap sludge added, treatments with lipids added generated more heat than the control treatments (no lipids added), and fulfilled the pathogen reduction regulatory requirement more readily.

g. The total heat produced and the overall lipid biodegradation for the treatments with lipids added was found to be well correlated ($R^2 = 0.82$).

h. In terms of biodegradation rate coefficient and biodegradation extent, lipids degraded faster and more easily when compared with volatile solids biodegradation. This result is in agreement with the literature that affirms that lipids are easily degradable.

i. Part of the original contribution of this study was the measurement of biodegradation rate coefficients (k) for lipid-rich wastes, as well as the 'k' values when these residues were added to composting mixes. For the high rate phase

of composting, treatments with added canola oil resulted in biodegradation rate values for volatile solids (k_{vs}) of 0.009-0.039 day^{-1}, while grease trap sludge treatments resulted in k_{vs} values of 0.009-0.033 day^{-1}.

j. Overall degradation or biodegradability of lipids (β_{lipid}) varied from 10 to 79% for the treatments with lipids added. There was no correlation found between the amount of lipids biodegraded and the initial lipid concentration.

k. As a practical recommendation, yard trimmings treatment with grease trap sludge added at 5% ds would result in enhanced thermal performance, improved rate and extent of biodegradation of solids and lipids, greater overall reduction in wet mass and water content, when compared with the composting of yard trimmings alone. Higher biodegradation rates would save the capital investment and operating costs for a composting facility.

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CHAPTER 4
IMPACT OF LIPID-RICH WASTES COMPOSTING ON THE AIR ENVIRONMENT

4.1 ABSTRACT

There are few studies concerning the composting of lipid-rich wastes materials of animal and vegetable origin, none of which has reported the impact of composting on the ambient air environment.

Treatments examined for the impact on the environment were the ones with either yard trimmings or food waste, with grease trap sludge added to the substrate. During the experimental treatments as reported in Chapter 3, virtually no leachate was produced from the composting mixtures under optimal moisture content conditions. The emphasis here therefore is placed on the gas emissions, which included ammonia, greenhouse gases (carbon dioxide, methane, and nitrous oxide), and odor.

Treatments with 10% dry matter grease trap sludge (GTS) added were found to result in the lowest ammonia emissions, being 0.3 and 0.1 NH_3-N as percentage of total initial nitrogen for the yard trimmings and the food waste substrate respectively, over a period of 168 hours. In contrast, the treatments with 5% dry matter grease trap sludge, and the control treatments, that resulted in total ammonia losses between 0.4 and 3.1 NH_3-N as percentage of total initial nitrogen during the same period of time.

Carbon dioxide emissions increased with increasing grease trap sludge concentration in the yard trimmings composting mixes, whereas methane levels were found to be less than 2 ppmv in the exhaust gas. The average nitrous oxide measured, as percent of total initial nitrogen, for the yard trimmings treatment with 10% dry matter grease trap sludge added, was approximately 100 times greater than the values for the yard trimmings treatment with grease trap sludge at 5% dry matter, and the control treatment (with no external lipid source). There was only a small difference of 15% in the nitrous oxide generated by the latter two treatments

In terms of odor generation, the treatments with yard trimmings substrate with or without grease trap sludge added had similar cumulative specific odor emissions of 1.2-1.6 x 10^{-5} ou/kg initial dry matter. However, treatments with food waste substrate produced cumulative odor emissions one order of magnitude larger than the ones for

yard trimmings. In addition, the food waste treatments with no lipid added resulted in the higher cumulative odor emission, when compared to those with grease trap sludge added; which might be due to the atypical composting performance exhibited by the latter.

4.2 INTRODUCTION

This section pertains to the literature review of greenhouse gas, ammonia, and odor emissions generated from aerobic composting. A summary of the parameters that affect these air emissions is also presented.

4.2.1 Greenhouse Gases Emissions

Nitrous oxide (N_2O) is a natural by-product of nitrification (aerobic microbial oxidation) and denitrification (anaerobic/anoxic microbial reduction). He et al. (2000) suggested that the N_2O generation during composting is largely due to denitrification, and that this gas production should be proportional to the amount of available carbon. The production of N_2O is associated with anoxic and anaerobic microsites in the composting mix, thus biological denitrification has been found to occur even at oxygen levels in the exhaust gas as high as 15% (Hao et al. 2001, He et al. 2000, Hwang and Hanaki 2000). According to Rynk (1992) a minimum oxygen concentration of 5% (within the compost heap pore space) is necessary to assure aerobic conditions. Nitrous oxide and nitrogen gas have presumably accounted for up to 40% of the total initial nitrogen losses (Moller et al. 2000, Pel et al. 1997, Kuroda et al. 1996, Mahimairaja et al. 1994).

In a study of emissions of malodorous compounds and greenhouse gases from composting swine feces, Kuroda et al. (1996) found that losses as NH_3 gas in the air, and as NH_4-N in the leachate amounted to 40.8% and 6.7% of the total nitrogen loss, whereas the loss as N_2O was merely 0.3%. The 40% of the total nitrogen loss unaccounted for was attributed to nitrogen gas, N_2. According to Morand et al. (1999), the nitrogen losses as N_2O were 0.5% of total initial nitrogen for poultry manure and poplar bark compost. Hellebrand (1998) reported that when farm waste (bedding plus horse/poultry manure) was used the nitrogen losses, as N_2O, were 2.2% of total initial nitrogen within a 60-day period; and in the case of yard trimmings (grass clippings and fallen leaves), the N_2O losses varied from 0.5% to 1.2% of total initial nitrogen. For food

waste composting, He et al. (2000) concluded that the generation of N_2O increased with increasing food content. Nitrous oxide concentration at the beginning of the composting process was relatively high, with a peak value of 10 ppmv; but it rapidly decreased to near atmospheric, with a values of 0.45 ppmv, after 2 days.

Eghball et al. (1997) showed the range of N losses during composting of beef cattle feedlot manure to be 19 to 42%, corresponding to a C:N ratio of 17 to 12. Moller et al. (2000) observed nitrogen losses during composting of strawed deep litter from an organic pig farming system to range from 0.85-3.01 kg N/tonne, depending on the amount of straw used per unit of liveweight gain. As expected, the composting heaps with less straw mixed into the deep litter, and hence less available carbon and lower C:N, had the highest N losses. Compact composting heaps also had high nitrogen losses as oxygen was depleted to an extent that promoted denitrification.

Carbon dioxide generation is an indication of microbial metabolism. Carbon dioxide is a result of the biological degradation of carbonaceous substrates, like sugars, starch, and lipids. Peak carbon dioxide emissions occurred in the first few days of composting, and carbon losses as CO_2 have been reported as 8-22% of total initial carbon (Morand et al. 1999, Hellman et al. 1997) present in the compost materials.

The presence of methane is also an indicator of anoxic and anaerobic pockets in the composting matrix. Methane has been found during composting even when the oxygen content of the exhaust air was no less than 15%. For instance, He et al. (2000) noted maximum methane concentrations in the active phase of composting, with values of 3 ppmv for food waste, and substantially higher values of 500 ppmv for food waste with cattle manure added.

4.2.2 Ammonia Production

Compost has value as fertilizer or soil conditioner, and nitrogen is the main nutrient determining its applicability and thus its market value. Compost may be used as fertilizer if the nitrogen content is greater than 1% (Barker 1997). In the high-rate phase of composting nitrogen-rich materials, like proteins, are transformed by biological and chemical reactions. This decomposition is accompanied by a high rate of ammonification (Bishop and Godfrey 1983).

The nitrogen losses during composting can vary from a low value of 3% to a very high value of 50% of total initial nitrogen. Most of the nitrogen losses are a result of

ammonia volatilization (with ammonia concentrations between 28-1900 ppm), particularly during the initial high-rate phase of composting (Morand et al. 1999, Sommer and Dahl 1999, Mahimairaja et al. 1994, Martins and Dewes 1992, Witter and Lopez-Real 1988, Godden and Penninckx 1986). Such magnitude of ammonia loss affects the agronomic value of the compost product, and produces a harmful effect on the environment (Boucher et al. 1999).

Moller et al. (2000) reported that the ammonia concentration in the gas phase of a compost heap ranged from 20 to 200 mg NH_3-N/m^3 of compost. According to Ekinci et al. (2000) ammonia losses decreased rapidly below pH 7, and increased rapidly when the pH value was greater than 8. This is supported by the chemistry of ammonia, since the ammonium ion is more prevalent whenever pH values are below 9, and conversely molecular ammonia is more common at basic pHs (Court et al. 1964).

4.2.3 Odor Generation

Composting being an aerobic biological oxidation process of organic matter involves the undesirable emission of gaseous products, particulate matters (dust and bioaerosols), and leachate. So far, odor control is the most difficult problem to solve in present composting practices (Feinbaum 2000, Gage 2000, Haug 1993, Benedict et al. 1988). Municipal solid waste (MSW) composting plants have been shut down due to odor problems.

Odors may come from the raw materials or the products of biochemical metabolism. Usually organic wastes high in sulfur or protein, such as manure, sludge, garbage, and grass, will tend to have significant odor potentials; due to their composition (high in nitrogen content), and their tendency to compact; thus becoming anaerobic. On the other hand, carbon-rich bulking agents like sawdust or yard trimmings (branches) may have their role in odor emissions.

Conventional measurement of odors involves two techniques: (1) measurement of the concentrations of individual odor-causing chemical compounds, and (2) measurement of total odor level via olfactometry using the human sense of smell. Physical and chemical detection methods include gas chromatography and mass spectroscopy (GC/MS). Olfactometry, which is directly related to receptors, is still a widely used method of measuring odors (Feddes et al. 2001, Chen et al. 1999, Bruce 1998, Krzymien and Day 1997, Callan 1993, Berglund et al. 1987).

Sensory evaluation of smells is accomplished by using a variety of devices: scentometers, odor observation rooms, static olfactometers, butanol olfactometers, and dynamic olfactometers (Watts 1999). Since the perception of odor by humans is still subjective, despite standard protocols developed for their use over the years (e.g. CEN 1995, ASTM 1991), the use of electronic noses to detect odors has been preferred lately (Nicolas et al. 2000, Krzymien and Day 1997).

A typical olfactometer (dynamic dilution device to measure odor units, or 'ou') is a forced choice, dynamic triangle olfactometer having 3 sniffing ports per panelist station. In this device the diluted sample is flushed into one sniffing port, whereas filtered and odorless air is flushed into the other two ports. The panelist needs to differentiate which port has the odorous sample (a 'yes/no' type of question). Usually, 6 to 8 panelists participate in the test. These persons need to be screened to ensure a "normal" sense of smell.

A full evaluation of odor involves five parameters: (1) threshold odor concentration, which refers to the minimum concentration of an odor-causing compound or odorant that will arouse a sensation; (2) odor intensity, a measure of odor strength; (3) hedonic tone, which is a measure of odor acceptability; (4) pervasiveness, which concerns the difficulty of eliminating an odor; and (5) character or odor quality, which describes what the odor smells like (Feddes 2001, Haug 1993).

The number of times a given amount or volume of sample needs to be diluted with odorless air to reach an odor threshold level is called the Threshold Odor Number (TON). Different names have been used for the TON: Odor unit (ou), effective dose at 50% level (ED_{50}), dilution-to-threshold value (D/T), dilution ratio Z, and dilution ratio K (or K_{50}). Usually the terms 'ou' and 'D/T' are preferred. It is important to note that this 'ou' term refers to the number of volumes (i.e. m^3) that a sample will occupy when diluted to the odor threshold. Thus, odor units (ou) are volume of sample diluted to threshold volume of original sample. In other words, 'ou' is a dimensionless value. However, most of the times the odor concentration is equivocally expressed as ou/m^3 (Haug 1993).

Typical composting odor concentration values, in odor units [ou], (values of odor concentrations of exhaust gas samples from typical composting operations) found in the literature are summarized below: received materials (800 – 7500); in-vessel pile (380 – 3400); curing pile (540 – 3200); windrow composting (5000 – 25000); rotating drum

MSW composting (25000 – 50000); and biofilter outlet (45 – 510) (Lau et al. 1996, Giggey et al. 1995, Haug 1993).

Lipid-rich wastes have a high degree of associated odor, mainly due to the oxidation of lipids that result in fatty acid production. However, by adding the lipid-waste to composting mixes the odor was largely reduced according to Alpert et al. (2001), who reported this result giving no other explanation about it.

4.2.4 Factors Affecting Composting Process Air Emissions

The major factors affecting air emissions generated during composting are: carbon-to-nitrogen ratio, chemical species of carbon and nitrogen, temperature, pH, aeration rate, partial pressure of oxygen, and moisture content.

4.2.4.1 C:N, carbon availability, and nitrogen species

In aerobic degradation, microorganisms use 15-30 parts of carbon for each part of nitrogen, hence the theoretical optimum composting C:N value of 15-30. However, Okereke and Meints (1985) found that the presence of a readily available source of carbon produced an immediate immobilization of nitrogen. Kayhanian and Tchobanoglous (1992) also suggested that the C:N ratio should be based on the readily available carbon. Furthermore, an example of the effect of carbon availability on air emissions was shown by the reduced ammonia emission when wheat straw and peat were added, at 34% and 26% respectively, to poultry manure (Mahimairaja et al. 1994).

The rate of denitrification (the conversion of nitrate/nitrite to nitrogen gas) is largely influenced by the amount of available carbon, as an increased biological decomposition rate would result in an environment where oxygen is readily depleted; thus favouring denitrification (Rolston 1981). He et al. (2000) put forward a similar argument, by stating that the N_2O gas production during composting is largely due to denitrification, and it should be proportional to the available carbon.

Bremner and Blackmer (1981) suggested that the N_2O amount would be greater with the presence of nitrifiable forms of nitrogen such as ammonium and urea; as compared to when only nitrates are present. In contrast, Schenk et al. (1997) found that the total N_2O losses during the thermophilic phase of composting were about 4 times higher for an aerated continuous flow (tunnel) reactor than for a batch compost heap. This difference was attributed to the larger proportion of biowaste in the tunnel reactor

that resulted in a higher nitrate content; when compared to the compost heap that had more carbon from woody materials in order to maintain a high porosity. In terms of the relative magnitudes of CO_2 versus N_2O emissions from the tunnel reactor, they observed a decrease of 80% in the CO_2 emission rate from day 3 to day 12 after the start of the thermophilic phase, with a similar reduction associated with the N_2O emission.

4.2.4.2 Temperature and pH

Temperature affects a number of parameters in the diffusive and convective transfer of gases during composting. Ammonia volatilization increases with temperature, since higher temperatures increase the relative proportion of NH_3 versus NH_4^+ (especially under high pH conditions), decrease the solubility of NH_3 in water, and increase NH_3 diffusion in the composting mass (Liang 2000). Such positive correlation of ammonia emission with temperature and pH has also been reported by Jakobsen (1994). Martin and Dewes (1992) found that when composting poultry manure, NH_3 and NO_x emissions ran parallel to the increase in temperature. In contrast, Hellman et al. (1997) reported that N_2O was primarily produced and emitted in the first days of composting when temperatures were still low, and during the curing phase; hence N_2O was detected only in minor amounts during the thermophilic stage.

Hao et al. (2001) based on findings by other researchers stated that during the early stage of composting, the conversion of nitrite to nitrate could be inhibited by high values of one or more of the following parameters: temperature, pH, ammonium concentration, and volatile fatty acids (VFA) concentration. They postulated that the accumulation of nitrite in combination with abundant concentrations of amine and phenolic compounds would result in the production of N_2O through chemo-denitrification. Regarding pH, Rolston (1981) observed very little denitrification at low pH, and increasing denitrification with increasing pH values.

The increase in vapor pressure of chemical compounds with temperature is well known. However, the B.C. Regulations (BCMWLAP 2002), and the USEPA (1993), require active compost temperatures greater than 55°C for pathogen elimination, hence, it is not possible to reduce air emissions via lowering the composting temperature. Nevertheless, odor emission rate, which is a product of odor concentration and aeration

rate, could be curtailed if a high aeration rate was not used for temperature control (excessive heat removal) of the composting process.

4.2.4.3 Aeration rate and oxygen partial pressure

Air emissions from composting are directly related to the aeration rate, with increased volumes of air emissions for larger aeration rates. Hao et al. (2001) reported that CO_2 losses doubled, CH_4 emissions increased by 25%, and N_2O emissions rose by 75%, when active aeration was used (as compared with passive aeration emission values).

Hwang and Hanaki (2000) applied a tracer technique to an oxygen controllable reactor with artificial refuse (mixture of compost, dog food, and soluble starch). They proved that biological denitrification was a main source of released N_2O even when the oxygen of the bulk atmosphere was as high as 15%. Furthermore, they found that nitrification began to occur simultaneously with denitrification when the oxygen concentration was above 5%.

He et al. (2000) measured N_2O and CH_4 from the aerobic composting, in 18-L reactors, of food waste. Their results showed that adding composted cattle manure increased N_2O emissions not only at the beginning (t=2 days) of composting, and in proportion to the amount of food waste; but also during the later period (from day 12 to day 40). Thus resulting in two peak emission events, as compared to single peak curve for treatments without manure. In addition, high concentrations of methane (up to 300-400 ppmv) were observed in treatments with manure added. The authors' observations demonstrate the presence of anoxic and anaerobic microenvironments in the composting mix.

4.2.4.4 Moisture content

Moisture content in the composting mix has an effect on the free air space and on the amount of ions that can be kept in solution. In the case of ammonia, a higher moisture content (other parameters being equal) would keep more ammonia in solution; thus decreasing its volatilization, and increasing immobilization (Liang 2000). In terms of N_2O emission, denitrification was again the main source of released N_2O when the moisture content was at 40-60%, and the oxygen concentration was held at 15% (Hwang and Hanaki 2001).

4.3 MATERIALS AND METHODS

4.3.1 Experimental configuration and recipe formulation

The compost reactors, composting process monitoring and control, feedstock characterization and preparation, as well as composting recipe formulation are presented in detail in Chapter 3 (Sections 3.3.1-3.3.3). Treatments examined for the impact on the environment were the ones with either yard trimmings or food waste, with grease trap sludge added to the substrate.

4.3.2 Analytical Measurements

Details of the analytical procedures used to test compost solids samples are described in Chapter 3 (Section 3.3.1.3). The sampling and analytical procedures used for the exhaust gases from the composting process are described in this section.

Samples from the exhaust gases were collected manually using the bag-and-vacuum technique (See Figure 4.1). During sampling, a Tedlar bag, with a volume of 5-L or 10-L and fitted with one plastic valve (Safety Instruments Inc., Edmonton, AB), was filled with exhaust gas by activating the vacuum-inducing pump. The sampling period lasted 2 minutes whereby the aeration pump was 'on' for 1 min, and then "off" for 1 min. In each sampling event, approximately 2-3 L of exhaust gas sample was collected for further analysis. The bags with the exhaust gas samples were kept in a cooler room at 4°C until delivery to a GC-equipped lab.

Samples bag were reused due to their high cost. In order to reuse them, a 'bag-cleaning system' was set up by using 2 peristaltic pumps (one for filling the bags, and one for emptying them, Cole Parmer, Model No. 7553-70), a carbon-activated filter, and an automatic chronometer (Chrontrol Model CD-4, Lindburg Enterprises Inc., San Diego, CA). Tubing used was Tygon R-1000, NSF 51. The bags were 'flushed' with activated carbon filtered air about 400-500 times (in a 10-minutes fill-empty cycle). The bags where then tested by direct sniffing; bags deemed clean where reused, while the ones with residual smell where flushed again or discarded.

Ammonia was measured using gas detection tubes (precision ± 10%, Part No. 10-100-15-3M, RAE Systems Inc., Sunnyvale, CA). Oxygen concentration in the exhaust gases was measured manually using an oxygen sensor and a controller (precision ± 0.1%, Model 1630, ESD Inc., Newark, DE. The oxygen sensor was regularly calibrated using lab grade air from a cylinder). Oxygen was monitored daily

during one aeration cycle (air pump 'on' for 1 minute, and 'off' for 2 minutes). The minimum oxygen concentration for the 3-minute aeration cycle was recorded manually once a day.

Greenhouse gases, as methane and carbon dioxide were measured in exhaust gas samples using a gas chromatograph (HP 5890 Series II) equipped with a flame ionization detector (with measurement precision of 3.8% for a 982 ppm CO_2 standard, and 2.0% for a 9.8 ppm CH_4 standard). Nitrous oxide was measured using a gas chromatograph (HP 5840A) with an electron capture detector (Measurement precision was 2.5% for a 1.1 ppm N_2O standard). Both gas chromatographs used the same type of column (2m x 3.2mm OD stainless steel column packed with Porapak Q 80-100 mesh).

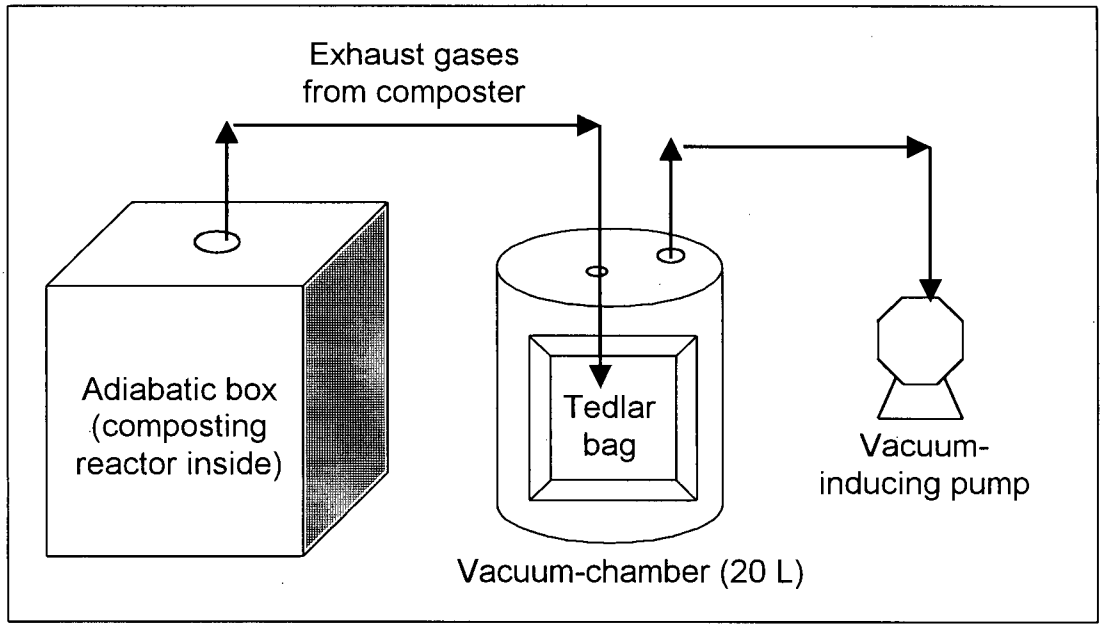

Figure 4.1 Bag-and-vacuum sampling set up.

Total odor concentration was measured using olfactometry within 24 hours of sampling. The olfactometer used is a prototype previously designed, assembled, and calibrated by Bruce (1998) in accordance with the ASTM E-679 Standard (1991), while taking into consideration some provisions of CEN (1995). This olfactometer is a three-panelist, triple port, forced-choice, dynamic dilution olfactometer (See Figure 4.2).

A forced-choice triangular test protocol was followed, using three sniffing ports associated with each station. The olfactometer set up allowed for the presentation of three samples (each at one sniffing port) at a time to the panelist. A set of 3 samples (2 with odor-free air and one with the test sample) was presented (in random order) to the panelist. The panelist evaluated each set of samples, and indicated which one of the three was the test sample. Each test sample was presented to the panelist in ascending order of sample concentrations, with a series of dilution ratios increasing two-fold, so as to avoid olfactory fatigue (odor habituation and loss of sensitivity) according to ASTM (1991). The airflow rate through each port was controlled at 5 lpm. To generate odour-free air, compressed air from the building air supply system was filtered with activated carbon.

Three 'screened' panelists (one female and two males, with ages between 22 to 38) assisted in each olfactometry analysis. The panelists were screened using the n-butanol method according to ASTM (1991), and all of them had 1+ year of experience as olfactometry panelists.

An automated system was used to record the panelists' input. The automated system used has six electrical switches arranged in two sets (one set for port selection, and the other for certainty of choice). With the second set of switches, the panelist can indicate the presence of odours as negative, inkling, or positive. The electrical switches are connected to a computer, which process the results with aid of a customized C^{++} software program (Bruce 1998).

The statistical analysis of the panelist responses produces a number called 'Best Estimated Threshold Concentration (BET)' of the panelist. The 'BET' numerically corresponds to the geometric mean of the last missed and the first correctly identified concentration. The 'ou' number (or D/T, Dilution-to-threshold) was then calculated as the statistical combination of all panelists' responses for a given sample.

In summary, olfactometry analysis produces a quantity called 'D/T' (dilution-to-threshold). The 'D/T' of a given sample is the number of volumes of clean (odor-free) air that would be necessary to dilute one volume of sample air to the level at which the average person (50% of the panelists) could not detect the odor. In other words, 'D/T' represents the detection threshold, and corresponds to the concentration at which 50% of the smellers can notice that an odor is present.

Odor emission rates were calculated using the odor concentration values measured by olfactometry, in combination with the airflow rate passing through the composting mass. Specific odor emission rates were then calculated as the odor emission rates per unit composting mass.

4.3.3 Experimental Design

Three manipulated variables were chosen on the basis of previous experimental runs: (1) substrate type, (2) presence or absence of lipids, and (3) initial lipids concentration. Substrates used were yard trimmings (YW, mainly grass clippings), and synthetic food waste (FW, dry dog food). Grease trap sludge (GTS) was added to the substrate used as lipid-rich waste. Chicken litter was used as inoculum at 3% wet basis. Details on the composition of the composting mixes were presented in Table 3.4 (pp. 50). A total of six experimental treatments were performed with details given in Table 4.1.

The main response variables were: (1) temperature profile, (2) greenhouse gases generation (carbon dioxide, methane, and nitrous oxide), (3) ammonia losses in the exhaust gases, and (4) odor concentration.

Table 4.1. Experimental treatments used for air emissions study.

Treatment	Main substrate	GTS added	Lipid concentration (% ds)
YG2	YW	Yes	10
YG1	YW	Yes	5
Control	YW	No	1*
FG2	FW	Yes	10
FG1	FW	Yes	5
Control	FW	No	3*

*Lipids naturally occurring in the raw materials. YW: yard trimmings, FW: Food waste, GTS: grease trap sludge.

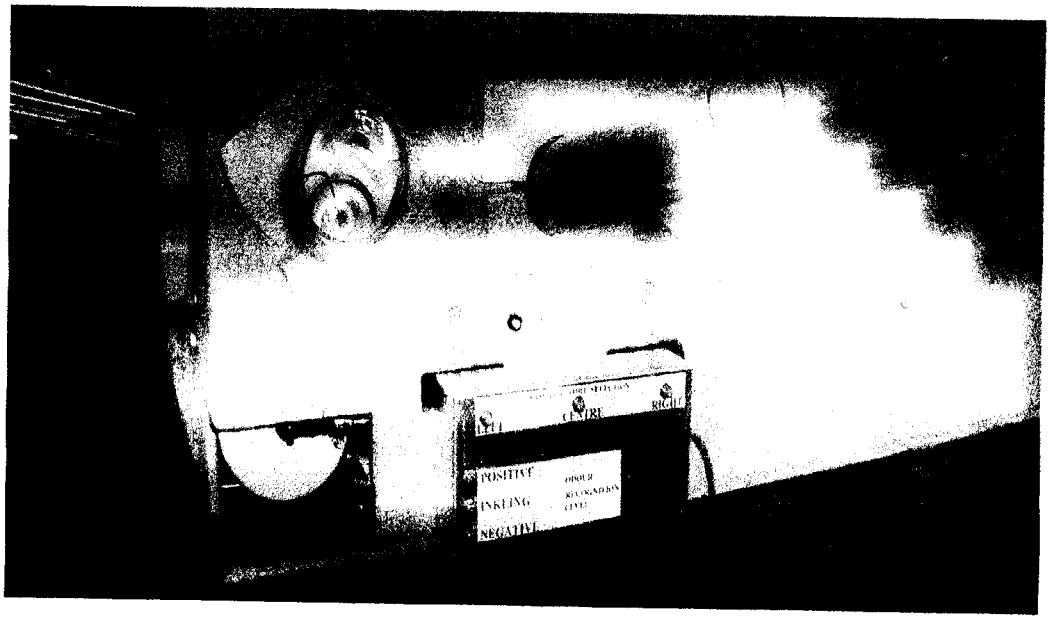

Figure 4.2 Olfactometer used for odor testing. Top: Panelist selecting a response. Bottom: One station (3 ports and input box) details.

4.4 RESULTS AND DISCUSSION

4.4.1 Temperature Profiles

The temperature profiles, mass changes (volatile solids, total mass, water content, and lipids), as well as the kinetic and thermal parameters were shown in detail in Chapter 3 (Sections 3.4.1 to 3.4.2). Figures 4.3 and 4.4 display the temperature profiles for treatments YG1, YG2, and Control; and FG1, FG2, and Control, respectively.

With yard trimmings (YW) as the main substrate, temperature curves typical of composting were seen to be similar for all the experimental treatments. The control treatment could only achieve the mesophilic temperature regime with a peak temperature around 49°C. In contrast, the treatments with lipid added were in the thermophilic regime with peak temperature values between 61-67°C.

In the case of synthetic food waste substrate (FW), the control treatment produced a higher peak temperature than the ones with lipid waste added (Treatments FG1 and FG2). Reduced porosity (higher bulk density) due to the presence of significant amount of lipids, and the fact that dry dog food already has higher lipid content than yard trimmings, could be the reason for these observations versus the opposite observations in yard trimmings tests with GTS (See Section 3.4.1.4, pp 62). In addition, as discussed in Chapter 3 (Section 3.4.1.5, pp. 68) the addition of GTS at 10% ds to synthetic food waste seemed to inhibit the composting process.

4.4.2 Greenhouse Gases Emissions

The concentrations of carbon dioxide, methane, and nitrous oxide were measured only on selected days for the yard trimmings treatments, due to analytical constraints. Results for each of the three gases are shown in Figs. 4.5 to 4.7, whereas the total emissions are summarized in Tables 4.2 - 4.4

The highest concentration of gases emitted during composting usually is paired to the period of peak temperatures, in these tests this coincides with the first 48-72 hours of composting. Carbon dioxide concentration (Figure 4.5) was higher for Treatments YG2 and YG1 (at 5.9% and 3.3% v/v respectively at hour 48) when compared to the control treatment at 2.5% v/v, indicating that higher lipid content could enhance the metabolic rate. The enhanced metabolic rate for treatments YG2 and YG1 is also indicated by the temperature profiles and the oxygen profiles (Figs. 4.3, and Fig.

3.13, pp. 67). In all the experiments, the oxygen content in the exhaust gases was almost always above 15%.

Methane concentrations were similar for all treatments tested, ranging from 1.4 to 2 ppmv at hour 48 of composting. As demonstrated in Figure 4.6, the very low concentrations of CH_4 were near background atmospheric level (approximately 2.0 ppmv), and were in line with the observations made by He et al. (2000) for composting of food waste without manure, and Kuroda et al. (1996) who recorded practically no CH_4 emission during the composting of swine feces. In comparison, the gas concentrations measured by Hellebrand (1998) during composting of "grass cuttings with manure" in 14 m^3 wooden boxes (depth 1.75 m) indicated values of CH_4 varying from 50-150 ppm in the early stage, likely caused by insufficient aeration and the onset of anoxic conditions in the heap; the low NH_3 concentrations also found (10-70 ppm) were supportive of this argument.

Nitrous oxide generation (Table 4.4 and Fig. 4.7) was markedly different between the treatments. The level of N_2O (160 ppmv) at 48 hours in the of yard trimmings with 10% grease added (YG2) treatment was significantly higher than the values for the treatments with 5% grease added (YG1) and the control.

Consequently, the average N_2O measured as percent of total initial nitrogen for YG2 was about 100 times greater than the other two treatments. There was only a small difference (15%) in the N_2O generated by the treatment with 5% lipid added and the control treatment. The larger amount of added grease trap sludge for YG2 might have resulted in more anoxic and anaerobic microsites inside the composting matrix. The oxygen concentration profile for treatment YG2 (Fig 3.13, pp. 67) consistently showed lower oxygen concentrations when compared with treatments YG1 and Control. According to Mahimairaja et al. (1994) more N_2O will be released under such circumstances. Similar readings (130 ppm) have been obtained by Hellebrand (1998) on day 2 of composting, with a drop in the N_2O concentration to 10 ppmv as of day 4.

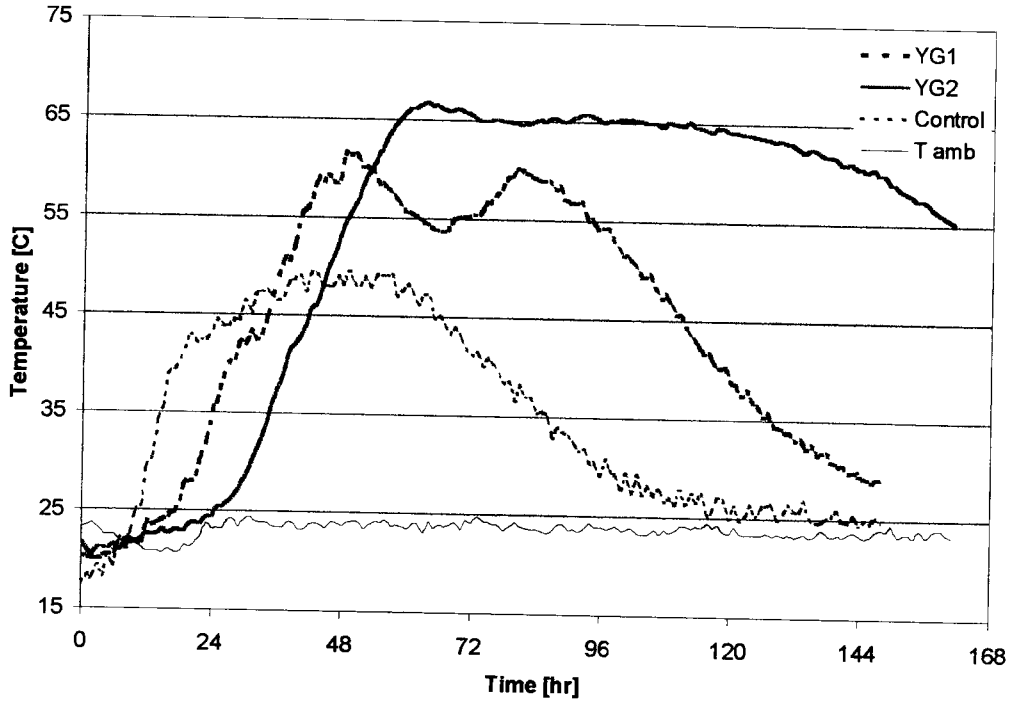

Figure 4.3 Temperature profiles for yard trimmings and grease trap sludge treatments. Experimental set 3. 10% GTS (YG2): red; 5% GTS (YG1): green; Control: blue; room temperature: black.

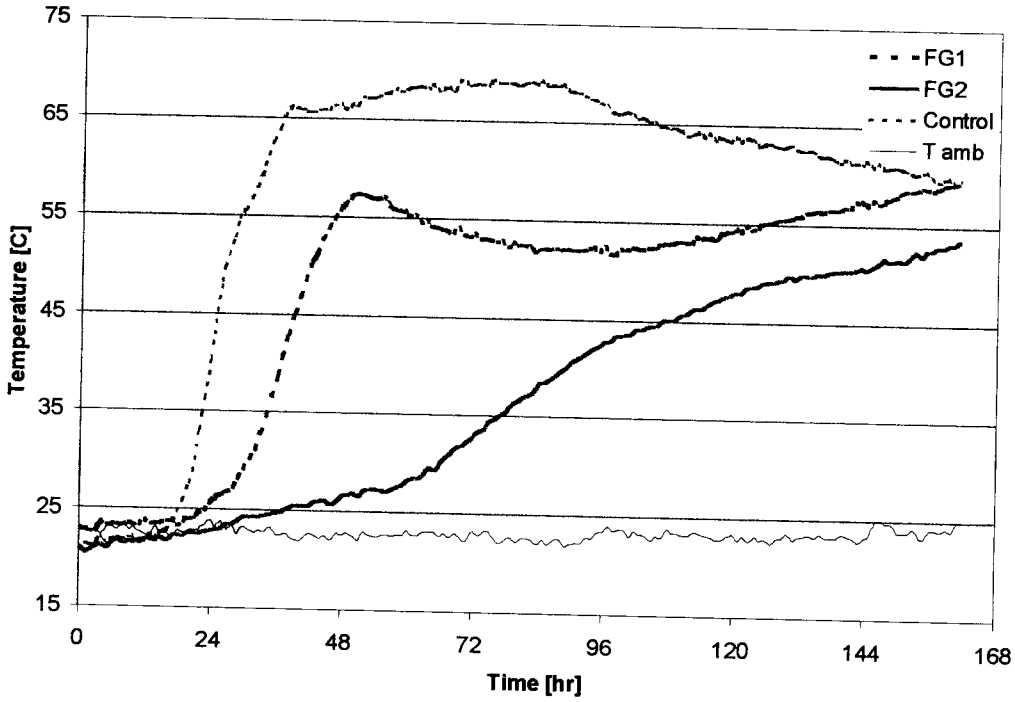

Figure 4.4 Temperature profiles for food waste and grease trap sludge treatments. Experimental set 4. 10% GTS (FG2): red; 5% GTS (FG1): green; Control: blue; room temperature: black.

Table 4.2. Carbon dioxide emission for yard trimmings and grease trap sludge treatments.

Treatment	Average CO_2 concentration (% vol)	Total mass emitted (g CO_2)	CO_2-C emitted as % of total initial carbon (%)
YG2 - 10%	4.4	108.4	10.3
YG1 - 5%	2.6	67.5	5.7
Control - 1%	1.4	40.6	6.1

Table 4.3. Methane emission for yard trimmings and grease trap sludge treatments.

Treatment	Average CH_4 concentration (ppmv)	Total mass emitted (mg CH_4)	CH_4-C emitted as % of total initial carbon (%)
YG2 - 10%	1.7	1.5	0.4×10^{-3}
YG1 - 5%	1.6	1.5	0.4×10^{-3}
Control - 1%	1.6	1.7	0.7×10^{-3}

Table 4.4. Nitrous oxide emission for yard trimmings and grease trap sludge treatments.

Treatment	Average N_2O concentration (ppmv)	Total mass emitted (mg N_2O)	N_2O-N emitted as % of total initial nitrogen (%)
YG2 - 10%	61.6	151.7	1.11
YG1 - 5%	0.6	1.5	0.01
Control - 1%	0.5	1.4	0.02

4.4.3 Ammonia Losses in the Exhaust Gases

Figures 4.8 and 4.10 show the concentration profiles for ammonia during the 168-hour composting period. Table 4.5 displays the average concentrations of daily ammonia measurements, as well as the total ammonia emissions. Total ammonia emissions were calculated using the average ammonia concentration and the total amount of air passing through the composting reactor for the 168-hour period of time.

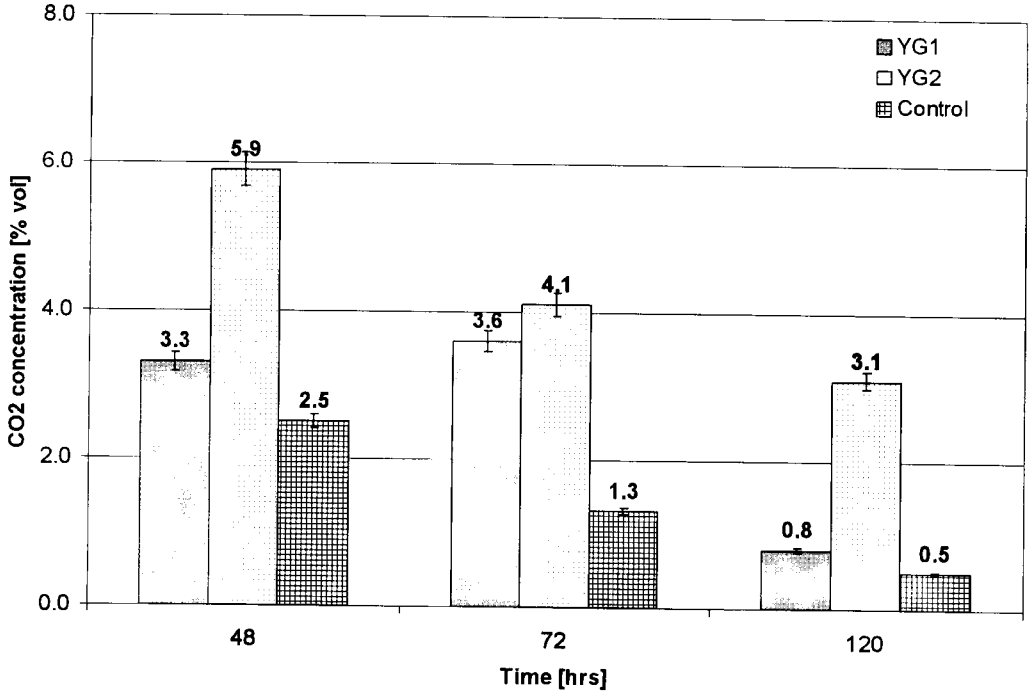

Figure 4.5 Carbon dioxide concentration for yard trimmings treatments. 10% grease addition (YG2): center column; with 5% (YG1): left column; no grease added (Control): right column. Error bars represent measurement error.

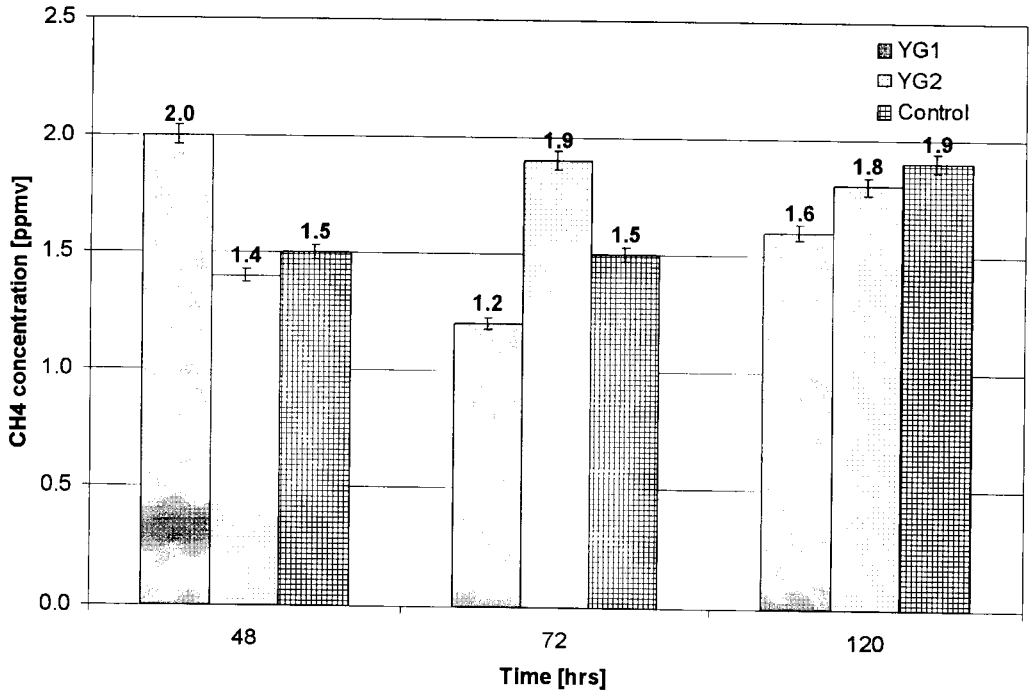

Figure 4.6 Methane concentration for yard trimmings treatments. 10% grease addition (YG2): center column; with 5% (YG1): left column; no grease added (Control): right column. Error bars represent measurement error.

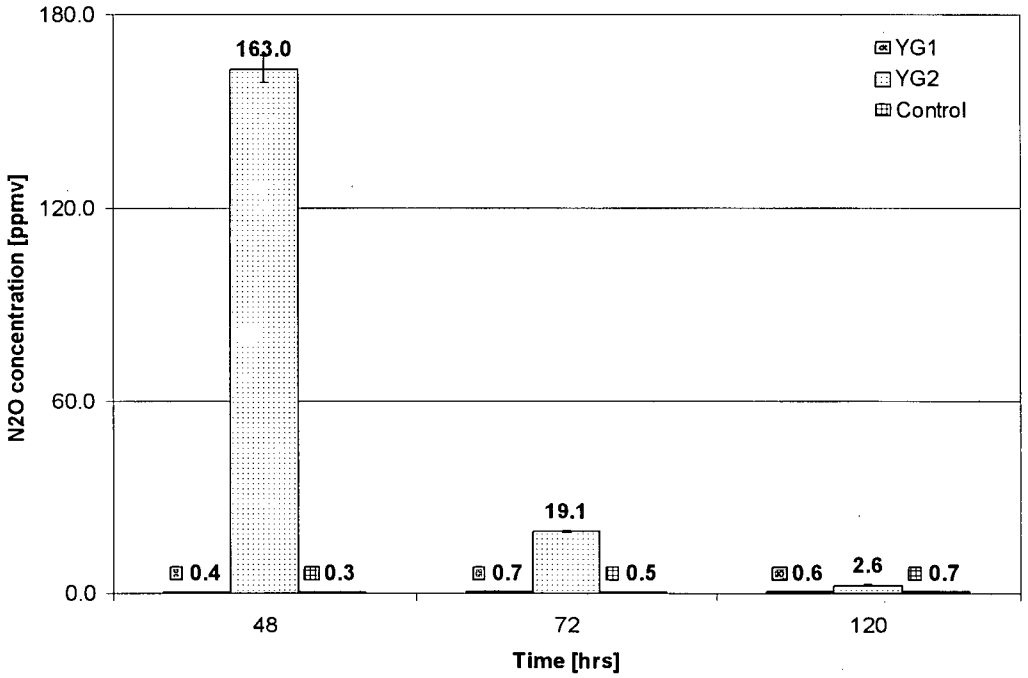

Figure 4.7 Nitrous oxide concentration for yard trimmings treatments. 10% grease addition (YG2): center column; with 5% (YG1): left column; no grease added (Control): right column. Error bars represent measurement error.

Table 4.5. Ammonia losses for all grease trap sludge treatments (n=6).

Treatment - lipid concentration (% ds)	Average NH_3 concentration (ppmv)	Total mass emitted (mg NH_3)	NH_3-N emitted as % of total initial nitrogen (%)	Average pH of the mix
YG2 - 10%	33.3	31.4	0.3	7.8
YG1 - 5%	352.6	353.1	3.1	8.1
Control - 1%	194.3	217.2	3.6	8.3
FG2 - 10%	12.5	17.7	0.1	4.8
FG1 - 5%	59.2	111.2	0.4	5.7
Control - 3%	142.5	253.0	0.8	7.1

Peak ammonia emissions occurred 48 hours after composting started for yard trimmings treatments, YG1, and its control. A comparison of Fig. 4.8 with Fig. 4.3 demonstrated that the timing for the peak ammonia emissions coincided with that for the peak temperatures. For the 5% grease treatment, ammonia concentration increased rapidly to 900 ppmv within 48 hours, and then gradually fell to 135 ppmv by hour 144;

these magnitudes are comparable to readings obtained by Kuroda et al. (1996). By comparison, the 10% grease treatment (YG2) had the highest temperature peak, but the lowest total ammonia emission. This might be explained by the greater lipid content; hence the larger amount of available carbon for immobilizing ammonia. Mahimairaja et al. (1994) found that the volatilization of ammonia was reduced by nitrogen immobilization whenever a carbon-rich source was added to manure composting mixes.

The values for cumulative ammonia emissions for the treatments with 5% ds or no grease trap sludge added ranged from 0.019 to 0.059 g NH_3 per L of reactor volume. In contrast, the treatments with 10% ds addition of grease trap sludge showed a cumulative ammonia emission of 0.003-0.005 g NH_3 per L of reactor volume. The values found in this study are much smaller than the values of 0.12 to 0.30 g NH_3 per L or reactor volume reported by Liang (2000) for cattle manure composting. The difference is probably due to the different substrate used.

As compared with the yard trimmings treatments, the food waste treatments resulted in smaller total ammonia emissions when grease trap sludge was added to the substrate (Table 4.5). Here, the control treatment achieved higher temperatures than the 5% and 10% grease treatments (Figs. 4.4 and 4.10), and it had total ammonia emission of 253.0 mg, a value similar to the total ammonia emission of 217.2 mg associated with the control treatment for yard trimmings treatments. The smaller total ammonia emission values for FG2 and FG1 could be due to the lower pH values of 4.8 and 5.7 exhibited during the high-rate composting phase, compared to a neutral pH of 7.1 for the control treatment (See Fig. 5.8, pp. 135).

Another difference between the yard trimmings and food waste composting treatments was that ammonia concentration reached a maximum at hour 48 for the former, versus hour 144 for the latter substrate. This might be linked to the temperature profiles, and is also evident in the different final C:N values for the 2 sets of experiments with values between 15.4 - 17.9 for yard trimmings, and 9.3 - 10.9 for food wastes mixes respectively; thus indicating a higher nitrogen concentration for the food waste treatments at the end of the experimental period (Fig. 5.5 and 5.6, pp. 134).

The calculation of total nitrogen gaseous losses for the yard trimmings treatments, in the form of ammonia and nitrous oxide are shown in Table 4.6. It is noticeable that the treatments YG1 and Control had very similar NH_3 and N_2O losses; while YG2 resulted in 10 times less NH_3 losses, and 100 times more N_2O losses when

compared with YG1 and Control. Nonetheless, YG2 had only about half the total nitrogen losses when compared with YG1 and Control. This might be explained by a larger nitrogen immobilization for the treatment with larger easily degradable carbon content (YG2).

Table 4.6. Total nitrogen losses due to NH_3 and N_2O.

Treatment	NH_3-N lost (% of initial N)	N_2O-N lost (% of initial N)	Total nitrogen losses (% of initial N)
YG2	0.3	1.11	1.41
YG1	3.1	0.01	3.11
YG1 - Control	3.6	0.02	3.62

4.4.4 Odor Concentration and Emission

The odor concentration profiles for the yard trimmings treatments exhibited a similar trend to those of the temperature profiles (Figs. 4.3 and 4.9), with peak odor concentration values occurring between 24 and 72 hours. In the case of food waste (Figs. 4.4 and 4.11), the peaks in odor concentration happened around hour 96, which was more than 48 hours later than the temperature peaks for treatments FG1 and Control.

For the yard trimmings treatments the ammonia and odor peaks (Figs 4.8 and 4.9) occurred at hour 72 for treatment YG2; while the odor peaks for YG1 and control treatments occurred 24 hours before the ammonia peaks. For all the treatments with food waste (Figs. 4.10 and 4.11) the odor peaks (at hour 96) preceded the ammonia peaks by 48 hours.

In terms of the cumulative specific odor emission (calculated as the addition of all 'ou' for a particular treatment, divided by the corresponding initial amount of dry matter), the values for the YG2 and the Control treatment where quite similar (Table 4.7), while the value for YG1 was slightly smaller. This trend is opposite to the ammonia emission one, where treatment YG1 emitted the largest amount of ammonia. The lack of correlation between ammonia and odor concentration profiles for the yard trimmings treatments is an indication that more pervasive odorants other than ammonia were present.

The opposite phenomenon was observed for the food waste composting mixes, where the control treatment emitted more odor than the treatments with lipid added. The cumulative specific odor emissions were one order of magnitude larger than the ones for yard trimmings. The cumulative specific odor emissions were very similar for the food waste treatments with grease trap sludge added, while the control treatment presented the largest odor emission. The unexpected behavior associated with FG1 and FG2 may correspond to the slow temperature climb and lower temperatures during the 168-hour composting period in these treatments.

The measured odor concentrations are compatible with typical odor levels found in enclosed (in-vessel) composting systems, and they indicate rather high concentration of odors emitting from the composting reactors with yard trimmings substrate, compared to synthetic food waste substrate. Despite an oxygen content of at least 15% detected in the headspace of the reactors, microenvironments within the compost mass could lead to the formation of reduced sulfurs (such as mercaptans and alkyl sulfides), and reduced carbonaceous compounds (such as butyric acid); which are pervasive odors having low odor thresholds.

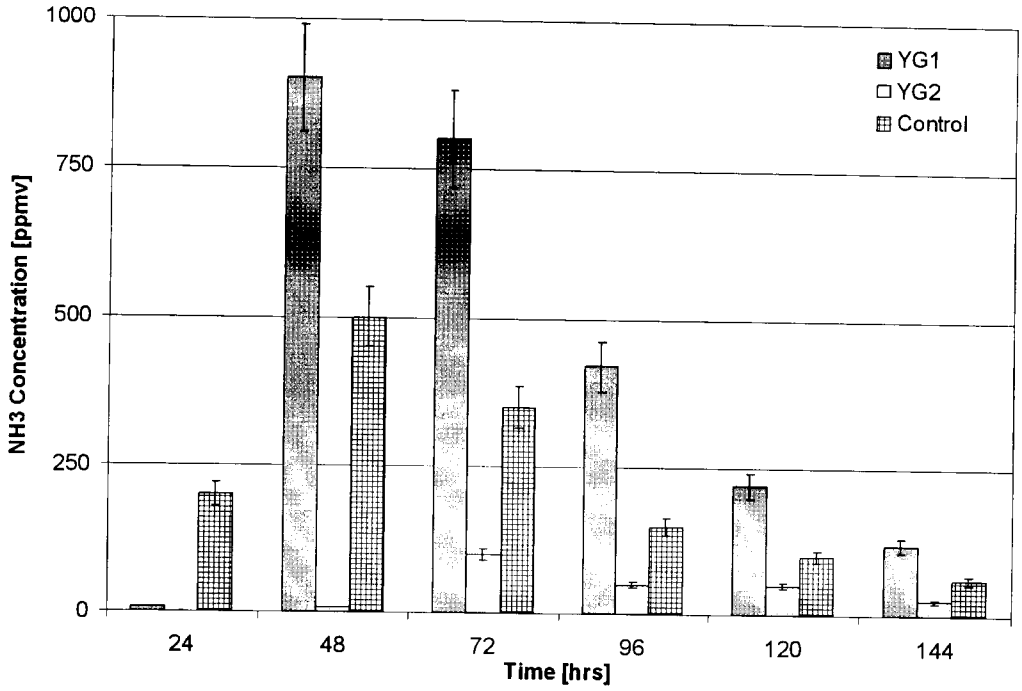

Figure 4.8 Ammonia concentration for yard trimmings treatments. 10% grease addition (YG2): left; with 5% (YG1): center; no grease added (Control): right.

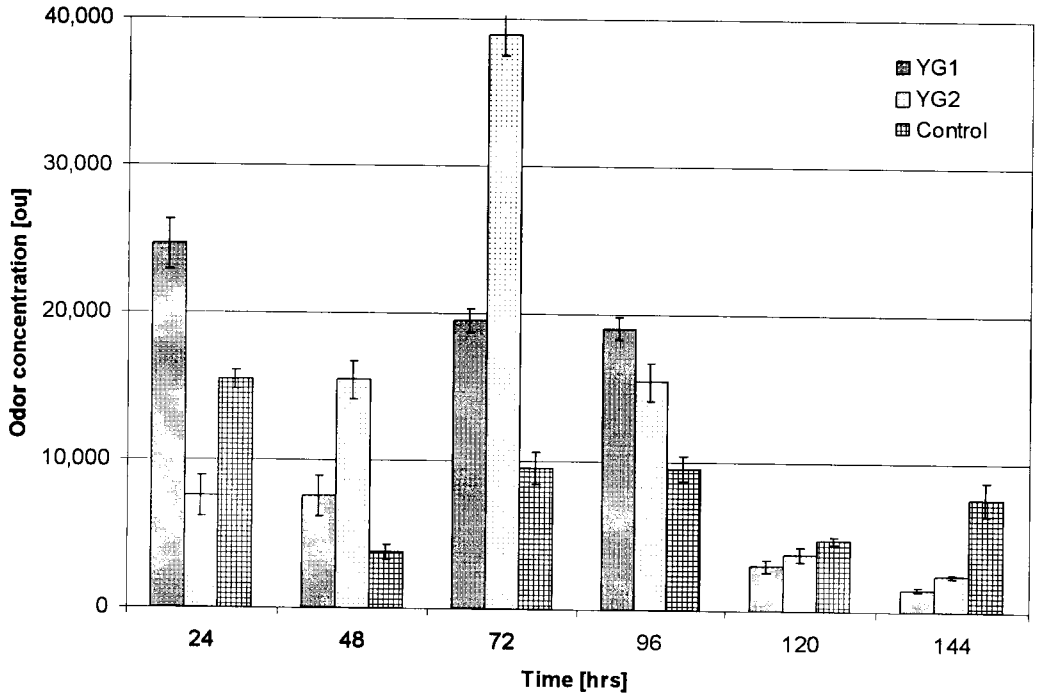

Figure 4.9 Odor concentration for yard trimmings treatments. 10% grease addition (YG2): left; with 5% (YG1): center; no grease added (Control): right.

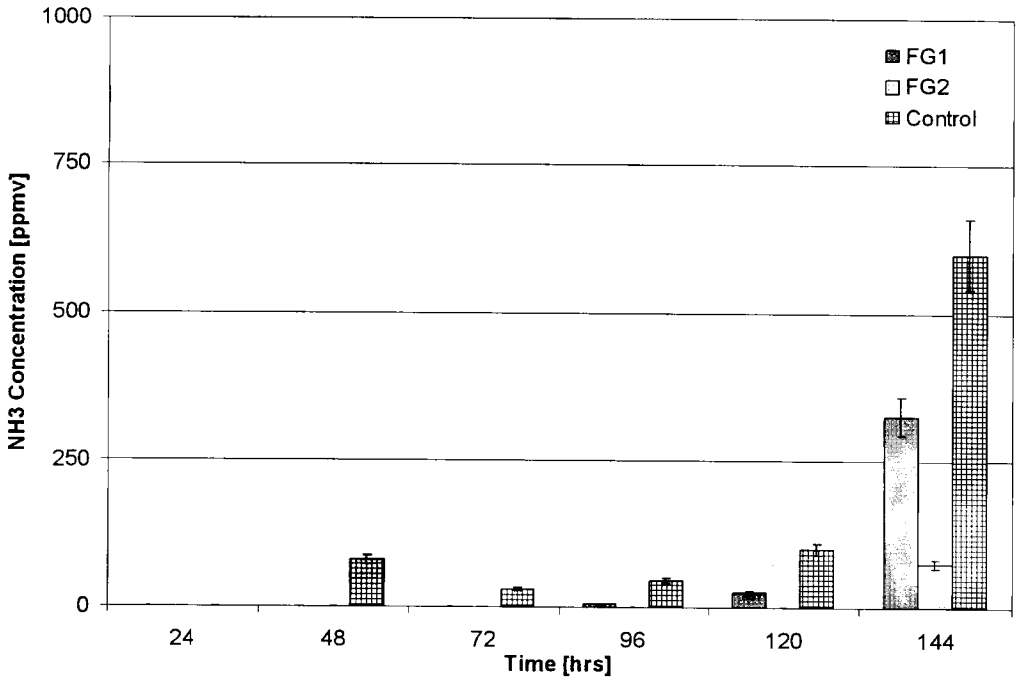

Figure 4.10 Ammonia concentration for food waste treatments. 10% grease addition (FG2): left; by 5% (FG1): center; no grease added (Control): right.

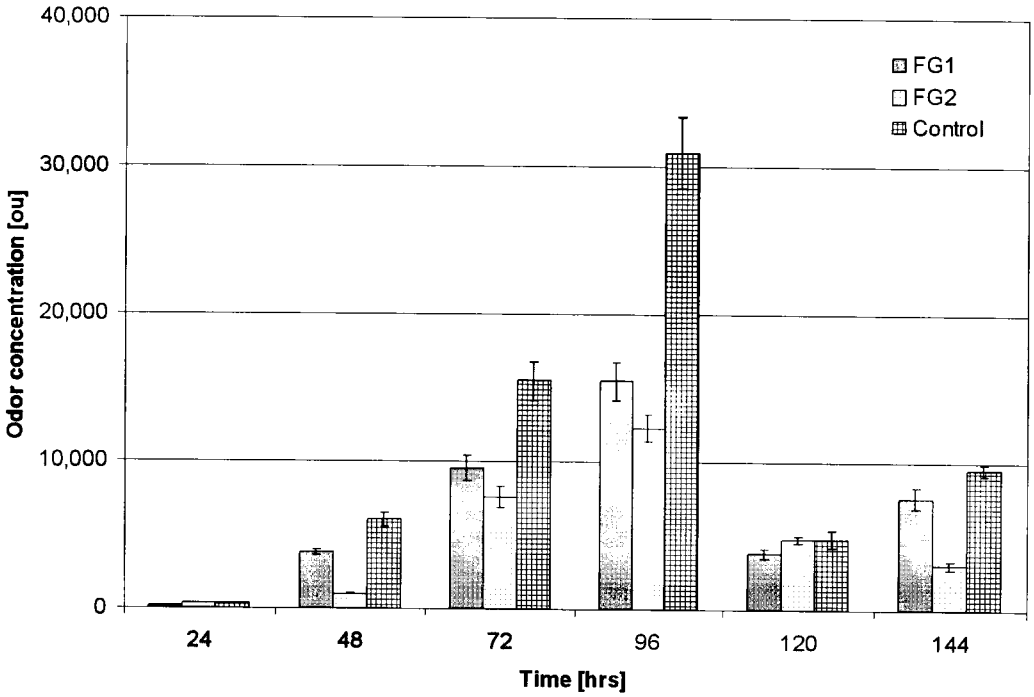

Figure 4.11 Odor concentration for food waste treatments. 10% grease addition (YG2): left; with 5% (YG1): center; no grease added (Control): right.

Table 4.7. Odor concentration and specific odor emission rate for grease trap sludge treatments.

Treatment (mass initial in kg ds)	Odor Odor flux Temp.	Time (hrs) 24	48	72	96	120	144	Mean	SD	Cumulative specific odor emission (ou/kg ds)*
YG2	ou	7586	15488	38905	15488	3802	2399	13945	13449	1.6×10^{-5}
(0.539)	ou/kg.min	510	1070	2761	1126	281	179	988	954	
	°C	24.5	54.2	65.3	64.9	64.1	60.6			
YG1	ou	24547	7586	19498	19055	3020	1479	12531	9722	1.2×10^{-5}
(0.605)	ou/kg.min	2493	781	2038	2015	232	159	1301	1001	
	°C	35.3	61.5	55.4	54.2	38.8	29.6			
Control	ou	15488	3802	9550	9550	4786	7586	8460	4189	1.5×10^{-5}
(0.340)	ou/kg.min	1687	420	1067	1082	549	878	947	452	
	°C	43.4	48.1	41.2	29.2	25.1	24.4			
FG2	ou	302	955	7586	12303	4786	3020	4825	4522	3.5×10^{-4}
(0.830)	ou/kg.min	20	63	506	829	326	205	325	304	
	°C	22.6	25.7	27.1	41.7	47.7	50.7			
FG1	ou	151	3802	9550	15488	3802	7586	6730	5400	3.9×10^{-4}
(1.029)	ou/kg.min	12	295	758	1258	317	651	548	439	
	°C	25.2	54.0	50.5	48.1	49.5	53.3			
Control	ou	302	6026	15488	30903	4786	9550	11176	10918	6.8×10^{-4}
(0.989)	ou/kg.min	23	479	1278	2660	430	893	961	936	
	°C	35.3	69.6	71.4	66.6	61.8	59.9			

Note: Average T_{amb} = 23.3°C. * Calculated as the sum of 'ou' per unit of initial dry mass.

4.5 CONCLUSIONS

The effect of adding lipid residues to composting mixes in terms of the impact on the air environment may be summarized as follows:

1. For treatments with yard trimmings substrate, the addition of 10% ds grease trap sludge to yard trimmings resulted in more nitrous oxide, more carbon dioxide, less ammonia, and similar odor emissions; when compared with the emissions of yard trimmings alone. Whereas, composting of yard trimmings with 5% ds grease trap sludge added had similar emissions (carbon dioxide, nitrous oxide, ammonia, and odor) as the control treatment. More specifically,

    a. The yard trimmings treatment with 10% ds grease trap sludge resulted in 10 times less ammonia emitted when compared with the treatment with 5% ds grease trap sludge and control treatment, with values of 0.3, 3.1, and 3.6 % on total initial nitrogen, respectively.
    b. The nitrous oxide emission for the yard trimmings treatment with 10% ds grease trap sludge added was consistently higher when compared to the treatment with 5% ds grease trap sludge. Furthermore, the nitrous oxide released was 100 times more when compared with the control treatment. The increased nitrous oxide emissions whenever lipids were added (at 10% ds) might be an indication of the presence of anoxic/anaerobic pockets in the composting mix.
    c. The nitrous oxide emissions for the yard trimmings treatments had similar values to the findings reported in the literature, with values of 1.11, 0.01, and 0.02 % of total initial nitrogen for the treatments with 10% ds, 5% ds grease trap sludge, and control, respectively.
    d. Yard trimmings treatment with 10% ds grease trap sludge added resulted in more carbon dioxide emitted (with a value of 10.3 CO_2-C as % of total initial carbon) when compared with the treatment with 5% ds grease trap sludge, and the control one (with values of 5.7 and 6.1 CO_2-C as % of total initial carbon, respectively). The higher carbon dioxide emission is an indicator that the addition of lipids (above 5% ds) resulted in enhanced microbial activity.

e. The amount of carbon lost as carbon dioxide (10.3 CO_2-C as % of total initial carbon) for the yard trimming treatments with 10% ds grease trap sludge added was similar to the value of the biodegradation extent for volatile solids, (with a value of 9% of initial volatile solids).
f. All the yard trimmings treatments (with or without grease trap sludge) produced similar methane emissions, with values very similar to the ambient ones (2 ppmv).
g. The odor emission trend for the yard trimmings treatments was similar to the trend of the temperature profiles, with peak values for these parameters in the first 72 hours after the composting start.
h. Cumulative specific odor emission for the yard trimmings treatments had similar values (1.6, 1.2, and 1.5 x 10^{-5} ou/kg ds initial mass, for 10% ds, and 5% ds grease trap sludge, and control treatments respectively).

2. For treatments with food waste substrate,

a. The addition of grease trap sludge to synthetic food waste resulted in less ammonia and odor emissions when compared to synthetic food waste alone. Nonetheless, the addition of grease trap sludge resulted in poor composting performance, and hence the lower emissions.
b. The ammonia losses for the food waste treatments were relatively small for the treatments with 10% ds, and 5% ds grease trap sludge, and control (with values of 0.1, 0.4, and 0.8 % of the total initial nitrogen, respectively). This small amount of ammonia emitted for the food waste treatments (with or without grease trap sludge) probably is due to the low pH values for the food waste treatments.
c. Cumulative specific odor emission for the food waste control treatment was larger than the value for the treatments with 10% or 5% ds grease trap sludge added (with values of 6.8, 3.5, and 3.9 x 10^{-4} ou/kg ds (initial mass), respectively). These values are one order of magnitude larger than the ones for the yard trimmings treatments.

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CHAPTER 5
EFFECT OF LIPIDS ADDITION ON COMPOST QUALITY

5.1 ABSTRACT

This chapter provides an overview of the changes in mass and nutrients for the composting process whenever lipid-rich wastes were added. Mass changes were studied by analyzing the total mass, water content, volatiles solids content, and the lipids content reduction. Finally, in order to measure the effect of the finished compost on potential plant growth, a phytotoxicity (seed germination) test was carried out.

In general, the addition of lipid wastes (up to 5% ds) to composting mixes did not create any unbearable problems, or major differences in the compost quality parameters examined. In contrast, the addition of grease trap sludge at 10% ds resulted in lower compost quality (according to the parameters measured), when compared with the treatments with 5% ds or no lipid added. Particularly, the addition of 5% ds grease trap sludge to either food waste or yard trimmings resulted in similar nitrogen changes when compared to the control treatments (no lipid added). Furthermore, the treatments with 5% ds or no lipid added resulted in improved root lengths and germination index for curly cress (*Lepidum sativum*) seeds when compared with the treatments with distilled water alone.

Lastly, treatments with 10% ds grease trap sludge added (with either food waste or yard trimmings) resulted in germination indices similar to the one for the distilled water treatment; but significantly smaller than the values for the treatments with 5% ds or no lipid added, showing that the addition of lipids at 10% ds does not have a beneficial effect in terms of curly cress (*Lepidum sativum*) seed germination.

5.2 INTRODUCTION

In the last two decades there has been a shift in the rationale of the composting process, from a waste management strategy to a resource recovery process. Composting has proven to be a very successful treatment process for mineral oil residues (hydrocarbons), and residues from oil extraction process (e.g. olive oil) (Filippi

et al. 2002, Wan et al. 2002, Kirchmann and Ewnetu 1998, Cegarra et al. 1996). However, composting of food residuals rich in oil and grease is not a common practice. The main constituents of lipid-rich wastes, such as fat, oil and grease residuals (FOGs), are animal fats and vegetable oils used in restaurants, institutions and industrial operations. These lipids include relatively simple compounds, with even or odd carbon chains in the C_{16} - C_{32} range; such as fatty acids, n-alcohols, n-alkenes, sterols, terpenes, fats, waxes, and resins (Lefebvre et al. 1998, Wakelin and Forster 1997, Fernandes et al. 1988).

The energy content of lipid-rich wastes almost doubles that of sugars and starch, thus giving them a thermodynamic advantage in terms of biological degradation (Wiley 1957). Jakobsen (1994) considered fats, oils, and grease residues (FOGs), which have a high concentration of readily available carbon and a high energy content, to be ideal substrates for aerobic composting. A few composting studies have included lipid-rich wastes in the form of wastewater scum or grease trap sludge. For instance, Viel et al. (1987b) observed 85% degradation of FOGs degradation during a 5-7 days composting period.

Notably, several authors (Joshua et al. 1994, Fernandes et al. 1988, Viel et al. 1987a,b) have reported that more organic matter could be conserved and converted to useful substances when its decomposition rate was at a low level relative to a high degree of lipid degradation. Similar findings were observed in this thesis research (Chapter 3).

The utilization of compost depends on its nutrient and organic matter contents. The value of compost as fertilizer is mainly based on its nitrogen content. Compost rich in nutrients, as indicated by nitrogen content > 1% (Barker, 1997), is used as fertilizer; otherwise it is preferred as soil conditioner when its organic matter content is high and nutrient content is low.

Nitrogen might be lost during the composting process, resulting in poor nutrient content in the finished product. It can be very significant with values as high as 50% of total initial nitrogen (Witter and Lopez-Real 1988). The carbon-to-nitrogen ratio (C:N) is an important parameter that affects nitrogen losses, with lower C:N values inducing greater nitrogen losses (He et al. 2000, Moller et al. 2000, Sommer and Dahl 1999).

Biological transformation of the nitrogen present in organic wastes is strongly linked to the carbon content, particularly to its availability. In the high-rate phase of

composting, nitrogen-rich materials such as proteins, are transformed by biochemical reactions. This decomposition is accompanied by a high rate of ammonification (Bishop and Godfrey 1983). Most of the nitrogen losses are a result of ammonia emissions (Martins and Dewes 1992). This ammonia loss has an impact on the agronomic value of the compost product, and as well, it produces a harmful effect on the environment (Jeong and Kim 2001, Boucher et al. 1999). Pel et al. (1997) stated that in order to control ammonia emissions, a better understanding of carbon mineralization and nitrogen transformation was needed. According to Liang (2000) and Okereke and Meints (1985), the incorporation of readily degradable carbon sources could lead to larger amounts of immobilized nitrogen.

Ekinci et al. (2000) found that ammonia losses increased rapidly for pH values greater than 8. This is supported by the chemistry of ammonia, in that the non-volatile ammonium ion is more prevalent under neutral to acidic conditions, and conversely volatile, molecular ammonia becomes predominant at basic pH values (Court et al. 1964). According to Jakobsen (1994) the formation of CO_2 and acetic acid during composting of oily wastes would result in acidic pH values. In this study (Chapter 4, Table 4.6. pp. 107) the yard trimmings treatment with grease trap sludge added at 10% ds caused lower ammonia (NH_3) emissions when compared with nitrous oxide (N_2O) emissions (0.3% vs. 1.1% of total initial nitrogen, respectively).

As suggested by Hao et al. (2001) more studies were needed to find composting strategies that resulted in less ammonia and nitrogen oxides emissions, while increasing the composting rate.

Jimenez and Garcia (1989) compiled a thorough review on compost maturity including physical parameters (such as temperature, color, and odor), chemical parameters (such as pH, C:N ratio, concentrations of nutrient and other compounds), and biological parameters (such as microorganisms count and germination test). Inbar et al. (1990) also suggested that one single parameter is not sufficient in determining the degree of maturity of compost; rather, a combination of several physico-chemical and biological parameters is preferred. Iannotti et al. (1994) emphasized the importance of conducting phytotoxicity measurements, since the characterization of a compost as 'stable' did not necessarily guarantee its potential for plant growth.

The aim of this study was to evaluate the effect of adding grease trap sludge to yard trimmings and food waste composting on compost maturity and quality.

5.3 MATERIALS AND METHODS

Details about the compost reactors and experimental setup, feedstock characterization and composting recipe formulation, along with composting process monitoring and control, have been described in Chapter 3 (Section 3.3.1). Treatments involved either yard trimmings or food waste as substrate, with grease trap sludge added.

The composting process consisted of two phases: a high-rate phase, lasting for 168 hours (7 days), and a curing phase, lasting for 126 days (4 months). The curing period was longer than the 3 weeks curing requirement of the B.C. Organic Matter Recycling Regulation (BCMWLAP 2002). The main variables studied were changes in nitrogen, C:N ratio, and lipid content. The phytotoxicity of compost extracts on seed germination was also studied.

5.3.1 Analytical Measurements and Calculations

Total composting mass was measured gravimetrically (Balance Model OHAUS I-10, Ohaus Corporation, Florham Park, NJ) before and after composting. Moisture content was measured by gravimetric analysis and oven drying at 101°C for 18-24 hours, according to APHA 2540B (APHA 1995). Lipid content was measured by the Soxhlet extraction method using n-hexane as solvent (test accuracy for restaurant wastes is ± 8%, USEPA 1998).

Volatile solids were measured by gravimetric analysis and ash content (ignition at 550°C for 2 hours, test accuracy ± 6.5%, APHA 2540E, APHA 1995). Carbon content was derived from volatile solids content, using Equation 1. The use of this equation for composting mixes gives carbon content results with 2-10% accuracy (Haug 1993). For the grease trap sludge treatments only, carbon concentration was measured using a CN Carlo Erba NA-1500 Analyzer (accuracy ±0.3%). On average the measured carbon concentration and that calculated using Equation 1 differed by 10% (See Appendix B for more detail).

$$\%\ Carbon = \frac{100 - \%\ Ash}{1.8} = \frac{\%\ Volatile\ Solids}{1.8} \qquad (1)$$

Nitrogen content in the composting materials and compost was measured using a Total Nitrogen analyzer (either a LECO FP228 Nitrogen Determinator, Leco Corp., St. Joseph, MI, accuracy ± 2.0% of the measured N value, or a CN Carlo Erba NA-1500 Analyzer, accuracy ± 1.6%). Ammonia and nitrate in the compost were measured in 2M KCl extracts (1:40 solids to solution), using an ion analyzer (Model QuickChem 8000, Lachat Instruments, Zellweger Analytics Inc., Milwaukee, WI), and QuikChem Methods (for NH_3, #10-107-06-1A with 0.1% precision; and for NO_3, #10-107-04-2-A with 0.5% precision).

The compost pH's was measured using the pH protocol for compost samples as proposed by Liang (2000), by which the pH is measured in 2 water extracts at different dilutions (ratio of solids to distilled water of 5 g to 200 ml for pH_{200}, and 5 g to 600 ml for pH_{600},). A straight line is fitted between the 2 pH values in order to find the 'y axis' intercept (no dilution) value which corresponds to the pH of the compost solids.

The C:N ratio was estimated by Equation 2, using the total carbon content calculated from Equation 1, and the measured total nitrogen content, both on a dry basis.

$$C/N = \frac{Total\ carbon\ content}{Total\ nitrogen\ content} \qquad (2)$$

Ammonia in the exhaust gases was measured using gas detection tubes (precision ± 10%, Part No. 10-100-15-3M, RAE Systems Inc., Sunnyvale, CA), whereas nitrous oxide in the exhaust gases was measured using a gas chromatograph (HP 5840A) with an electron capture detector, and a 2m x 3.2mm OD stainless steel column packed with Porapak Q 80-100 mesh (Measurement precision was 2.5% for a 1.1 ppm N_2O standard).

Phytotoxicity was measured via a modification of the germination test (Zucconi et al. 1981a,b, Spohn 1969), as suggested by Manser and Keeling (1996), and Shiralipour and McConnell (1990). The modification refers to an increase in the incubation period from 24 to 48 hours, to facilitate root length measurement. Briefly, the germination test was as follows: water extracts (4:1, distilled water to compost) were mixed for 1 hour in an orbital shaker, and later filtered using Whatman No.1 filter paper. Two ml of a 10% v/v dilution of the water extracts was poured into Petri dishes (10 cm Φ) lined with filter

paper. Grebus et al. (1994) found that dilutions less than 10% almost always resulted in a germination index smaller than the 40% break point (large germination inhibition).

Six seeds of either radish (RD672C, *Raphanus sativus*) or curly cress (MS495D, *Lepidum sativum*) were placed in the Petri dishes (7 replicates per sample). Seeds were purchased from West Coast Seeds Ltd., Delta, BC.

The samples were incubated for 48 hours in the dark at 27 ± 2°C. After that period, 1 ml of a 50% v/v ethanol solution was added to halt the seeds germination. The status of seed germination (yes or no), and root length, in [mm] were measured and compared to blank samples that had only distilled water added to the seeds, without any compost sample.

A germination index (GI) of 40% or less would indicate a potential phytotoxicity (Zucconi et al. 1981a,b). The germination index was calculated using the following formulae:

$$Germination = \frac{Average\ germinated\ seeds\ in\ sample}{Average\ germinated\ seeds\ in\ blank} \quad (3)$$

$$Root\ Length = \frac{Average\ root\ length\ in\ sample}{Average\ root\ length\ in\ blank} \quad (4)$$

$$Germination\ Index = GI = Germination * Root\ Length * 100 \quad (5)$$

5.3.2 Experimental Design

Three manipulated variables were chosen based on the values suggested in the literature, and from previous experimental treatments: 1) substrate type, 2) presence or absence of lipid-rich waste as grease trap sludge (GTS), and 3) initial GTS concentration. Substrates used were yard trimmings (YW, mainly grass clippings), and synthetic food waste (FW, dry dog food). Chicken litter was used as inoculum at 3% wet basis. Details on the composition of the composting mixes were presented in Table 3.4 (Chapter 3, pp. 50).

The main response variables were: (1) temperature profile, (2) overall changes in total wet mass and moisture content, (3) overall changes in volatile solids and lipids, (4) nitrogen in the compost product, (5) changes in the carbon-to-nitrogen ratio and pH, and

(6) compost phytotoxicity (as an indicator of compost quality). A total of six experimental treatments were performed, as shown in Table 5.1.

Table 5.1 Experimental treatments used for compost quality study.

Treatment	Main substrate	GTS added	Lipid concentration (% ds)
YG2	YW	Yes	10
YG1	YW	Yes	5
Control	YW	No	1*
FG2	FW	Yes	10
FG1	FW	Yes	5
Control	FW	No	3*

*Lipids naturally occurring in the raw materials. YW: yard trimmings, FW: Food waste, GTS: grease trap sludge.

5.4 RESULTS AND DISCUSSION

5.4.1 Temperature Profiles

The temperature profiles, mass changes (total mass, water content, volatile solids, and lipids), and the kinetic and thermal parameters for the high-rate phase of composting were shown in detail in Chapter 3 (Sections 3.4.1 to 3.4.2). Figures 5.1 and 5.2 display the temperature profiles during the high-rate phase only, for the yard trimmings treatments (YG1, YG2, and Control), and food waste treatments (FG1, FG2, and Control) respectively.

Qualitatively the temperature profiles were similar for all the experimental treatments, following a typical temperature vs. time curve for composting. The control treatments for yard trimmings (Treatment 'Control') performed in the mesophilic range with temperature peak around 49°C. In contrast, the yard trimmings treatments with lipid added (treatments YG2 and YG1) were in the thermophilic range with peak values between 61-67°C.

In the case of synthetic food waste substrate, the control treatment produced a higher peak temperature than the ones with lipid waste added (Treatments FG1 and FG2). Considering the fact that dry dog food already has higher lipid content than yard trimmings, reduced porosity due to the presence of significant amount of lipids could be the reason for these observations versus the opposite phenomenon in yard trimmings

tests with GTS. As discussed in Chapter 3 (Section 3.4.1.5) the addition of GTS at 10% ds to synthetic food waste seemed to inhibit the composting process.

Although the compost temperature during the curing phase was not monitored, a sharp decrease to ambient temperature was observed in the hours following the transfer of the composting mix from the Dewar bioreactors to the curing reactors.

5.4.2 Total Mass and Moisture Content Changes

Table 5.2 presents a summary of the total mass reduction values calculated as percentage of the total initial mass. Wet mass reductions (as percentage of initial mass) were between 9.8 and 25.3% for the high-rate phase, and between 38.0 to 49.0% for the curing phase, with overall mass reduction of 52.3-63.3%. These overall mass reduction values are in agreement with that reported by Haug (1993) and Rynk (1992) to be about 50%.

It is noticeable that the yard trimmings control treatment lost less total mass during the active phase when compared with the treatments with lipid waste added (YG1 and YG2). In contrast, for the food waste treatments the control and the treatments with 5% ds grease trap sludge added had very similar overall mass reduction values (62.6 and 62.4% respectively). The overall mass reduction for FG2 had a lower value (53.6%) when compared with treatments 'control' and FG1.

As illustrated in Table 5.3, the overall losses in water were fairly large with values above 92%. Water content losses were between 20.3 to 36.2% for the active phase of composting, while the bulk losses of moisture occurred during the curing period. Moisture contents of the cured compost were in the low range of 7-10%, resulting in a very dusty and dry product.

Table 5.2. Total mass reduction (as percentage of total initial mass wet weight).

Treatment -	Active phase		Curing phase		Overall reduction	
lipid content	Mean	SD	Mean	SD	Mean	SD
YG2 - 10%	14.2	1.3	49.0	3.9	63.2	5.2
YG1 - 5%	25.3	4.2	38.0	4.2	63.3	8.4
Control - 1%	18.4	4.1	33.9	2.8	52.3	6.9
FG2 - 10%	9.8	4.1	43.8	2.6	53.6	6.7
FG1 - 5%	18.0	3.3	44.4	2.8	62.4	6.1
Control - 3%	24.5	1.4	38.1	4.1	62.6	5.5

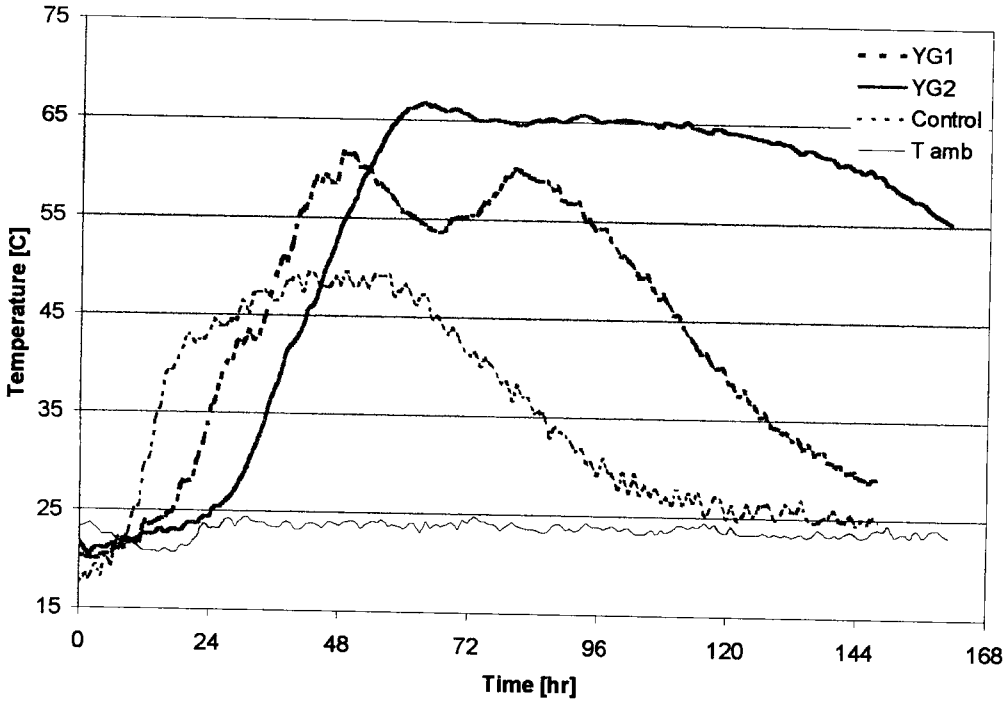

Figure 5.1 Temperature profiles for yard trimmings and grease trap sludge treatments. Experimental set 3. 10% GTS (YG2): red; 5% GTS (YG1): green; Control: blue; room temperature: black.

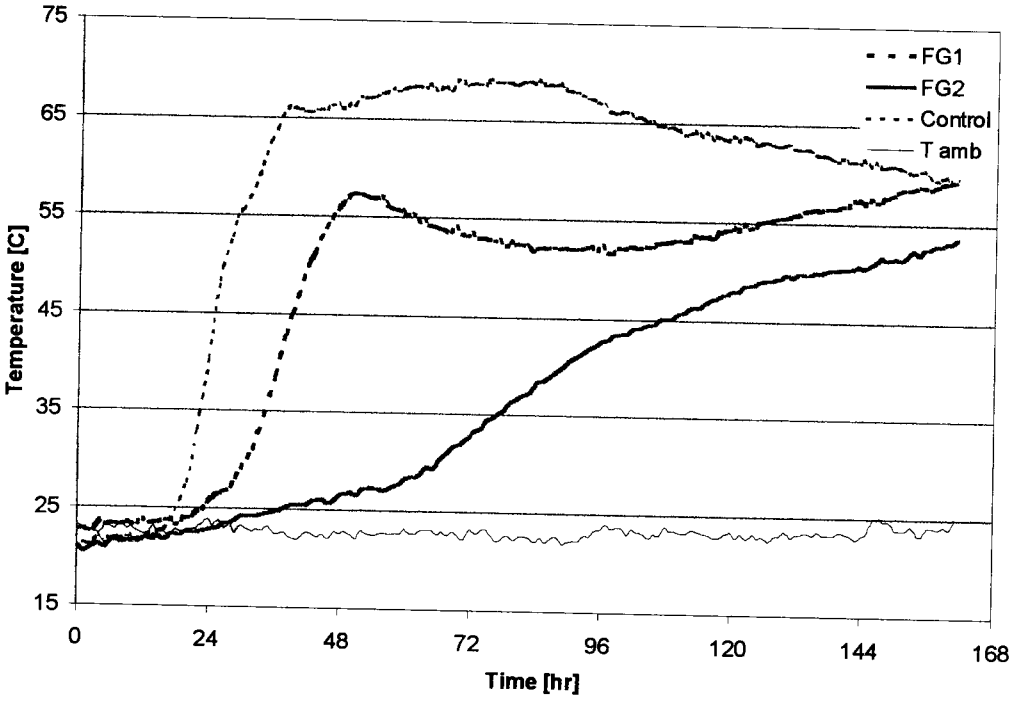

Figure 5.2 Temperature profiles for food waste and grease trap sludge treatments. Experimental set 4. 10% GTS (FG2): red; 5% GTS (FG1): green; Control: blue; room temperature: black.

Table 5.3. Water content reduction (as percentage of total initial water).

Treatment -	Active phase		Curing phase		Overall reduction	
lipid content	Mean	SD	Mean	SD	Mean	SD
YG2 - 10%	20.3	4.5	74.2	0.1	94.5	4.6
YG1 - 5%	29.3	4.8	65.5	0.5	94.8	5.3
Control - 1%	24.5	4.5	68.0	0.7	92.5	5.2
FG2 - 10%	31.6	1.2	62.5	0.3	94.1	1.5
FG1 - 5%	26.7	5.1	67.6	0.5	94.3	5.6
Control - 3%	36.2	4.5	58.8	0.5	95.0	5.0

5.4.3 Volatile Solids and Lipids Changes

Table 5.4 presents a summary of the volatile solids changes based on the initial volatile solids content. For the active phase, volatile solids reduction for the yard trimmings treatments ranged from 9 to 21% while the food waste treatments reductions ranged from 7 to 13%. The reductions in volatile solids during the curing period were between 2 to 32%. Note that the treatments with lower values for volatile solids reduction value for the high-rate phase (YG2, FG1 and FG2) showed the largest values for volatile solids for the curing period. Indicating that the presence of FOGs might inhibit the degradation of volatile solids during the high-rate phase of composting. However, it seems that after the high-rate phase, or after a certain amount of FOGs is degraded, the inhibitory effect diminishes allowing for more volatile solids degradation during the curing phase.

The overall reduction in the volatile solids content for both substrates with lipid added ranged from 21 to 41%, which coincides with the findings of Viel et al. (1987b) that only a small amount (20%) of organic matter as volatile solids was degraded when composting flotation foams (lipid waste). In addition, Fernandes et al. (1988) found organic matter reductions of 15-44% of initial dry weight. The yard trimmings treatments with lipid added exhibited higher overall volatile solids reductions, when compared with the control treatment, while all the food waste treatments showed similar values (21-22%).

Table 5.4 Volatiles solids reduction (as percentage of total initial volatile solids).

Treatment - lipid content	Active phase Mean	SD	Curing phase Mean	SD	Overall reduction Mean	SD
YG2 - 10%	9	3	32	0	41	3
YG1 - 5%	21	5	8	0	29	5
Control - 1%	13	2	2	0	15	2
FG2 - 10%	7	2	14	1	21	3
FG1 - 5%	7	2	15	1	22	3
Control - 3%	13	5	9	0	22	5

Lipid waste concentration was reduced by 43 to 96% for all treatments during the entire composting period (See Table 5.5) The lipid reduction of 96% for the yard trimmings control treatment is not typical; its numeric value is so high due to the low initial lipid content for this treatment. It was observed that food waste treatments had the lowest lipids biodegradation values (10-29%) for the high-rate phase of composting. For the same phase, the yard trimmings treatments with lipids added had lipids reductions of 39-51%. These values are similar to the values of 43 to 95% lipids biodegradation reported in the literature (Fernandes et al. 1988, Viel et al. 1987b, Wiley 1957).

All the experimental treatments, but FG2, had the highest reduction of lipids during the high-rate phase (vs. the curing phase) suggesting that, as reported by Haug (1993), lipids were readily degradable under aerobic composting conditions.

Table 5.5. Lipids reduction (as percentage of total initial lipid content).

Treatment - lipid content	Active phase Mean	SD	Curing phase Mean	SD	Overall reduction Mean	SD
YG2 - 10%	39	5	9	0	48	5
YG1 - 5%	51	4	26	1	77	5
Control - 1%	96	19	0	0	96	19
FG2 - 10%	10	8	45	2	55	10
FG1 - 5%	27	8	16	1	43	9
Control - 3%	29	4	15	1	44	5

Note that the yard trimmings treatment with no lipid added had almost double the overall lipid decomposition (96%) when compared with the treatment of food waste with

no lipid added. This was probably, because there were hardly any lipids to start with, and analytical errors could have magnified this result. For both substrates (yard trimmings and food waste) the addition of grease trap sludge at a concentration of 5% ds produced similar lipids reduction values when compared to the control treatments.

5.4.4 Nitrogen Concentration Changes during Composting

Figures 5.3 and 5.4 illustrate the changes in nitrogen concentration for all treatments and the two composting phases studied. Note that total nitrogen concentration increased (relative to its initial concentration) for all the treatments (except for a small reduction during curing for treatment YG2 - See Table 5.6). Overall, the nitrogen increases were between 34-76% for the yard trimmings treatments, and 29-33% for the food waste ones.

Table 5.6. Total nitrogen changes (as percentage of total initial nitrogen).

Treatment -	Active phase		Curing phase		Overall change*	
lipid content	Mean	SD	Mean	SD	Mean	SD
YG2 - 10%	66	11	-32	4	34	15
YG1 - 5%	34	7	20	2	54	9
Control - 1%	37	16	39	14	76	30
FG2 - 10%	15	2	14	11	29	13
FG1 - 5%	30	0	3	1	33	1
Control - 3%	28	14	3	1	31	15

*A positive number indicates an increase; a negative one means a reduction.

For all the treatments most of the nitrogen concentration increase happened during the high-rate phase of composting. Vuorinen and Saharinen (1999) found that the increase in nitrogen content per kg of dry solids during composting is due mainly to a decrease in total amount of carbon, which has been converted to CO_2. In addition, Mahmood et al. (1991) found a net increase in nitrogen content of 33-38%. This might be explained by the fact that solids disappeared at a faster rate than nitrogen. In my study, the net increase of nitrogen varied from 29 to 76%.

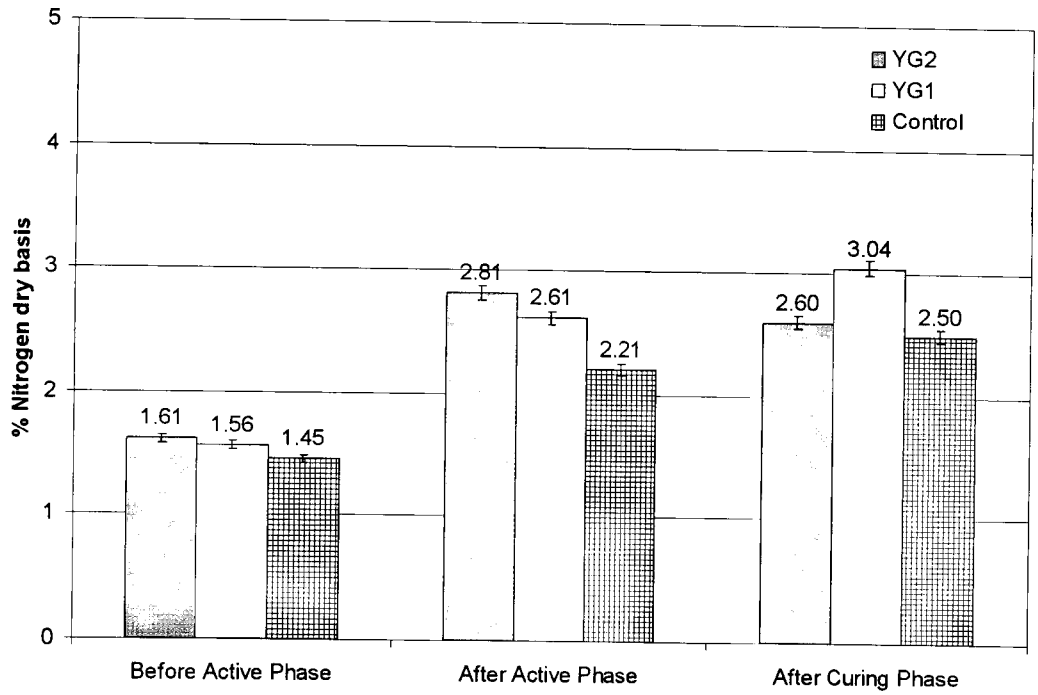

Figure 5.3 Nitrogen concentration (% ds) for yard trimmings treatments. YG2: left column, YG1: center column, Control: right column.

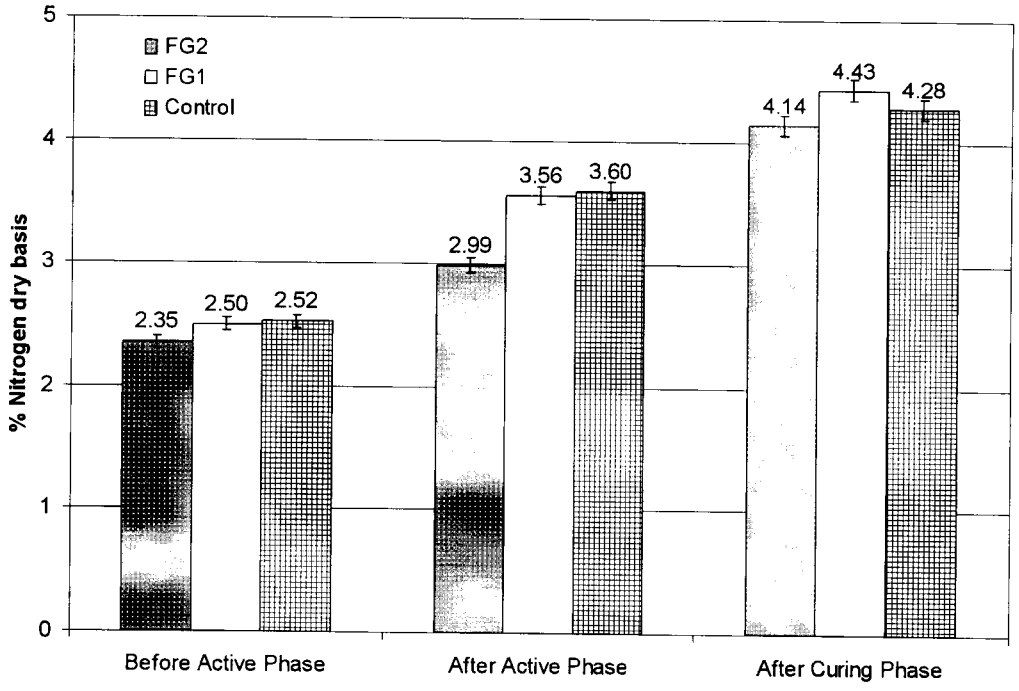

Figure 5.4 Nitrogen concentration (% ds) for food waste treatments. FG2: left column, FG1: center column, Control: right column.

For the yard trimmings treatments, the higher nitrogen concentration net increase (76%) was for treatment control, the same one that had the highest ammonia losses. However, the yard trimmings control treatment also had the lowest total volatile solids reduction (15%). Conversely, the lowest net nitrogen increase (34%) was for YG2, the same treatment that had the lowest ammonia losses (when compared with to Control and YG1), and furthermore YG2 had the largest reduction in total volatile solids (41%).

Table 5.7 presents a summary of the ammonia and nitrate content for the cured compost produced by all experimental treatments. There is a 2 to 3 order of magnitude difference between the concentrations of ammonia compared with nitrate content. This might be caused by the high initial ammonia production during the high-rate phase (through the breakdown of organic nitrogen compounds like proteins). In comparison, mineralization of nitrogen is a very slow and demanding process that usually occurs late in the curing process (Mathur et al. 1993).

Table 5.7. Extractable ammonia and nitrate in the compost product.

Treatment - lipid content	Ammonia in compost			Nitrate in compost		
	mg/kg ds	As % of initial TN	As % of final TN	mg/kg ds	As % of initial TN	As % of final TN
YG2 - 10%	2128	11	7	152	0.8	0.10
YG1 - 5%	9750	50	27	39	0.2	0.03
Control - 1%	6805	48	23	25	0.2	0.02
FG2 - 10%	7548	29	16	54	0.2	0.03
FG1 - 5%	10753	31	21	48	0.1	0.02
Control - 3%	11135	34	22	74	0.2	0.04

Liao et al. (1994) found NH_3 values of 10,300 mg/l (after 141 days) and NO_3 of 50 mg/L, for fish waste and sawdust compost. Additionally, Levi-Minzi et al. (1992) reported NO_3+NO_2 values of 0.1% of initial total nitrogen, and NH_4^+ values of 1.5% of initial total nitrogen, for city refuse composted for 120 days. In this study, the extractable nitrate concentration (as percentage of the initial total nitrogen) was between 0.1 to 0.8%, similar to the values reported in the literature. However, the ammonia values were relatively larger, between 11 to 50% of the initial total nitrogen. The large

concentration of extractable ammonia in the final compost might be caused by the slightly acidic pH values for the end product.

5.4.5 Carbon-to-Nitrogen and pH Changes during Composting

Figures 5.5 and 5.6 confirm the expected trend of C:N changes. Initial C:N measured values were around the optimum of 25, and decreased to about 15 through the composting process for the yard trimmings treatments, and to around 10 for the food waste ones. The final C:N values for the yard trimmings treatments are in accordance with the vector attraction requirement for 'Class A Compost' of the B.C. Organic Matter Regulation (BCMWLAP 2002), which requires final C:N values from 15 to 35. Thus, curing the compost for more than 21 days, 126 days in this case; and having a C:N of approximately 15-20 fulfilled the vector attraction reduction requirement.

The changes in pH are illustrated in Figures 5.7 and 5.8. Generally, the trend in composting is from slightly neutral, to acidic, and finally to a slightly basic pH (Haug 1993). For this study the final pH values were in the slightly acidic region. This is expected since the grease trap sludge alone had a pH of about 4; hence the lower pH values for the treatments with higher GTS added. Jakobsen (1994) notes that during protein breakdown large amount of carbon dioxide and bicarbonate ion would be forming; thus producing a decrease in the pH values.

5.4.6 Phytotoxicity Test using Cress and Radish Seeds.

One purpose of compost production is to close the loop of organic matter recycling, meaning to return to the land at least some of the organic matter removed. There is no standard test on how to check for the 'goodness' of compost for land application. However, Zucconi et al. (1981a,b) proposed to test the germination of rapid-growing seeds to evaluate the potential acute phytotoxicity of compost.

The results of phytotoxicity tests, in this case germination tests are presented in Table 5.8. Two different seeds, curly cress (*Lepidum sativum*), and radish (*Raphanus sativum*) were used. In either case the critical value for the germination index, % GI, was set at 40% according to Zucconi et al. (1981a,b). This means that treatments with % GI lower than 40% would likely present a phytotoxic effect to plants. A 100% GI value corresponds to the Blank, or distilled (DI) water tests.

Table 5.8. Germination test results using curly cress (*Lepidum sativum*) and radish (*Raphanus sativum*). (n = 42 each seed)

Treatment	Germination*			Root Length**			% GI
	Mean	SD	%G	Mean	SD	%RL	
Curly Cress (*Lepidum sativum*)							
Blank (distilled water)	0.8	0.4	100	10.3	8.8	100	100
YG2 - 10%	0.8	0.4	103	9.3	7.7	90	93
YG1 - 5%	0.7	0.5	88	17.8[a]	14.1	173	152[a]
Control - 1%	0.7	0.4	91	18.6[a]	14.4	180	164[a]
FG2 - 10%	0.7	0.4	91	12.9	11.9	125	114
FG1 - 5%	0.8	0.4	94	14.9[b]	10.5	144	136[b]
Control - 3%	0.8	0.4	100	15.5[b]	11.9	151	151[b]
Radish (*Raphanus sativum*)							
Blank (distilled water)	1.0	0.0	100	24.7	8.6	100	100
YG2 - 10%	1.0	0.2	98	22.1	9.8	90	88
YG1 - 5%	1.0	0.0	100	27.7	9.9	112	112
Control - 1%	1.0	0.2	95	28.1	14.1	114	108
FG2 - 10%	1.0	0.0	100	26.8	8.5	108	108
FG1 - 5%	1.0	0.2	95	21.0	10.3	85	81
Control - 3%	1.0	0.2	98	26.2	10.1	106	104

* Seeds germinated are coded as 1, non-germinated as 0. ** Root length units are mm.
a: refers to means significantly different than the blank mean at a 0.01 confidence level.
b: refers to means significantly different than the blank mean at a 0.05 confidence level.

Statistically significant differences for germination and root length between the blank treatment and the compost treatments were tested using single factor ANOVA and t-test. For the case of germination, there were no statistically significant differences among treatments for either seed. However, among the root length means (for curly cress seeds), only treatments YG1 and Control, and FG1 and Control, were significantly higher than the blank treatment root length. Indicating that the addition of compost extract from compost without lipids added and with lipids added at 5% might be of benefit in terms of root elongation for Cress seeds.

None of the treatments exhibited phytotoxic potential based on the Germination Index, since all GI values were well above the 40% threshold. In terms of the GI for curly cress seeds, the addition of compost from yard trimmings or yard trimmings with 5% ds grease trap sludge added, resulted in improved results when compared to the GI of the treatment with distilled water alone (blank). The same improvement in the GI values is observed for the treatments FG1 and Control.

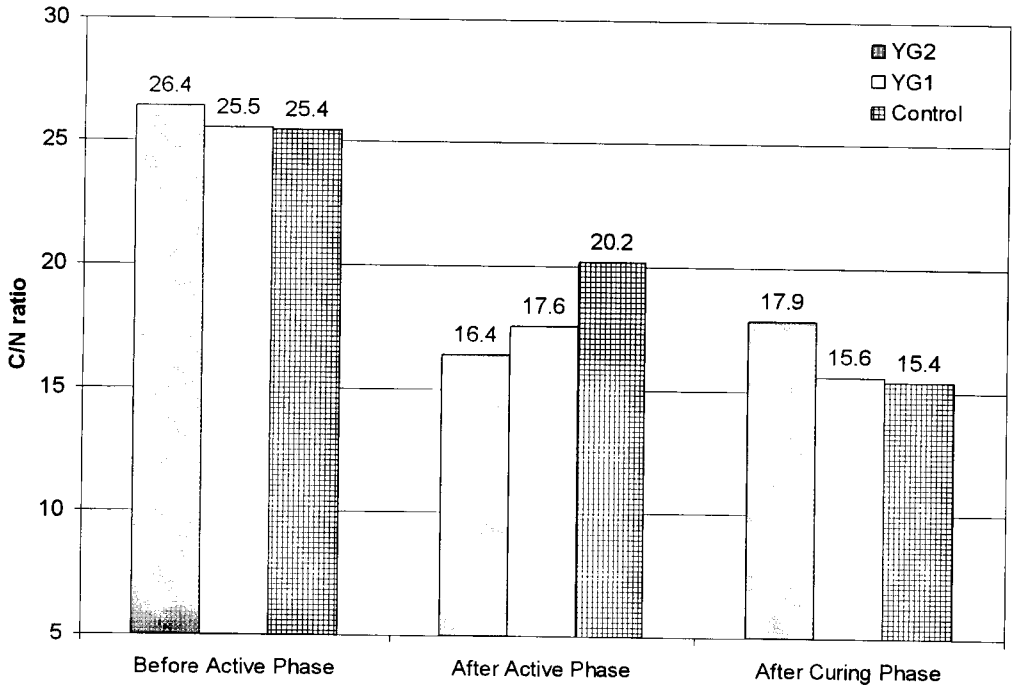

Figure 5.5 Carbon-to-nitrogen ratio during composting for yard trimmings treatments. YG2: left column, YG1: center column, Control: right column.

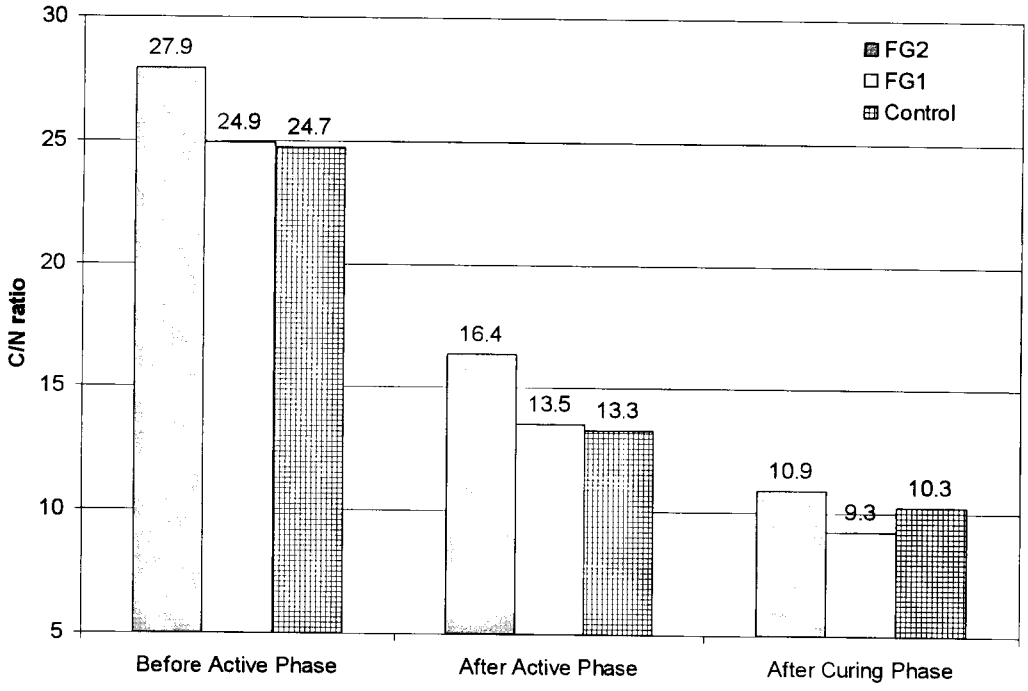

Figure 5.6 Carbon-to-nitrogen ratio during composting for food waste treatments. FG2: left column, FG1: center column, Control: right column.

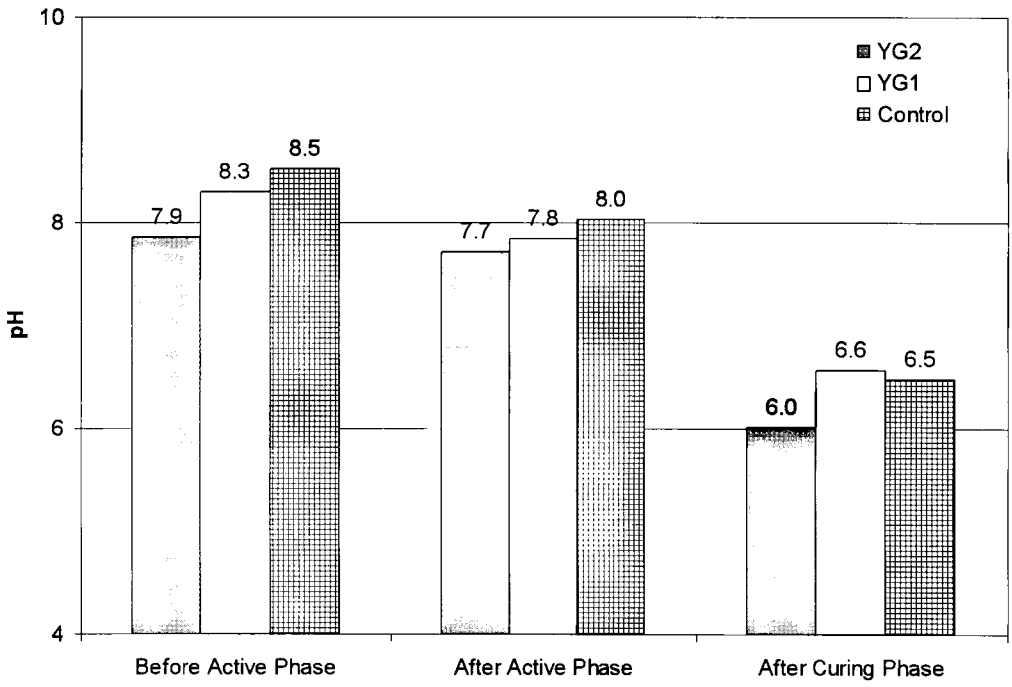

Figure 5.7 pH changes during composting for yard trimmings treatments. YG2: left column, YG1: center column, Control: right column.

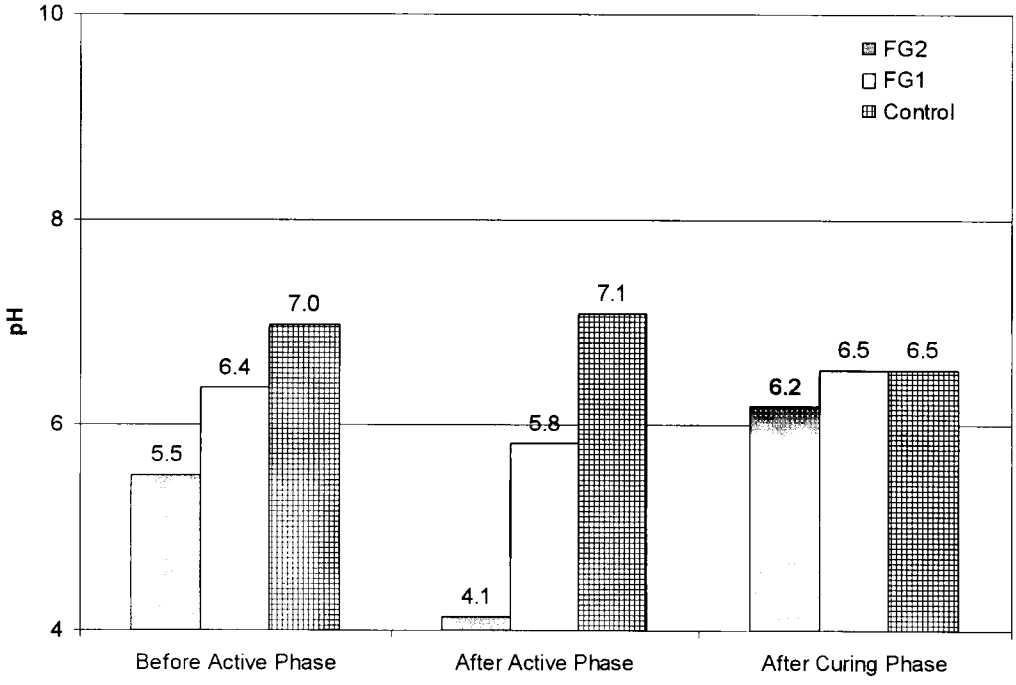

Figure 5.8 pH changes during composting for food waste treatments. FG2: left column, FG1: center column, Control: right column.

The addition of 10% ds grease trap sludge to either yard trimmings or food waste composting, resulted in no benefit in terms of the results of the percentage germination index (GI) for the curly cress seeds (when compared with the blank). In addition, treatments YG2 and FG2 showed GI values significantly smaller than those for treatments with 5% ds or no lipid added, thus indicating that the addition of grease trap sludge at 10% ds to composting substrates might be detrimental to cress seed germination.

For the radish seeds germination and root length, results were statistically similar among all treatments; hence radish seeds were not sensitive to phytotoxicity potential testing in this case.

5.5 CONCLUSIONS

The following summarizes the main conclusions derived from this Chapter:

a. Temperature profiles were higher for the treatments with lipids added to the yard trimmings substrate. In the case of food trimmings treatments, there seemed to be a need to lower the bulk density of the composting mass to improve the composting process.
b. Overall (for the high-rate and curing phases) mass changes for all the treatments were between 52.3 and 63.3% of the total initial wet mass. This is in agreement with the value of 50% reported in the literature.
c. Overall water content reductions were above 92%, resulting in very dusty and dry compost.
d. Yard trimmings treatments with grease trap sludge added (at 5 or 10% ds) had the largest overall volatile solids reductions (29 and 41% respectively), when compared with the yard trimmings control treatment (15%). The food waste treatments had very similar values for volatile solids reduction (21-22%).
e. Lipids reduction for the entire composting period was between 43 and 77% for the treatments with grease trap sludge added, and between 44 and 96% for the control treatments (food waste and yard trimmings respectively).
f. For yard trimmings, the addition of grease trap sludge at 10% ds resulted in the largest increase in nitrogen concentration (66%) when compared to the control treatment (37%), for the high-rate phase. However, in terms of overall nitrogen

changes, the yard trimmings treatment with 10% ds grease trap sludge added resulted in an increase of only 34%, when compared to 76% for the control treatment.

g. The addition of 5% ds grease trap sludge to either food waste or yard trimmings resulted in similar overall nitrogen changes when compared to the control treatments.

h. Treatments with 5% ds lipids addition or no lipid added, resulted in improved root lengths and germination index for Curly cress seeds when compared with the treatments with distilled water alone.

i. Treatments with 10% ds grease trap sludge added (with either food waste or yard trimmings) resulted in germination index similar to the one for the blank treatment (distilled water), but significantly smaller than the values for the treatments with 5% ds or no lipid added, thus showing that the addition of lipids at 10% ds does not have a beneficial effect in terms of curly cress seed germination.

j. Radish seeds' germination, root length, and germination index values were statistically similar for all treatments; hence radish seeds were not affected by the different treatments examined.

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CHAPTER 6
BIODEGRADATION COMPOSTING MODEL

6.1 ABSTRACT

This chapter concerns the modeling and simulation of the composting process. Dynamic models are useful engineering tools to improve the understanding of a process and the relationships between process variables. Laboratory experiments can provide data such as organic content, composition, biodegradability, thermal properties, and heat production rate of the substrate. The simulation model can then use these results in a predictive mode to help in the analysis and optimization of the process, and guide the engineering design of composting facilities. In this thesis, the objective of modeling was to determine if experimental observations could be explained by process dynamics, and whether the performance of the composting system could be improved given the operating parameters, and hence better experimental design could be recommended for future experimental studies.

To the knowledge of this author, no previous modeling and simulation studies have been performed on lipid-rich organic wastes during composting. A composting model was adapted from the literature (Haug 1993), which is capable of handling two different types of easily decomposable substrates, with different compositions, biodegradabilities, biodegradation rates, and heat of combustions. The use of two different substrates is of interest in the case of lipid-rich wastes, since their properties are quite different to other more commonly compostable substrates such as yard trimmings and food waste.

The model showed the impact of changes in the kinetic and thermal parameters on the composting process as represented by the temperature profile, energy production, and biodegradation rate. The results of the model were compared with the experimental results reported in Chapter 3. The developed model simulated fairly well the trends of temperature profiles, temperature peak, mass changes, and the final moisture content. However, it underestimated the values of the mass biodegraded. In addition, the model predicted an opposite trend for the 'time to reach the peak temperature' when compared with the actual data.

The limitations of the model were mainly due to its 'macrokinetic' approach by which only the biodegradation rate coefficients (of solids and oil) are used to represent the microbial and chemical changes.

6.2 COMPOSTING PROCESS MODELING

Previous studies on lipid-rich waste composting (Fernandes et al. 1988, Viel et al. 1987a,b) have not addressed the modeling of the composting process, particularly when an energy rich amendment was added. The composting process has been modeled by several authors, mainly following the modeling methodology suggested by Haug (1993). However, more recent approaches include modeling for process optimization (Keener et al. 2002), and modeling composting kinetics using a deductive approach (Hamelers 2002). Composting modeling includes kinetic as well as thermal parameters, to represent the changes in the mass and energy balances.

6.2.1 Process Kinetics

The composting process has been modeled by a number of authors, including the classic methodology used by Haug (1993). Most of the computer models have been used to describe kinetics, which deals with the rate of biodegradable matter decomposition during composting, and hence provides the driving force for the composting process.

Haug (1993) used a 'macrokinetic' model to simulate the composting process. The 'macrokinetic' model uses a single kinetic parameter (the rate of disappearance of dry matter per unit of compostable matter per unit time, also called biodegradation rate coefficient, 'k', in units of time^{-1}) to describe and predict the changes in the composting mass over time.

The rate of mass disappearance is affected by parameters such as substrate composition, carbon-to-nitrogen ratio, temperature, water content, free air space, microbial population and diversity, and carbon dioxide and oxygen concentrations in the composting matrix (Ekinci 2002, Keener et al. 2002, Haug 1993). The biodegradation rate coefficient is affected by those parameters in a multiplicative manner, represented as follows:

$$k = k_r * F1 * F2... * Fn \quad (1)$$

where 'k_r' is the rate of mass disappearance at a reference temperature in (day^{-1}), and 'F1' to 'Fn' represent the different mathematical functions for the effect of the different physico-chemical, and biological parameters.

Haug (1993) adopted as 'k_r' the values for the biodegradation rate coefficient obtained through BOD, COD, or similar respirometric analyses. In particular, BOD data is readily available for a variety of liquid organic wastes. These values are often referred as 'k_{20}' or 'k_{25}', as they are obtained at 20-25°C, generally under anaerobic conditions. Haug (1993) reported k_{20} values ranging from 0.0015 to 0.0699 day^{-1} for several substrates, bearing a difference of one order of magnitude, which is attributed to the different types of organic wastes. These substrates usually have a fraction that will decompose slowly (like Kraft papermill sludge with k_{20} of 0.0015 day^{-1}), and a fraction that will decompose relatively fast (like Rye grass with k_{20} of 0.0699 day^{-1}).

Furthermore, Haug (1993) related the 'rate of mass disappearance' to the mass of biodegradable volatile solids (BVS), implying that all biodegradable volatile solids were potentially degraded during composting.

The first order reaction equation as proposed by Haug (1993) is as follows:

$$\frac{d(BVS)}{dt} = -k * (BVS) \quad (2)$$

where 'BVS' is the biodegradable fraction of the volatile solids, in (kg ds).

In contrast, Keener et al. (1992) proposed an slightly different first-order reaction model (Eq. 3) for mass disappearance during composting, by introducing the concept of 'equilibrium mass' (m_e); which implies that the microbial degradation of organic matter is limited. Thus implying that there is a fraction of organic matter that does not disappear after a relatively long composting period of 6 months to 1 year.

$$\frac{d(m_c)}{dt} = -k * (m_c - m_e) \qquad (3)$$

where 'm_c' is the mass of composting mix at any given time, in (kg ds); 'm_e' is the equilibrium mass, in (kg ds); and 'k' is the biodegradation rate coefficient in (day^{-1}).

Keener et al. (1997) summarized the measured values for the rate of mass disappearance for several composting studies (with durations from 14 to 54 days), which ranged from 0.012 to 0.111 day^{-1} for various combinations of organic wastes. However, the predicted 'k' values using the equilibrium mass concept (Eq. 3) were 0.022 to 0.259 day^{-1}, and were higher than the actual 'k' values for similar substrates. The differences between the predicted and actual 'k' values were attributed to the uncertainty in the estimation of the value for the equilibrium mass.

Marugg et al. (1993) extended the model of Keener et al. (1992), onto the composting of yard waste (mixture of grass clippings, brush, leaves, and cardboard). Based on pilot-scale test data obtained using 208-L bioreactors. The 'k' values calculated using energy and materials balances lay between 0.165 to 0.190 day^{-1}. The authors concluded that the use of published heats of combustion gave accurate results for the energy balances.

From the previous discussion it can be concluded that knowledge of the rate of mass disappearance, or biodegradation rate coefficient (k), is critical for the modeling of the composting process. However, the determination of the 'k' values is still largely an estimation process, and depends on frequent monitoring of the total composting mass for its calculation; which is not always possible with small composting masses, and it might require 'destructive' sampling. Several research studies have used the rates of oxygen consumption to calculate the mass degraded over time for composting processes (Bari et al. 2000a,b, VanderGheynst et al. 1997).

Using a first order reaction kinetic model, Hamoda et al. (1998) examined the kinetics of municipal solid waste composting in order to find the relationship between temperature, particle size, moisture content, and C:N ratio on the biodegradation rate coefficient. Those authors found that the biodegradation rate coefficient (with values between 0.048 to 0.330 day^{-1}) presented the highest values at a temperature of 40°C, a 60% moisture content, a particle size of 40 mm, and a C:N ratio of 30. This study

provided very insightful information on the effect of different parameters on the values of the biodegradation rate coefficient, however the results could not be readily extrapolated since a relatively small composting mass of 0.5 kg ww was used.

Bari et al. (2000a,b) studied the dependence of the biodegradation rate on temperature for different aeration modes, and considered kinetic models with zero, first, and second orders of reaction. It was found that the degradation during composting could be adequately predicted using a first order reaction model. Their observations were in agreement with the following findings: Marugg et al. (1993) for yard trimmings; Keener et al. (1996) for a variety of substrates including yard waste, food or kitchen waste, chicken manure and biosolids; Boni and Musmeci (1998) for the organic fraction of municipal solid waste; Hamoda et al. (1998) for municipal solid waste; and Ndegwa et al. (2000) for caged layer manure. First-order reaction kinetics were used in all of the above-mentioned modeling studies, with various degree of success in validating the model outputs with the actual data.

Ekinci et al. (2002) used a second order kinetics decomposition model to investigate recirculated airflow effects on moisture, temperature, and decomposition profiles in the composting system (with paper mill sludge and broiler litter as substrate); but followed Haug's (1993) handling of the key environmental factors, such as temperature, moisture content, oxygen content, and free air space.

6.2.2 Thermal Performance

The thermal behaviour of the composting process is associated with the amount of energy exchanged during the composting process. For in-vessel composting, this energy is essentially a product of the exothermic microbial biodegradation; which results from the breakdown in the chemical bonds of the organic molecules.

Pathogenic microorganisms will be inactivated or killed if enough heat is generated and allowed to accumulate (for a certain period of time) in the composting mass. In the case of in-vessel composting, the BCMWLAP (2002) regulation stipulates a composite temperature-time requirements of 55°C or above for at least 3 days. However, it shall be noted that excessively high temperatures can be detrimental to the composting process, since most of the 'beneficial' organisms would be killed at temperatures above 65-70°C (Gray and Biddlestone 1971).

McKinley and Vestal (1984) found that temperature is the most critical parameter affecting the composting process and compost quality. They found that higher levels of microbial activity took place at mesophilic temperatures between 25 and 45°C, and that there was a relatively low level of activity for thermophilic temperatures of 55 to 74°C. However, it has been suggested by several authors that the optimum temperature for composting should be between 55 and 60°C (Bach et al. 1985, Finstein et al. 1983, McGregor et al. 1981, Suler and Finstein 1977, Jeris and Regan 1973, Wiley 1957). For lipid-rich waste, such as composting mixtures containing 8% ww flotation foams, Viel et al. (1987b) found that the highest degree of microbial activity took place at 60-70°C. For modeling purposes, Bari et al. (2000a) found that temperature is the parameter with the largest influence on the rate of mass biodegradation.

The amount of heat generated during composting may be estimated using an energy balance; which accounts for heat production, heat accumulation in the composting mass, and heat loss. Viel et al. (1987b) found that the energy release during composting of mixtures containing 8% wet weight of flotation foams was 4180 kJ.kg dry matter^{-1}.week^{-1} (25 kJ.kg^{-1}.hr^{-1}). VanderGheynst et al. (1997) found an energy release value of 9500 kJ per kg of oxygen consumed (equivalent to 14.3 MJ.kg vs^{-1}, assuming an oxygen demand of 1.5 kg O_2.kg vs^{-1}). For comparison with other forms of organic carbon (carbohydrates), Kaiser (1996) used a heat generation value of 14,000 kJ per kg of oxygen consumed (approximately 21 MJ.kg vs^{-1}), based on the values of the heats of oxidation of glucose and cellulose.

Energy losses are mainly due to conduction/convection, and latent heat of vaporization associated with water vapor in the exhaust gases. A critical parameter for the heat lost is the overall heat transfer coefficient (U), which includes the effects of convective and conductive heat transfer. While composting materials are considered to have low thermal conductivity and hence good insulation properties (Haug 1993, Hogan et al. 1989, Mears et al. 1975), improving the insulation of the composting reactor may reduce heat loss due to conduction/convection.

6.2.3 Dynamic Modeling

Dynamic modeling is an integration of the process kinetics with heat and mass balances and thermal parameters. The majority of composting models are based on the heat and mass balances summarized by Haug (1993). The kinetics of the composting

process are included in the model under a 'macrokinetic' approach, meaning that the kinetics of the process are described solely by the biodegradation rate coefficient (k) of the first order biodegradation kinetics.

Nakasaki et al. (1987) used a 31.5-L reactor for the experimental study of composting a mixture of sludge, seed, and rice husks. They observed that the temperature difference between the upper and bottom parts of the reactor differed by 2°C at the most, and proposed a model having identical air and solid temperatures all the time within the reactor. Their model could predict CO_2 evolution rate, volatile matter conversion, temperature, and moisture content under various aeration operation conditions, and it was applied to determine the optimum conditions for substrate decomposition and effective drying.

Hogan et al. (1989) studied the physical modeling (based on process thermodynamics) of the composting ecosystem in order to compare the system behaviour in lab trials (using ground rice hulls and rice flour as substrate, on a 19-L reactor), and the modeled behaviour. The authors found that the behaviour of the composting system could be reasonably simulated by using the physical model, and that the bulk of heat losses (76-88%) were due to latent heat losses in the exhaust gases.

In 1994, Person and Shayya developed a user-friendly computer package COMPOST® as a tool for design, management, and educational purposes. The proprietary software is based on the composting process model equations used by Haug (1993), and allows for the comparison of 'what if' scenarios for different manually-input initial conditions for the composting process. Among the model outputs are the amount of mass consumed, the water loss, and airflow needs. The authors did not present any information on the software validation by comparing the model output with actual composting data. Furthermore, the model does not have provision for predicting the composting process temperature profile.

The model proposed by Stombaugh and Nokes (1996) was a breakthrough in the field of composting since it introduced a simple dynamic model based on the microbial growth kinetics as the driving force of the composting process. Their model simulated with a relatively high level of accuracy, the changes in temperature, oxygen uptake, moisture exchange, and substrate degradation. However, their model involves a large

number of biological parameters (growth and maintenance of biomass) that are not readily available in the literature, and such inputs needed to be estimated.

Kaiser (1996) developed a simulation model for the composting mass as a microbial ecosystem. The model was very detailed in the microbial aspect, and included the degradation of 4 substrates (sugar and starches, hemi-cellulose, cellulose, and lignin) by a 4-component microflora (bacteria, actinomycetes, brown-rot fungi, and white-rot fungi). The model predicted the composting dynamics fairly well; nevertheless, it had the same limitations as the model developed by Stombaugh and Nokes (1996), which required the assumption of the values for a large number of biological parameters. Even with the breakdown of the composting substrate into 4 subgroups, the model did not take into consideration other important main subgroups of the substrates, namely proteins and fats.

VanderGheynst et al. (1997) developed an energy transport model for a high solids, aerobic degradation process in order to study the compost temperature's spatial and temporal changes. The model included terms for the amount of heat generated from microbial activity, energy accumulated in the compost matrix, and energy lost through the exhaust gases. The authors validated the model, with a large degree of success, by comparing simulated temperature profiles to experimental temperature profiles obtained from pilot-scale composting of a mixture of dog food and wood chips in a 770-L reactor. The changes on mass degraded were based on measurements of oxygen consumption and carbon dioxide generation.

Das and Keener (1997) used a numerical model for the dynamic simulation of a large scale composting system, with an aim to solve, among others, the airflow pattern, and hence the aeration energy requirements and costs. The model was successfully validated at a commercial composting facility with a mixture of biosolids, bark, and sawdust.

Liang (2000) adopted the mathematical model of Stombaugh and Nokes (1996) for simulating substrate decomposition; however, modeling work was extended to consider nitrogen transformation, incorporating factors that influence ammonia volatilization. Microbial growth on substrate was described using Michaelis-Menten or Monod-type kinetics; oxygen was treated as a rate-limiting factor similar to substrates. Substrates were subdivided into three groups: soluble C, non-fibre C, and fibre C. Like Haug (1993), temperature and water content were defined as state variables. In

addition, three correction factors were incorporated to account for sub-optimal environmental conditions due to temperature, moisture content, and pH. Independent action of the three factors was assumed and realized by multiplication. The model was deemed to be successful in reflecting well-known phenomena of the composting process. The disadvantage of this model was also the requirement for estimating a large number of parameters.

6.3 MODELING AND SIMULATION PROCEDURE

To the knowledge of this author, no previous modeling and simulation studies have been performed on lipid-rich organic wastes during composting. The model presented here integrates the process kinetics with conservation principles (heat and mass balances). In addition, the model allows for the input of 2 or more different substrates (in this case for food waste or yard trimmings, with lipid-rich wastes), with different biodegradation rates, biodegradabilities, heats of combustion, and molecular formulae.

6.3.1 Assumptions Used in the Mathematical Model

1. Lumped capacity system for a bioreactor, which has a uniform temperature and moisture content. In other words, the substrate mixture is homogeneous, and no temperature gradients or moisture gradients exist in the composting mass.
2. The oxidation of the biodegradable volatile solids 'BVS' (or its biodegradation rate) is assumed to be first order with respect to the quantity of BVS.
3. Only the fast degrading fraction of the organic matter is being modeled for the active phase of composting; the slow degrading fraction associated to the curing phase is outside the scope of this model.
4. Changes in particle size and hence density of the composting mass are not modeled.
5. The model accounts for the effects of temperature, moisture content, free air space, and aeration rate as environmental parameters; whereas the C:N ratio and pH are not modeled.
6. Loss of water via surface evaporation is assumed negligible in an enclosed composting system.

7. Loss of water via leachate is assumed negligible, given the observations made in the experimental treatments.
8. Modeling and simulation are done for the active phase of composting, defined as the time during which the composting substrate is readily available for microbial decomposition.
9. Oxygen content was considered not to be a limiting factor, since according to Haug (1993); oxygen content is not likely a limiting factor if its concentration is above 5%. As shown in Chapter 3 (pp. 67 and 69), the experimental treatments had oxygen contents always above 10%.

6.3.2 Model Description and Equations

The model was mainly adopted from Haug (1993) for the dynamic modeling of the composting process. It starts with a heat balance of the composting process:

$$Heat_{accumulated} = Heat_{produced} - Heat_{lost} \quad (4)$$

$$Q_a = M * C_p * \frac{dT}{dt} = Q_p - Q_l \quad (4a)$$

where 'Q_a' (MJ) is the heat accumulated by the composting mass, 'M' (kg ww) is the total mass (comprised of solids, water and oil), 'C_p' ($MJ.kg^{-1}.{}^{o}C^{-1}$) is the specific heat of the mixture, 'T' (°C) is the temperature of a compost mixture, 'Q_p' (MJ) is the biological heat produced within the compost mass, and 'Q_l' (MJ) is the sensible heat loss plus latent heat loss to the surroundings.

The main objectives of modeling were to simulate the composting mix temperature profile, and the changes in mass (solids, oil, and water). Hence, the composting mix temperature (T) was one of the modeled variables. Equation 4b illustrates how the composting mix temperature was calculated. The subscript 't' refers to the current time step, and 't + dt' refers to the next time step.

$$T_{t+dt} = T_t + \frac{Q_{a,t}}{(M * C_p)_t} \quad (4b)$$

The compost mass, 'M' (kg ww), changes during the composting process due to biodegradation of volatile solids and oil, and net loss of water; which may be determined by a mass balance (See Eq. 5). The expression '$M*C_p$' is calculated (Eq. 4c) based on the average mass per component per time step multiplied by the appropriate 'C_p' value (which is assumed to be constant for the relatively small temperature range, 20-70°C, of the model). The subscript 'i' refers to each major substrate component of the composting mix out of a total of 'n' components (oil, water, solids).

$$(M * C_p)_t = \sum_{i=1}^{n} \left(\frac{M_t + M_{t+dt}}{2} * C_p \right)_i \qquad (4c)$$

The heat produced, Q_p, was estimated from the heat of combustion of the materials (Q_{cs} and Q_{co}, in MJ.kg ds^{-1}, for solids and oil respectively) multiplied by the amount of material degraded; following the suggestions of Ndegwa et al. (2000) who stated that the heat liberated by the microbial degradation is equivalent to the amount of energy released if the materials were to be combusted, and Marugg et al. (1993) who showed that readily available (published) data on heats of combustion give accurate results when used in the energy balances for compost process modeling (See Eq. 4d):

$$Q_p = \frac{dM_{BVS\ solids}}{dt} * Q_{cs} + \frac{dM_{BVS\ oil}}{dt} * Q_{co} \qquad (4d)$$

Heat loss (Q_l, in MJ) is primarily made up of two components, the conductive and convective heat loss to the surroundings, and the removal of sensible heat and latent heat in the exhaust gases; represented by the two terms on the right side of Eq. 4e.

$$Q_l = UA_s * (T - T_o) + m_a * (h_2 - h_1) \qquad (4e)$$

where 'U' is the overall heat transfer coefficient ($W.m^{-2}.°C^{-1}$), 'A_s' is the surface area (m^2), and 'T_o' is ambient temperature (°C), 'm_a' is the aeration rate (kg air), and 'h' is the enthalpy ($kJ.kg^{-1}$) of the inlet air and exhaust gases.

In this research for simulation of the in-vessel composting process, the small quantity of materials used (1.5-2.5 kg) did not constitute a critical mass for retaining the biologically produced heat and hence sustaining the high temperature required during the high-rate phase of composting. For this purpose, the composting reactor had to be very well insulated. The Dewar flask with vacuum-induced insulation was first wrapped with a custom made Reflectix® (heating tank insulation, Reflectix Inc., Markville, IN) sleeve; and then placed inside an insulated box made out of R-5 Styrofoam board, where the space in between the reactor and the box was filled with R-28 roof insulation (Owens Corning, Toledo, OH).

The 'U' value, for the composting reactor and insulation box set up, was determined to be 0.028 ± 0.002 $W.m^{-2}.K^{-1}$ (See Appendix H). With a volume-to-surface area ratio of 1:275 $m^3.m^2$ for the reactor setup with insulation, its volumetric heat transfer coefficient would be 11.6 $W.m^{-3}.K^{-1}$; which is in line with a 'U' value, for a 1 m^3 composter, of 30 $kJ.h^{-1}.K^{-1}$ (or 8.3 $W.m^{-3}.K^{-1}$) as reported by Kaiser (1996).

At any given time Equation 5 represents the mass balance. The rate of change of the composting mass was attributed to three components, that of the 'dry' matter (solids and oil), and that of the water in the composting mixture (See Eq. 5a).

$$M_{total} = M_{water} + M_{solids} + M_{oil} \tag{5}$$

$$\frac{dM_{total}}{dt} = \frac{dM_s}{dt} + \frac{dM_w}{dt} + \frac{dM_o}{dt} \tag{5a}$$

where 'M_{total}' is the total composting mass (kg ww), 'M_s' is mass of solids (kg ds), 'M_w' is mass of water (kg water), and 'M_o' is mass of oil (kg oil). Furthermore:

$$M_s = s * M_{total} = (1 - mc) * M_{total} \tag{5b}$$

$$M_w = mc * M_{total} \tag{5c}$$

$$M_o = oil * M_s \tag{5d}$$

$$M_{vs} = vs * M_s \tag{5e}$$

$$M_{BVS} = \beta * M_{vs} \tag{5f}$$

where, 's' is solids content, in (decimal); 'mc' is moisture content, wet mass basis, in (decimal); 'oil' is the oil content, dry mass basis, in (decimal); 'vs' is volatile solids content, dry mass basis, in (decimal); 'β' is the biodegradability coefficient, as percent of volatile solids, in (decimal); 'M_{vs}' is mass of volatile solids, in (kg ds); and 'M_{BVS}' is mass of biodegradable volatile solids, in (kg ds).

Haug (1993) suggested the inclusion of different biodegradation rates for materials that have a fraction that decomposes slowly, and a fraction that decomposes quickly. The fast-degrading fraction is usually associated with the high-rate phase of the composting period whereas the slow fraction is associated with the curing phase. For modeling purposes in this thesis, only the first 168-hour period was of concern; hence the slow fraction (associated with the curing phase) was ignored as stated in the assumptions.

The biodegradable volatile solids were subdivided into two components, that of solids and that of oil. Hence, the changes in the compost 'dry' fraction included the changes in the non-water components, such as solids and oil. Accordingly, the changes in the solids and oil phase could be formulated as the change in biodegradable organic matter (See Eq. 6).

$$\frac{dM_s}{dt} = \frac{dM_{BVS}}{dt} = \frac{dM_{BVS\ solids}}{dt} + \frac{dM_{BVS\ oil}}{dt} \tag{6}$$

Biodegradation during composting is characterized by two major parameters: the biodegradation rate coefficient (k), and the substrate biodegradability (β). The biodegradability of the substrate is an important parameter, which determines the amount of energy available to drive the composting process. The biodegradability of solids and oil, and hence their biodegradation rate coefficients were treated as separate components in this model.

$$\frac{dM_{BVS}}{dt} = -k * M_{BVS} \quad (6a)$$

$$\frac{dM_{BVS\ solids}}{dt} = -k_{solids} * M_{BVS\ solids} \quad (6b)$$

$$\frac{dM_{BVS\ oil}}{dt} = -k_{oil} * M_{BVS\ oil} \quad (6c)$$

The following expression, which is an extension of the Arrhenius equation, may be used to describe the effects of temperature on the biodegradation rate coefficient (k):

$$k_m = k_{T1} * \left(C_1^{T-T1} - C_2^{T-T2}\right) \quad (7)$$

where 'T1' and 'T2' are reference temperatures, 'C_1' and 'C_2' are temperature coefficients, and 'k_{T1}' is the biodegradation rate coefficient at T1.

Based on the experimental data from co-composting of garbage and digested dewatered sludge (as reported by Haug 1993 using data from Schulze 1962), the reference temperatures are 20°C and 60°C, whereas the temperature coefficients are 1.066 and 1.21, respectively. Thus,

$$k_{m,\ solids} = k_{20,\ solids} * (1.066^{T-20} - 1.21^{T-60}) \quad (7a)$$

$$k_{m,\ oil} = k_{20,\ oil} * (1.066^{T-20} - 1.21^{T-60}) \quad (7b)$$

The subscript 'm' stands for the maximum possible biodegradation rate coefficient at the ongoing composting temperature. This equation corresponds to a Q_{10} value (the change in reaction rate for each 10°C rise in temperature) of 1.9 for the temperature range of 10 to 60°C. Beyond 70°C, inactivation of enzymes becomes rapid stopping the composting process.

The maximum possible biodegradation rate coefficient may now be corrected by multiplicative factors for the effects of moisture content (mc), and free air space (FAS) of the compost mixture, to arrive at a composite degradation coefficient:

$$k = k_m * F1 * F2 \tag{8}$$

According to Haug (1993), the rate modification factor F1 accounts for the effect of moisture content, mc (decimal), on the microbial activity:

$$F1 = \frac{1}{EXP(-a_1 * mc + a_2) + 1} \tag{8a}$$

while the modification factor F2 accounts for the effect of free air space, FAS, in (decimal) (Haug 1993):

$$F2 = \frac{1}{EXP(-a_3 * FAS + a_4) + 1} \tag{8b}$$

In turn, the physical effect of moisture content on FAS (inside the voids of the composting matrix) may be described by a linear equation fitted to the garbage and sludge data presented by Haug (1993):

$$FAS = \frac{(a_5 - mc)}{a_6} \tag{8c}$$

The values of the coefficients a_1 to a_6 are presented in Table 6.1.

The values of the modification factors F1 and F2 vary from 0 to 1, and are greater than 0.9 for moisture contents above 50%, and FAS greater than 22%. Hence, in this model, the effects of the changes in moisture content (mc) and free air space (FAS) are relatively small, when compared to the effect of temperature on the biodegradation rate coefficient. The temperature correction factor given by Equation 7

changes from a value of 1 at 20°C, to approximately 18 (one full order of magnitude higher) at 70°C, and drops back to 1 at 80°C.

Other methods used to estimate 'k' as a function of temperature, were the one proposed by Bari et al. (2000a), who found an empirical relationship for the in-vessel composting of a mixture of food waste and paper waste, as follows:

$$ln(k) = 17.73 - \frac{6688.5}{T + 273} \tag{9}$$

where 'k' is in (day^{-1}), and 'T' is in (°C);

and the theoretical relationship proposed by Nielsen and Berthelsen (2002) for the composting process rate and temperature, based on the enzyme activity as applied to the Arrhenius equation:

$$k = A * EXP(a * (T - T_o)) * \frac{1}{EXP(b * (T - T_1)) + 1} \tag{10}$$

where 'k' is in (day^{-1}), 'A' is 1.42 mg O_2. g vs^{-1}. hr^{-1}, 'a' is 0.065 $°C^{-1}$, 'b' is 0.45 $°C^{-1}$, 'T_o' is 40°C (an arbitrary T constant), and 'T_1' is 65°C (the 'optimum' biodegradation temperature for 'fatty' residues composting).

Furthermore, Stombaugh and Nokes (1996) based on an optimum composting temperature in the mesophilic range, proposed the following temperature factors influencing the biodegradation rate coefficient: in a multiplicative manner:

$$F_T = \frac{T}{30} \qquad 0°C < T < 30°C \tag{11a}$$

$$F_T = 1.0 \qquad 30°C < T \leq 55°C \tag{11b}$$

$$F_T = 3.75 - \frac{T}{20} \qquad 55°C < T \tag{11c}$$

where T corresponds to the composting mix temperature in °C, and F_T is the 'temperature correction' factor.

The relationship between the degradation rate coefficient and temperature, using the four methods (Eqs. 7, 9-11) are shown in Fig. 6.1. As the typical composting temperature profile follows more closely the pattern depicted by Eq. 7, the traditional method proposed by Haug (1993) was the one chosen to use in this simulation study.

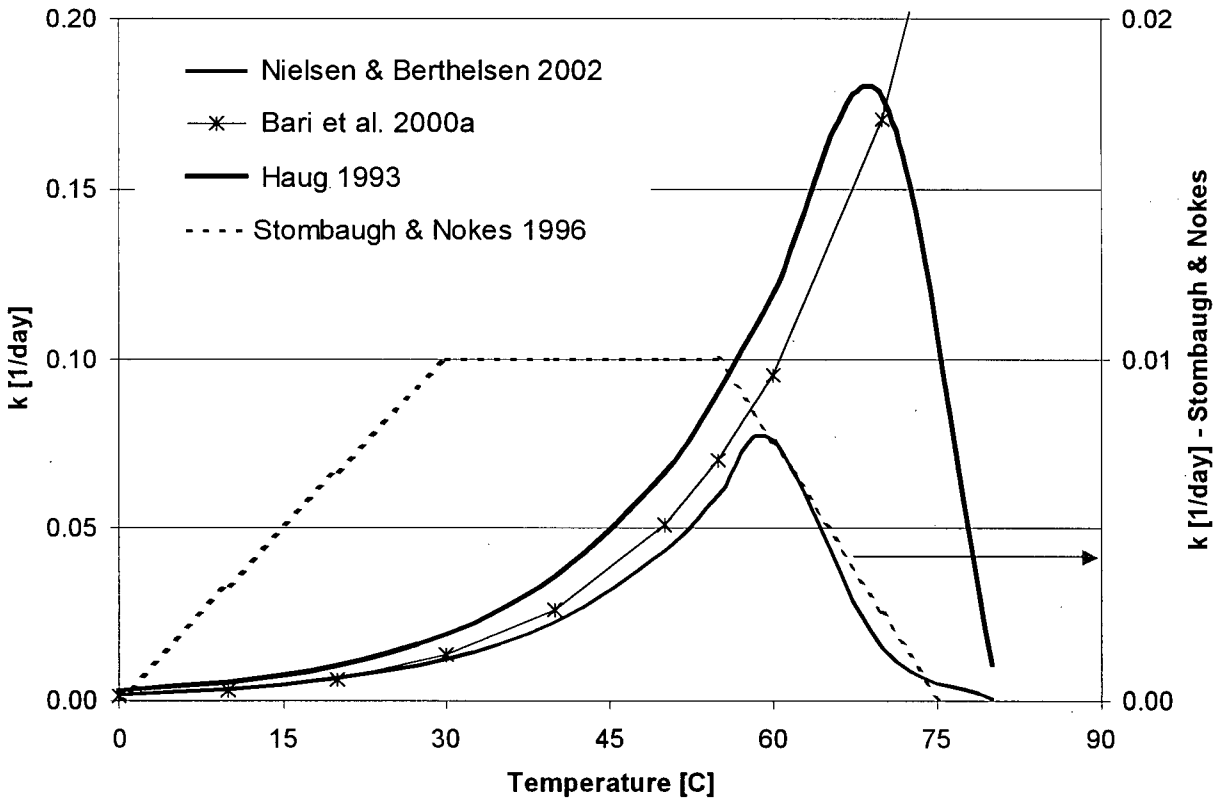

Figure 6.1 Effect of temperature on the biodegradation rate coefficient as calculated by various methods. Fine line: Nielsen and Berthelsen 2002 method. Line with asterisks: Bari et al. 2000a for food waste and sawdust. Thick line: Haug 1993, assuming k_{20} = 0.01 day^{-1}. Dotted line: Stombaugh and Nokes 1996, assuming k_{20} = 0.01 day^{-1}.

The change in the mass of water with time is represented by the difference between the water produced and the water lost during the composting process:

$$\frac{dM_w}{dt} = \frac{dM_{w,p}}{dt} - \frac{dM_{w,l}}{dt} \tag{12}$$

The amount of water produced may be obtained from stoichiometric considerations:

$$\frac{dM_{w,p}}{dt} = a_7 \frac{dM_{BVS\ solids}}{dt} + a_8 \frac{dM_{BVS\ oil}}{dt} \tag{12a}$$

where the coefficients 'a_7' and 'a_8' depend on the substrate composition ($C_aH_bO_cN_d$), and the stoichiometry of the biodegradation reaction since the substrate is oxidized mainly to carbon dioxide, water, and ammonia.

The amount of water lost is primarily due to the loss of water vapor in the exhaust gas stream and it may be calculated as

$$\frac{dM_{w,l}}{dt} = m_a(W_2 - W_1) \tag{12b}$$

where 'W_1' and 'W_2' are the humidity ratios (moisture contents, in kg water. kg air^{-1}) of the inlet and outlet air streams, and 'm_a' is the aeration rate (in kg air).

Exhaust air is assumed to have a temperature equal to the compost temperature, and have a saturated humidity, as was the case for the experimental treatments presented in Chapter 3 (pp. 38). This assumption followed Stombaugh and Nokes (1996), who modeled a composting vessel with stratified layers; where air leaving a layer was assumed to be at the dry bulb temperature of the layer, and air was assumed saturated at that temperature, if the moisture content of the layer was greater than 18% ww. On the other hand, Bach et al. (1987) has shown that air leaving compost piles is saturated when the moisture content of the composting mix was greater than 50%, and was at 95% relative humidity when the moisture content fell between 45 and 50%.

The model in this study had provision for a temperature control strategy using different aeration rates, based on the Rutger's strategy, in which aeration rate is at baseline level (25 CFM or 0.72 lpm.kg dry matter^{-1}) and intermittent (on 33% duty cycle) below the temperature setpoint, 'T_{set}'. In contrast, the aeration rate is continuous (on 100% duty cycle) above the temperature setpoint.

Aeration rate, in ($kg.s^{-1}$, as converted from lpm) is given by:

$$m_{a,\ intermittent} = M_s * 0.72 * \rho_{air} * 0.33 \qquad T < T_{set} \qquad (13a)$$

$$m_{a,\ continuous} = M_s * 0.72 * \rho_{air} * 1 \qquad T > T_{set} \qquad (13b)$$

6.3.3 Model Equations Solution

In solving the set of ordinary differential equations, a forward-marching finite difference strategy was used; the time step selected was small enough (3 minutes) to reflect the changes in the aeration rates during composting (the actual aeration rate cycle used in the experimental section was 3 minutes, in which the aeration pumps were 'on' for 1 min, and 'off' for 2 min, for the intermittent part of the cycle). Simulation was performed for the first 168 hours of the active phase of composting.

Starting with a known initial temperature of the compost mixture (set equal to ambient temperature), the model proceeds to calculate heat accumulation rate, based on the biodegradation rate, and the rate of change of water content. Then, the process temperature at time (t+dt) is estimated form Eq. (1b) knowing the change of energy content during that particular time step. Afterwards, the simulation process marches onto the next time interval. A psychrometrics subroutine (see Appendix J) was included in the model for computing the values of enthalpy and humidity ratio.

The mathematical solution of the compost model equations has been achieved by using custom made software (Person and Shayya 1994, Haug 1993), or a commercial modeling software package such as STELLA® (Ndegwa et al. 2000, Mohee at al. 1998).

The solution of the model equations, in this study, was carried out using an Excel 2000® spreadsheet (Microsoft, Redmond, WA). The spreadsheet (which was about 6 MB in size) was set up to allow for the manual input of the initial composting conditions and substrate characteristics. The model solution consists of 3360 time steps (equivalent to 3-min time steps for a total of 168 hours simulated), which running on a PC (with a Pentium III processor at 550 MHz, and 128 MB RAM), took a couple of

seconds to solve. Tables 6.1 and 6.2 show a summary of the model inputs and outputs, respectively.

6.3.4 Model Simulation Input Values

Model inputs were selected, based on values reported in the literature, as well as values derived from the experiments in this study (Chapter 3). Different scenarios are presented for food waste and yard trimmings as the main composting substrates. The effect of incorporating lipid-rich wastes into the composting mix is presented in detail, particularly in regard to the biodegradation kinetics, and composting thermal processes.

Total initial mass corresponds to the average wet mass (1.5 - 2.5 kg) held in a metallic 6-L Dewar flask (consisting of two stainless steel walls, that produce a thermos-like effect). Initial moisture content was set up to be either at the optimal value, which is 55% (which results in a free air space of 30%), or at the particular value for the particular treatment being simulated.

The theoretical biodegradability of any organic substrate is 100%, since organic matter would ultimately decompose to carbon dioxide and water (Golueke 1977). However, during the composting process, the biodegradability of organic wastes is often less than 100%, due to time constraints, or to the presence of compounds resistant to microbial decay. Haug (1993) estimated values of degradability to be 45% for refuse, and 86% for lipids. Viel et al. (1987a,b) and Fernandes et al. (1988) also found that biodegradability of lipid-rich wastes was 85%. For the model validation the biodegradabilities of the different experimental treatments were used for the simulation of the respective treatment.

The heats of combustion (for food waste, yard waste, and 'fats as foodstuff wastes' for grease trap sludge) were as reported by Tchobanoglous et al. (1993) (See Chp. 3, pp. 52). The reported heat of combustion of canola oil (38.5 $MJ.kg^{-1}$, Munchen 1989) is very similar to the value reported by Tchobanoglous et al. (1993) for 'fats' (with a value of 38.3 $MJ.kg^{-1}$). The molecular formulae and weights (for food waste, yard waste, and 'fat and oil' for grease trap sludge) were based on data reported by Haug et al. (1993), and for canola oil as reported by the Canola Council of Canada (2000). Again, the elemental composition of 'fat and oil' ($C_{50}H_{90}O_6$), and the one for canola oil ($C_{18}H_{33}O_2$) were approximately the same.

Table 6.1. Composting model inputs.

Parameter	Symbol	Values & Units**
Temperature ambient *	T_o	23 ± 2 °C
Total initial composting mass *	$M_{t=0}$	1.5 - 2.5 kg ww
Initial moisture content	$\%mc_{t=0}$	% ww
Initial proportion of oil *	$\%oil_{t=0}$	% ds
Biodegradability of solids *	β_{solids}	% ds
Biodegradability of oil *	β_{oil}	% oil
Heat combustion of solids §	Q_{cs}	Food waste: 13.9 $MJ.kg^{-1}$ Yard trimmings: 15.1 $MJ.kg^{-1}$
Heat combustion of oil §	Q_{co}	38.3 $MJ.kg^{-1}$
Overall heat transfer coefficient *	U	0.028 ± 0.002 W m^{-2} $°C^{-1}$
Surface area for heat conduction *	A_s	1.65 m^2
Specific heat for solids (compost) †	C_{ps}	0.65 kJ kg^{-1} $°C^{-1}$
Specific heat for water †	C_{pw}	4.18 kJ kg^{-1} $°C^{-1}$
Specific heat for oil ‡	C_{po}	1.90 kJ kg^{-1} $°C^{-1}$
Biodegradation rate coefficient at T=20°C, solids	$k_{20,\ solids}$	day^{-1}
Biodegradation rate coefficient at T=20°C, oil	$k_{20,\ oil}$	day^{-1}
Temperature set point	T_{set}	65 °C
Relative humidity air inlet *	$\%RH_{in}$	25 %
Relative humidity exhaust gases *	$\%RH_{out}$	100 %
Elemental composition substrate	$C_aH_bO_cN_d$	Food waste§: $C_{18}H_{26}O_{10}N_1$ Yard trimmings§: $C_{27}H_{38}O_{16}N_1$ Oil‡: $C_{18}H_{33}O_2$
Molecular weight substrate	MW	Food waste: 416 Mol.g mol^{-1} Yard trimmings: 632 Mol.g mol^{-1} Oil: 281 Mol.g mol^{-1}
Time step	dt	1/480 day (3 min)
Coefficients†	a_1	17.684
	a_2	7.0622
	a_3	23.675
	a_4	3.4945
	a_5	1.0375
	a_6	1.525
	a_7	Food waste: 0.50 Yard trimmings: 0.50
	a_8	1.06
	C_1	1.066
	C_2	1.21

Sources: * from experimental data, ** no value indicates a variable input parameter, † Haug 1993, ‡ Canola Council of Canada 2000, § Tchobanoglous et al. 1993

Table 6.2. Composting model outputs.

Parameter	Symbol	Units
Temperature composting mix	T	°C
Total mass degraded	$M_{biodegraded}$	kg BVS
Mass reduction (as % of total initial mass)	$\%M_{biodegraded}$	%
Peak temperature	T_p	°C
Time to reach the peak temperature	t_p	day
Final moisture content	mc_{final}	%
Area under the temperature curve	A_{curve}	°C.hr

The overall heat transfer coefficients, and the area for heat transfer by convection, were as calculated from the calorimetry experimental data (See Appendix H for more detail). Specific heat values for compost solids, and water were as reported by Haug (1993); and for canola oil as reported by the Canola Council of Canada (2000).

The relative humidity of the inlet air and ambient temperature correspond to the conditions in the lab where the composting reactors were set up. Relative humidity in the exhaust gases was measured to be at saturation (See Chapter 3, pp. 38); this coincides with the value suggested by Bach et al. (1987) who stated that the exhaust gases would be at 95-100% saturation whenever the composting mix had a moisture content above 45%.

Among the initial objectives of this research was to study the biodegradation of the composting substrate through time by measuring the carbon dioxide evolution (as an indicator of aerobic and anaerobic metabolisms), and the oxygen concentration (as a measure of aerobic microbial activity). The measurement of carbon dioxide from daily samples for different treatments could not be accomplished due to analytical constraints (gas chromatograph suffered an irrecoverable failure). In addition, the oxygen probe and sensor used (Model 1630, Engineered Systems and Designs Inc., Newark, DE) for the oxygen concentration measurement gave variable results under saturated air conditions; hence the oxygen concentration values (as shown in Figs. 3.13 and 3.15, pp. 67 and 69) were used only as indicators of aerobic metabolism rather than a reliable measure for the estimation of composting mass biodegradation.

Various methods were used to derive the value of k_{20}, the biodegradation rate coefficient at a reference temperature of 20°C, to be used as inputs to the model.

Firstly, the published 'k' values for the different substrate components (0.01 day^{-1} for food waste reported by Keener et al. 1997; 0.0232 day^{-1} for yard waste assumed as Bermuda grass reported by Haug 1993; and 0.015 day^{-1} for oil as 50% larger than the value reported for mineral oil by Haug 1993) were used as input values. A limited degree of success was achieved in terms of prediction accuracy of temperature profiles and changes of mass as shown in Table 6.3 and Figure 6.2.

Next, attempts were made to estimate k_{20} using the overall biodegradation rate coefficient determined from the empirical data of overall mass reduction during the high-rate phase of the composting in Chapter 3 (See Table 3.19, pp. 79). The challenge was to find an "equivalent temperature" to which the overall biodegradation rate coefficient corresponded. For this end, different statistical representations of the actual temperature profile - arithmetic mean, geometric mean, median and mode, were used as inputs to the computer model. Again, as shown in Table 6.3 and Fig 6.2, the simulated results had relatively low correlation values (R^2) with measured results.

The term 'R^2' is the regression coefficient of a linear equation fitted to the modeled versus the measured temperature data. In addition, '% mass error' corresponds to the difference between the modeled and the measured percentage mass change (as % dry basis of initial mass).

Due to the limited success with the previous two methods, eventually, a trial-and-error procedure was adopted to determine the value of 'k_{20}' using a correction factor, α, on the empirical k-value, with an aim to achieve a correlation coefficient (R^2) as large as possible for the simulated versus measured temperature profile, while maintaining the percentage mass error between the predicted and actual mass changes at a minimum. Thus,

$$k_{20} = \alpha * k_{empirical} \qquad (14)$$

The 'k' values obtained using the trial and error procedure (Table 6.4) with the 'α' factor are in the range of 0.007-0.072 day^{-1} for solids, and 0.006-0.303 day^{-1} for lipids. These values were in the same range for the 'k' values summarized by Haug (1993) and Keener et al. (1996), ranging from 0.0015 to 0.0699 day^{-1}, and 0.012 to 0.111 day^{-1}, respectively. In addition, the model developed by Keener et al. (1997) produced 'k'

values varying from 0.022 to 0.259 day^{-1}. Furthermore, the chosen 'α' values, yielded a temperature (using Eq. 7a,b) that was well correlated to the biodegradation rate coefficient following the Arrhenius equation (See Appendix I).

Table 6.3. Calculation of 'k_{20}' using 'k' values from the literature, and based on the Haug's method (Eq. 7a,b) for different temperatures. Treatment YG1.

Method	Temperature factor in Eq.7	$k_{20,\ solids}$	$k_{20,\ oil}$	R^2	% mass error
Using 'k' as reported in the literature, with $k_{yard\ trimmings}$=0.023 day^{-1}, k_{oil}=0.015 day^{-1}					
		0.023	0.015	0.09	27
Using Eq. 7a and 7b, with empirical k_{solids}=0.029 day^{-1} and k_{oil}=0.101 day^{-1}					
Minimum T (20.0°C)	1.00	0.029	0.101	0.24	19
Geo. Mean T (41.8°C)	0.25	0.007	0.025	0.42	91
Average T (44.1°C)	0.22	0.006	0.022	0.43	93
Median T (46.4°C)	0.19	0.006	0.019	0.43	94
Mode T (55.2°C)	0.11	0.003	0.011	0.45	97
Maximum T (61.5°C)	0.08	0.002	0.008	0.52	98

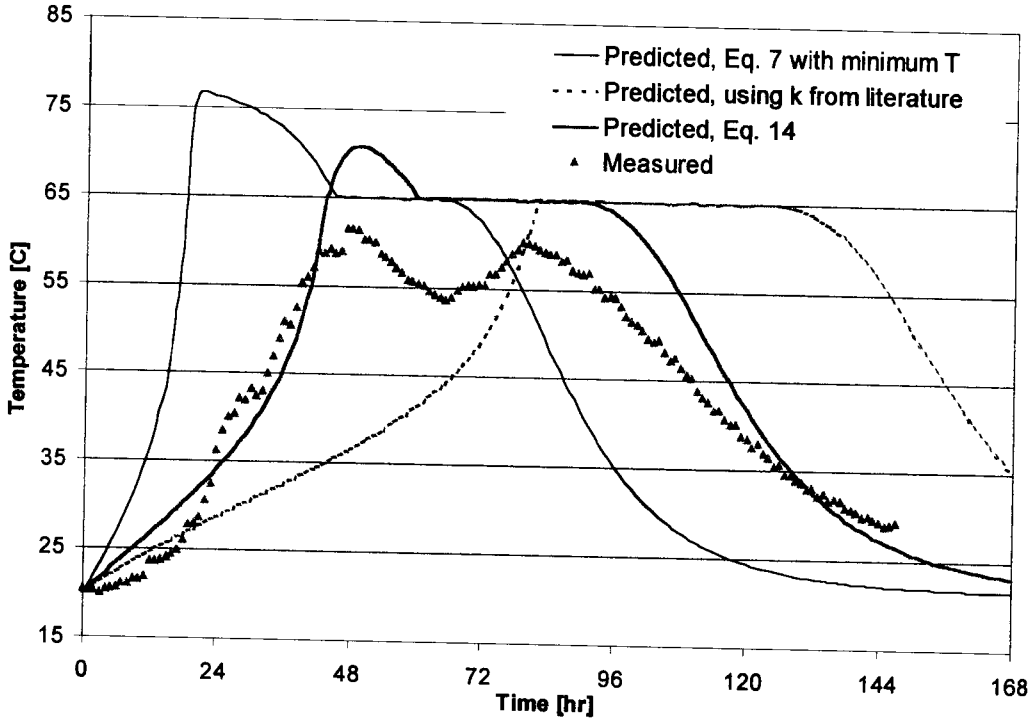

Figure 6.2 Empirical and modeled temperature profiles for treatment YG1, yard trimmings and grease trap sludge (at 5% ds). Solid line: Modeled profile using Eq. 7a,b to calculate 'k'; dotted line: Modeled profile using 'k' values from the literature; Thick solid line: Modeled profile using Eq. 14 according to this study; Triangles: Empirical temperature profile.

Using the above-mentioned procedure, the model produced acceptable results for the prediction of temperature profiles and mass changes as demonstrated in the next section, despite the limitations of the model kinetics of being described only by the biodegradation rate coefficients.

Table 6.4 Correction factors used to estimate k_{20} as a function of the empirical biodegradation coefficient. 'R^2' is the regression coefficient of a linear equation fitted to the modeled versus the measured temperature data. 'k' values are in (day^{-1}).

Treatment	For solids		For oil		R^2	% mass
	α	Estimated 'k_{20}'	α	Estimated 'k_{20}'		error
YC1	0.10	0.013	0.06	0.008	0.75	87.0
YC2	0.50	0.050	0.06	0.006	0.94	34.8
YC3	0.25	0.031	0.05	0.007	0.76	17.6
Control	0.06	0.012	1.00	0.205	0.74	75.0
FC1	1.00	0.026	1.00	0.012	0.37	31.8
FC2	1.00	0.007	0.36	0.030	0.96	41.7
FC3	1.00	0.012	0.59	0.038	0.73	46.7
FC4	2.50	0.031	0.22	0.023	0.49	33.3
Control1	1.40	0.033	0.33	0.031	0.02	24.3
FC5	1.00	0.010	0.21	0.027	0.52	45.1
Control2	1.10	0.029	1.00	0.049	0.58	37.5
FC6	1.00	0.013	0.20	0.024	0.32	38.9
FC7	1.00	0.018	0.08	0.013	0.27	34.8
YG1	0.67	0.020	0.50	0.051	0.94	27.5
YG2	2.50	0.023	0.56	0.039	0.33	12.1
Control	1.67	0.024	0.67	0.303	0.57	62.7
FG1	5.00	0.041	1.00	0.045	0.22	14.7
FG2	6.67	0.072	1.00	0.015	0.03	16.7
Control	2.50	0.044	1.00	0.049	0.20	8.3

6.4 RESULTS AND DISCUSSION

The following sections present the various results for the model simulation. An example of model calculations can be found in Appendix J.

6.4.1 Effect on Temperature Profiles

The effect of adding lipid-rich wastes to the composting mixes on the temperature profile is demonstrated in Figures 6.3 to 6.7, for the selected experimental

treatments with either food wastes or yard trimmings, and canola oil or grease trap sludge. Figures 6.5 and 6.6 show that the model underestimated the area under the temperature profiles for the treatments with food wastes as main substrate. However, the modeled temperature profiles followed very well the trends in the actual profiles for yard trimmings (Figures 6.3, 6.4, and 6.7).

The model also overestimates the peak temperatures for the treatments with yard trimmings, and either canola oil or grease trap sludge (Figs. 6.3 and 6.7), this difference might be due to the estimated values of the factor 'α'.

The difference in the temperature peak and the time to reach it is presented in more detail in Figures 6.8 to 6.11. Figure 6.8 shows that the temperature peaks for treatments in experimental sets #1 and #2, were within or very near to the 10% error range of the actual temperature peaks. For Set #1, a 'zig zag' pattern was observed with increasing oil content, however treatment YC1 (35% ds) had a relatively low initial moisture content (40% ww) that resulted in a temperature profile performing only in the mesophilic region (below 50°C); hence, the relatively low temperature peak (also reflected in the model temperature peak). For Sets # 2a and #2b, there was not much difference in the temperature peaks corresponding to an increase in oil concentration from 3 to 10% ds. The model also reflected the same trend.

Figure 6.9, presents the modeled and actual temperature peaks for experimental sets 3 and 4. The model predicted similar values for the peak temperature with increasing oil content for the yard trimmings treatments and the food waste treatments (except for treatment FG2), as compared to the actual peak temperature values. The modeled and actual temperature peaks are within or very near to the 10% error brackets of the actual measurement.

Results pertinent to another response variable, the time to reach the peak temperature (t_p) are depicted in Figures 6.10 and 6.11 for experimental sets #1 to #4. For set #1, except for treatment YC2, the predicted and the actual 't_p' values were different, though their values were within 1 day (24 hours) or less. Simulated results were more accurate for sets #2a and #2b, in that they were within or close to the 10% error brackets of the actual values.

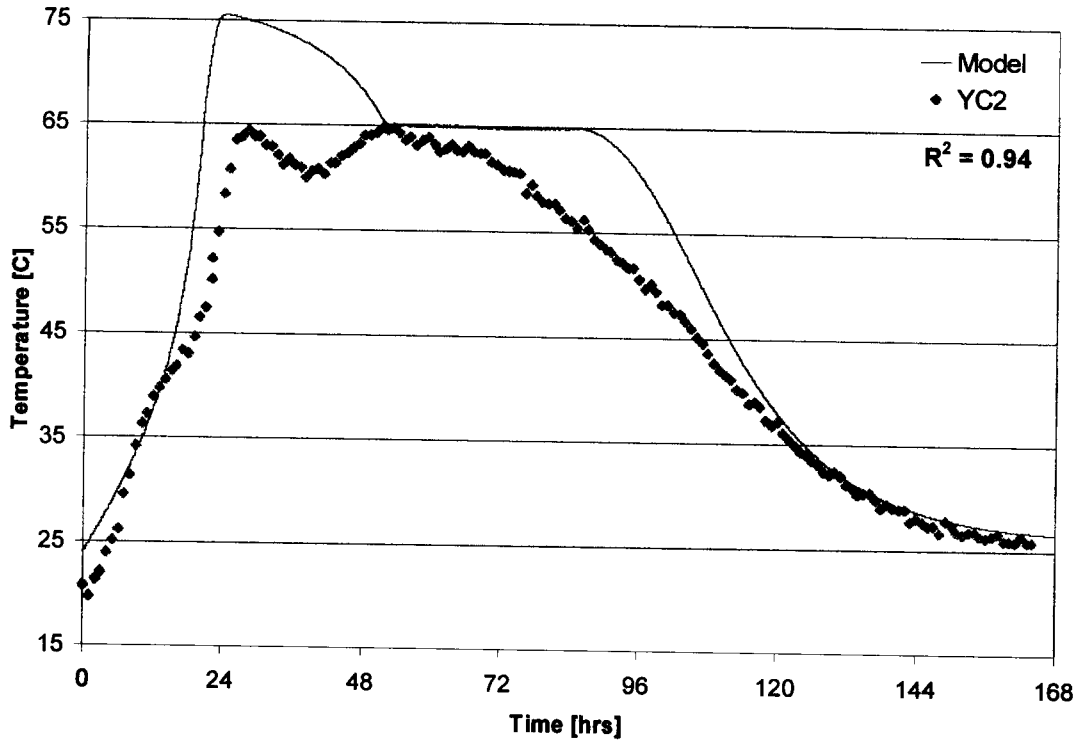

Figure 6.3 Modeled and experimental temperature profiles for yard trimmings and canola oil (35% ds) treatment YC2. 'R^2' is the regression coefficient of a linear equation fitted between the modeled versus measured temperature data.

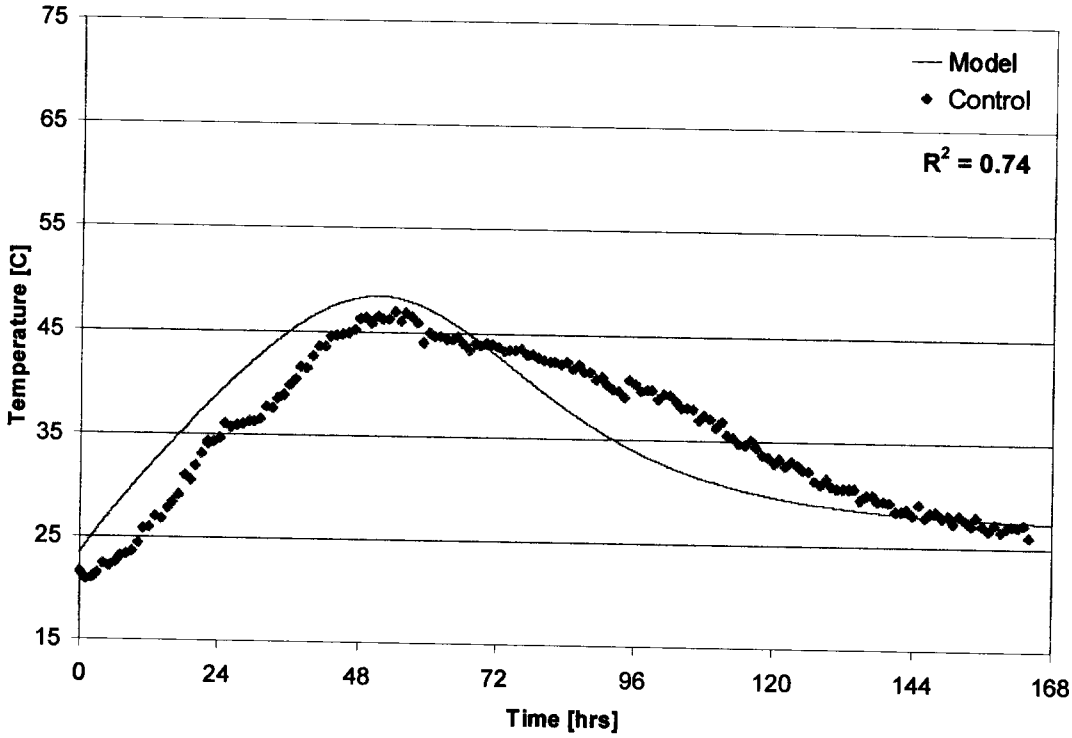

Figure 6.4 Modeled and experimental temperature profiles for the yard trimmings treatment 'Control' from Set # 1. 'R^2' is the regression coefficient of a linear equation fitted between the modeled versus measured temperature data.

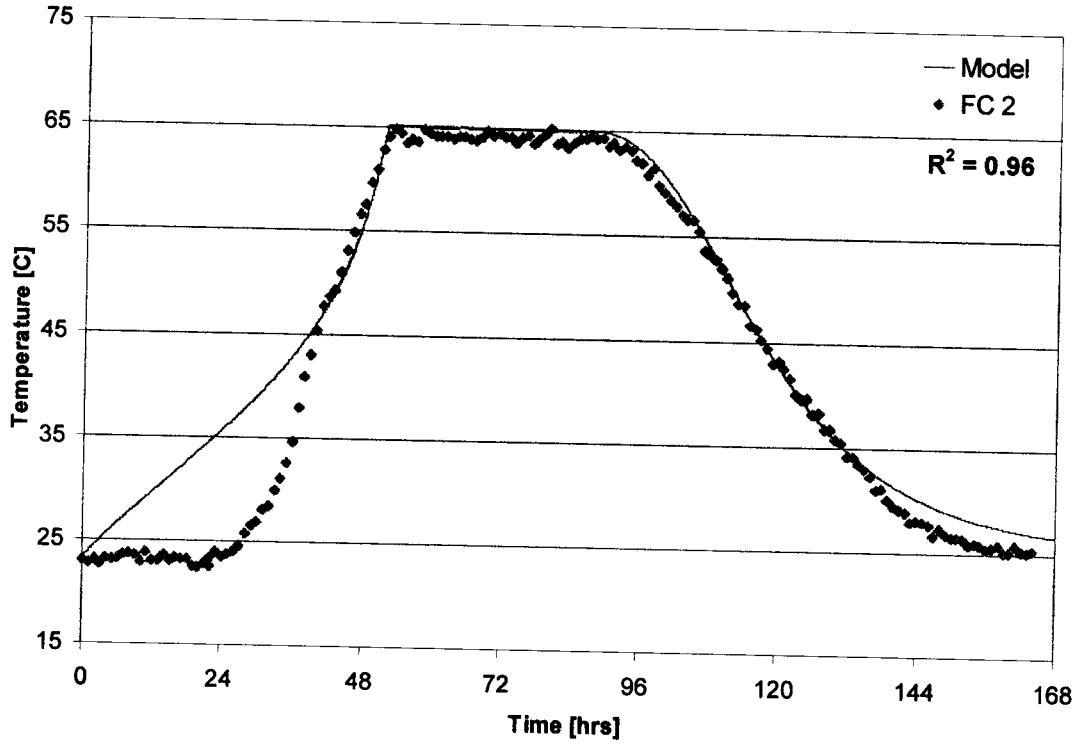

Figure 6.5 Modeled and experimental temperature profiles for food waste and canola oil (10% ds) treatment FC2. 'R^2' is the regression coefficient of a linear equation fitted between the modeled versus measured temperature data.

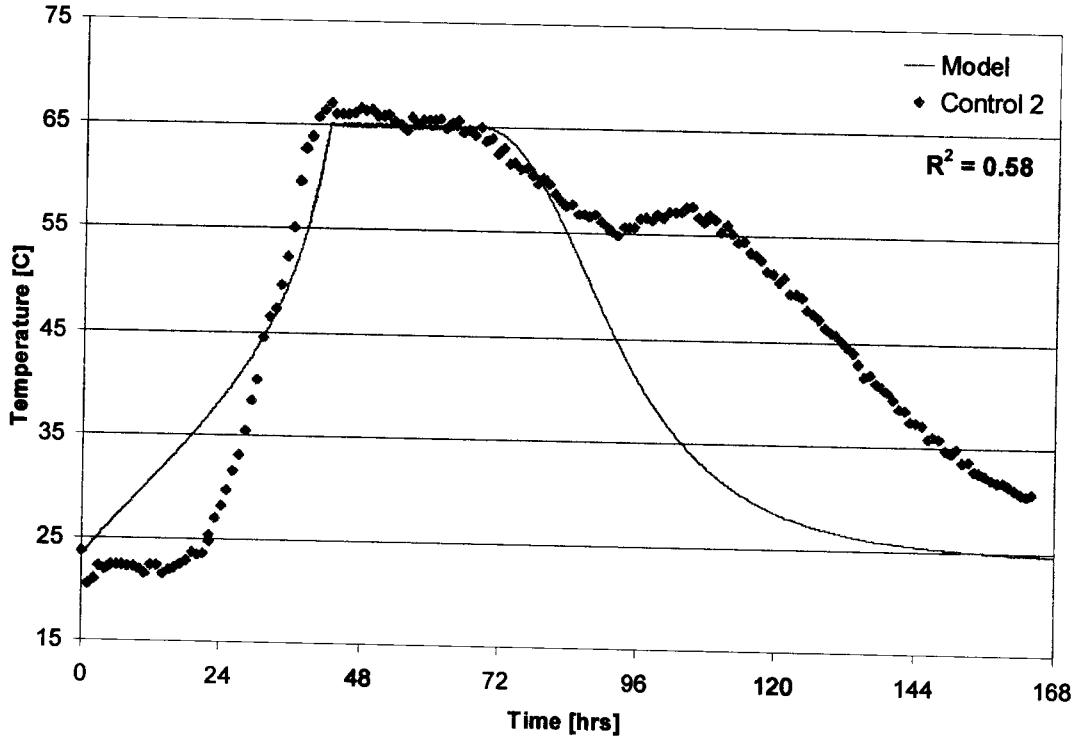

Figure 6.6 Modeled and experimental temperature profiles for food waste treatment Control 2. 'R^2' is the regression coefficient of a linear equation fitted between the modeled versus measured temperature data.

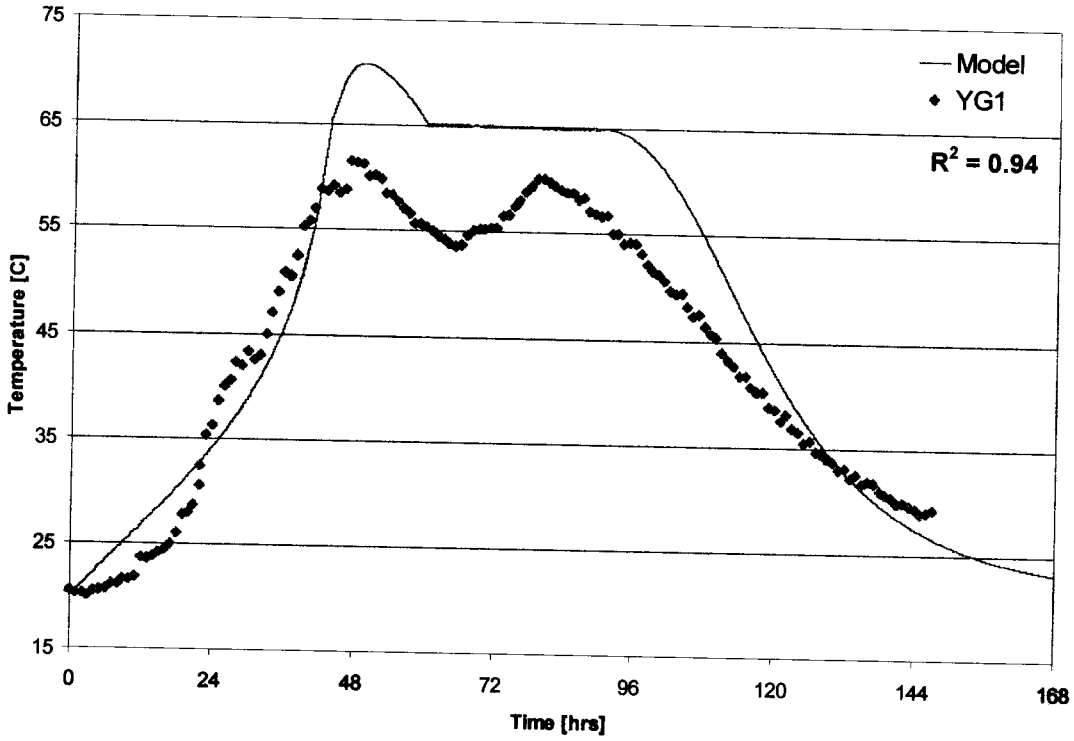

Figure 6.7 Modeled and experimental temperature profiles for yard trimmings and grease trap sludge (5% ds) treatment YG1. 'R^2' is the regression coefficient of a linear equation fitted between the modeled versus measured temperature data.

For set #3 (yard trimmings and grease trap sludge treatments), the modeled values were reasonably close to the empirical ones. However, model predictions for set #4 (food wastes and grease trap sludge treatments), except for the Control treatment, had a large discrepancy with actual values. Apparently, the model could not adequately predict the unusual thermal performance exhibited by treatments FG1 and FG2 (with 5 and 10% ds grease trap sludge respectively). For sets #3 and #4, the actual time to reach peak temperature increased with increasing lipids content from grease trap sludge; this might be explained by the lower biodegradability of grease trap sludge when compared with canola oil.

6.4.2 Effect on Total Mass Biodegradation

A comparison between the simulated and the actual values for mass changes (as % of the initial mass ds) is demonstrated in Figures 6.12 and 6.13. In general, the predicted values were smaller than the actual values in terms of mass reduction (except

for set #4). For set #1 (Fig. 6.12), the modeled and actual values were quite different, and this was probably due to the relatively small 'α' values used in this experimental set. However, for sets #2a and #2b, the modeled values for mass changes followed the same trend as the actual values. For set #3 (Fig. 6.13), the predicted values followed the trend of the actual mass changes quite well. In the case of treatments with food waste and grease trap sludge added (set #4), the model was able to predict the mass changes within 10% error of the measured mass changes.

6.4.3 Final Moisture Content

For the predictions of the final moisture contents (Figures 6.14-6.15) the simulated trend was similar to the trend of the actual values (set #2) as a function of lipid content when canola oil is added to food waste. However, the model consistently overestimated the values to different extents for sets #2a and #2b, while underestimating the final 'mc' values for set #1.

Figure 6.15 shows the modeled and actual values for the final moisture contents for treatments in experimental sets #3 and #4. In both scenarios, the predicted values were within or very close to the 10% error bracket, suggesting that the model could accurately predict the final moisture content for yard trimmings and food waste composting with grease trap sludge added.

6.4.4 Predicted Effect of Oil Concentration

Figures 6.16 to 6.18 show the effect of changing the initial oil concentration (% ds) on the peak temperature, time to reach the peak temperature, and mass degradation; as well as the area under the temperature curve and final moisture content. As expected due to the increased amount of readily available energy, the peak temperature shows an increase in value with increasing oil concentration. Accordingly, the model predicts shorter values for the time to reach the peak temperature as initial oil concentration increases.

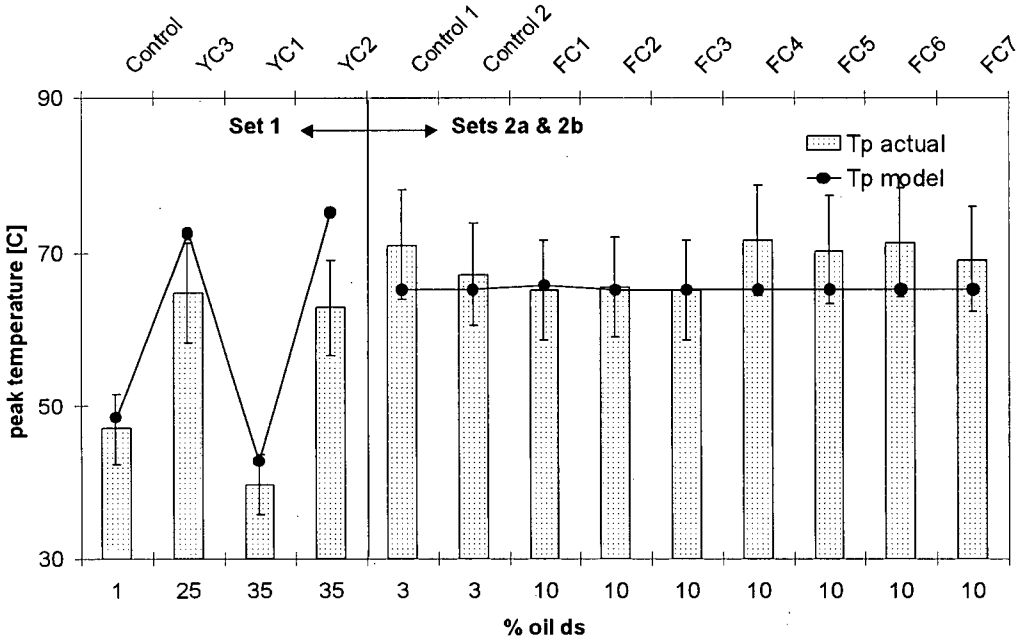

Figure 6.8 Actual and modeled values for the temperature peak for experimental sets 1 and 2. Error bars represent 10% of the actual measurement.

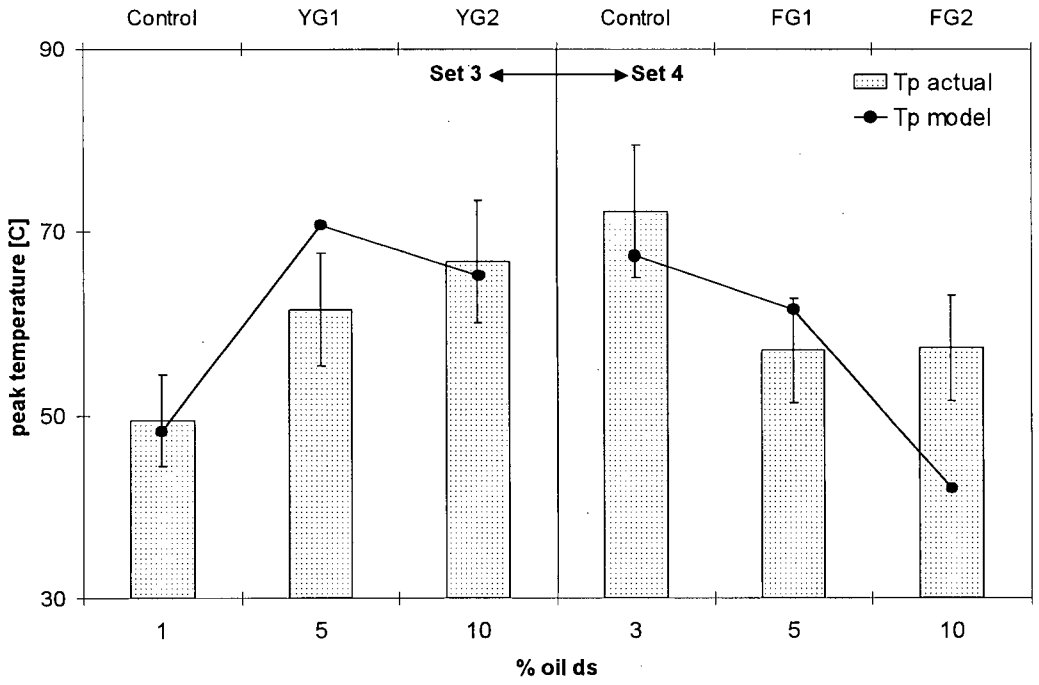

Figure 6.9 Actual and modeled values for the temperature peak for experimental sets 3 and 4. Error bars represent 10% of the actual measurement.

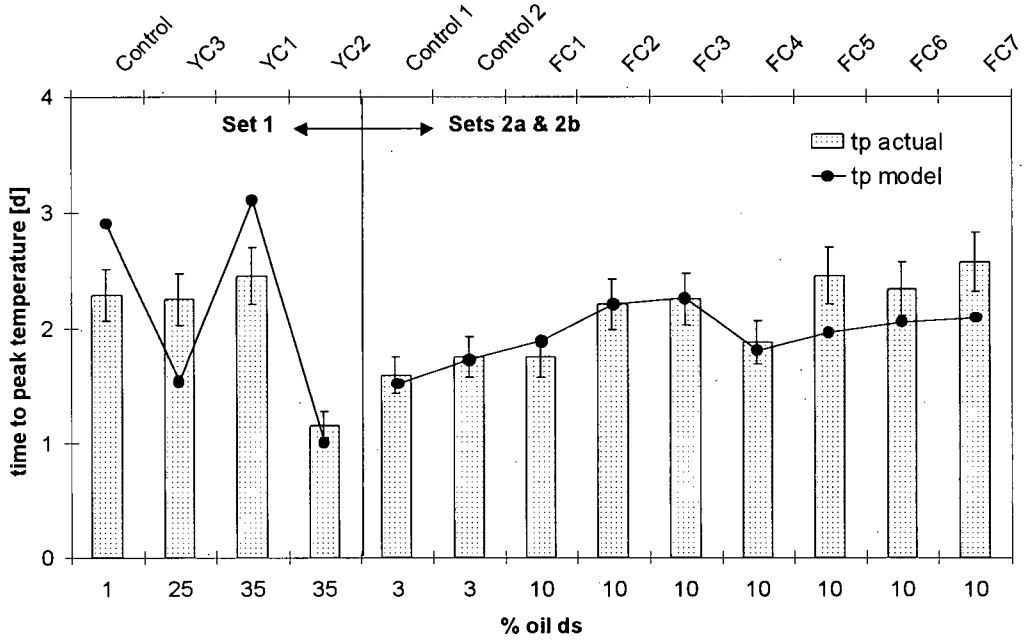

Figure 6.10 Actual and modeled values for the time to reach temperature peak for experimental sets 1 and 2. Error bars represent 10% of the actual measurement.

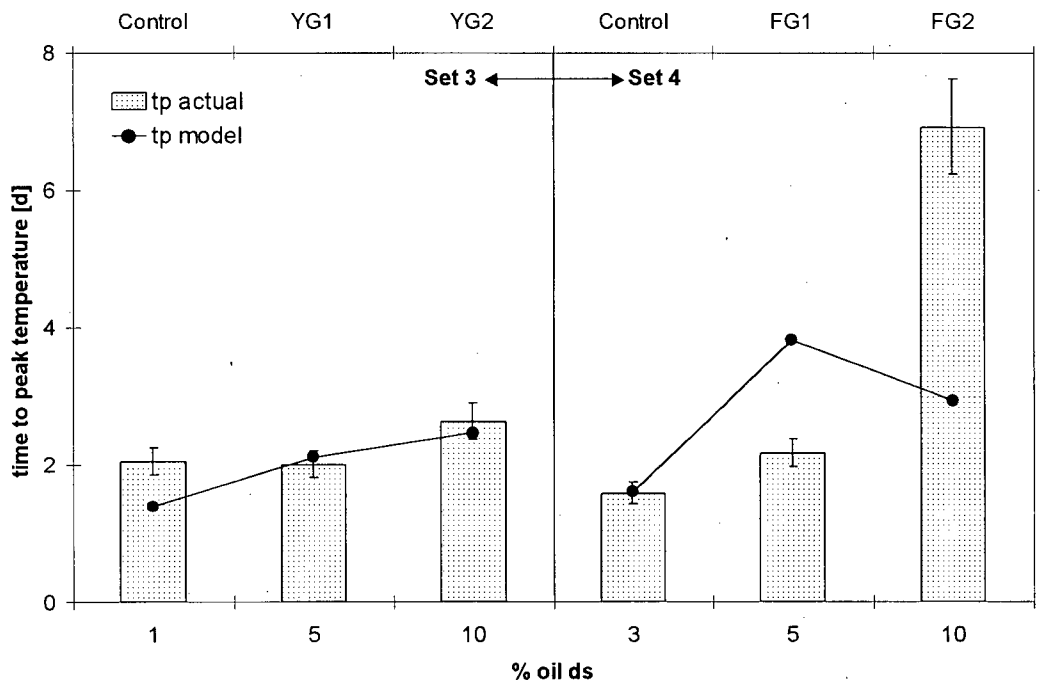

Figure 6.11 Actual and modeled values for the time to reach temperature peak for experimental sets 3 and 4. Error bars represent 10% of the actual measurement.

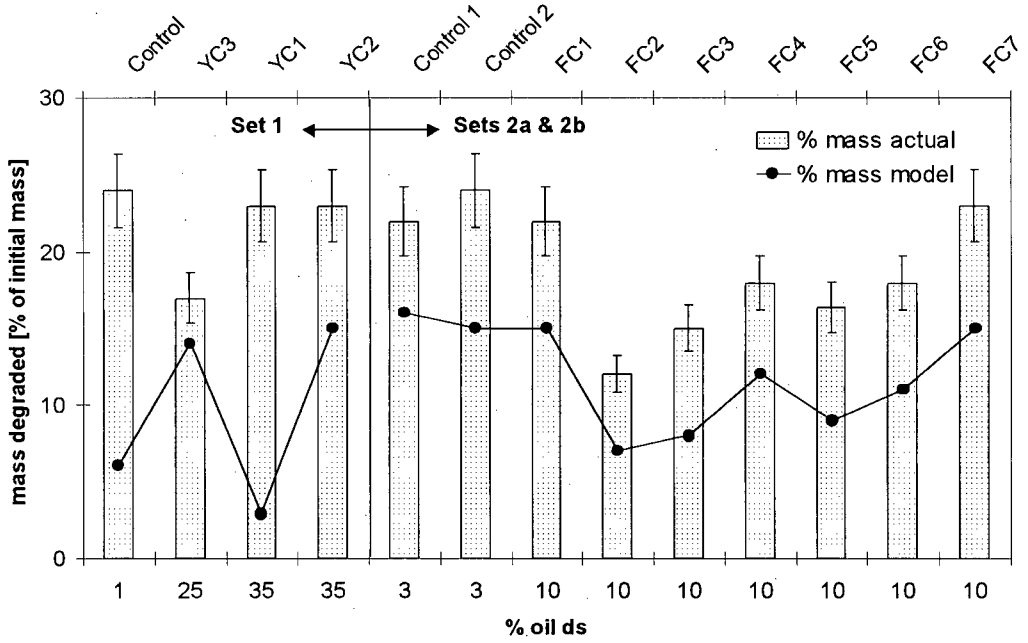

Figure 6.12 Actual and modeled values for the mass biodegraded (as % of the mass initial) for experimental sets 1 and 2. Error bars represent 10% of the actual measurement.

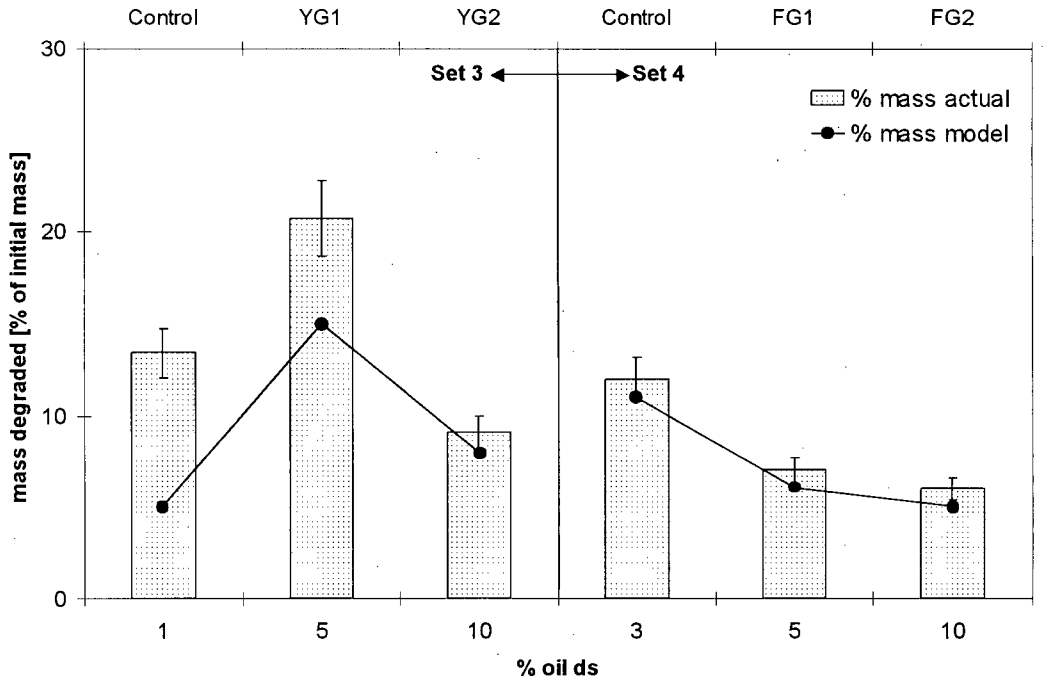

Figure 6.13 Actual and modeled values for the mass biodegraded (as % of the mass initial) for experimental sets 3 and 4. Error bars represent 10% of the actual measurement.

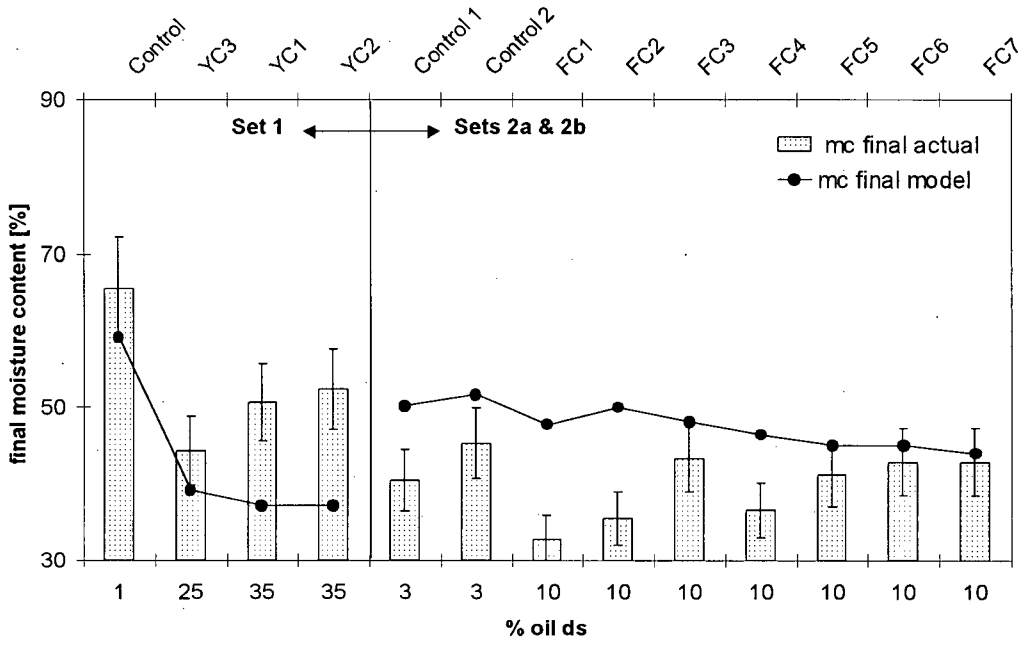

Figure 6.14 Actual and modeled values for the final moisture content (% ww) for experimental sets 1 and 2. Error bars represent 10% of the actual measurement.

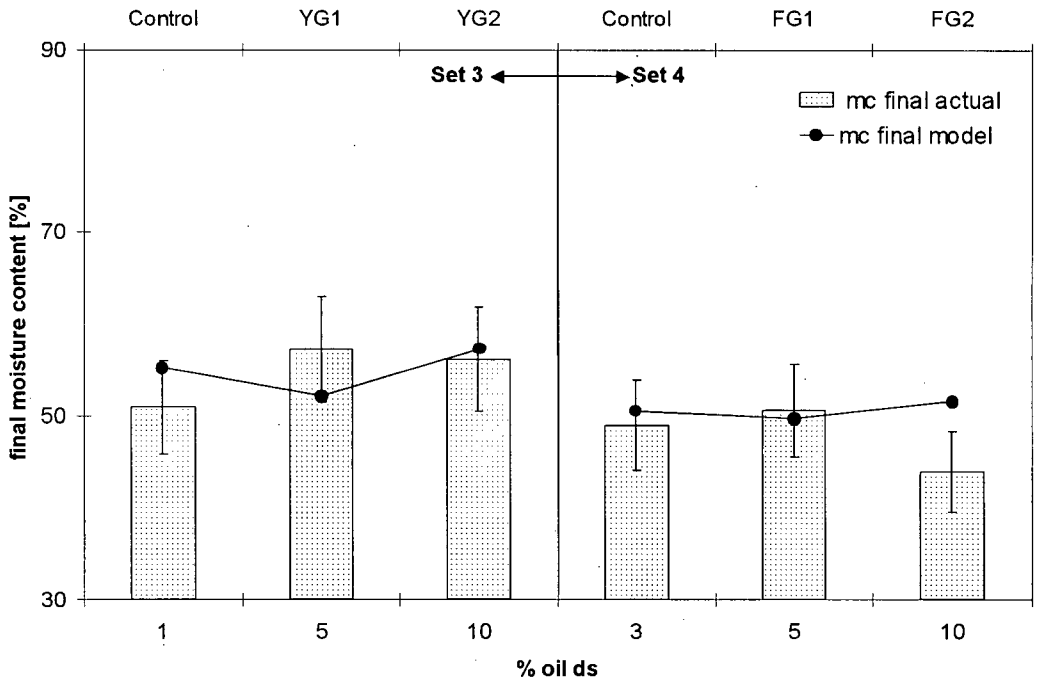

Figure 6.15 Actual and modeled values for the final moisture content (% ww) for experimental sets 3 and 4. Error bars represent 10% of the actual measurement.

The empirical data largely supported the trend for the peak temperature for increasing oil concentrations. However, the predictions of 'time to reach the peak temperature' had an opposite trend when compared with the empirical data (Figures 6.10 and 6.11), with the predicted values having a sharp decrease with increasing oil concentration from 0% to 10% ds. The discrepancies between the simulated and the actual 'time to reach peak temperature' might be explained by the fact that the model is a 'macrokinetic' model, where the rate of mass change (thus the microbial activity) is represented only by the biodegradation rate coefficients. Thus, the transient effect due to an extra 'liquid phase' (in this case oil) into the composting matrix could not be adequately modeled. With an increasing amount of oil in the compost matrix pores, it is possible that the free air space could shrink to the point that imparts a detrimental effect on the biofilm surrounding the compost particles (by either covering the 'liquid film', or by impeding the flow of air through the pores), according to Haug (1993). The effects of the presence of an 'extra liquid phase' would ultimately result in longer times for the compost temperature to reach its maximum value. In addition, larger amounts of a mixture of weathered fats, oil, and grease might also demand longer 'acclimation/lag times" from the microbes.

From Fig 6.17 it can be seen that the largest impact on mass degraded is for the initial oil concentration to change from 0 to 10% ds. Following the model, increase in the initial oil concentration up to 10% ds would enhance the mass degradation. Lipids addition above 10% ds would not be recommended due to the decrease in the extent of mass degradation, and the potential for leachate production (See Chapter 3, pp. 59).

According to Fig 6.18, increasing oil addition would result in larger areas under the temperature curve, which is in agreement with the experimental data (except for treatments FG1 and FG2 which did not perform as expected). In terms of the final moisture content (Fig. 6.18), an increase in the initial oil concentration would result in more energy released; hence more drying would take place, resulting in lower final moisture contents. This was the trend observed in the experimental sets #1, #2a, and #2b, though the model overestimated the magnitude of the decrease in the final moisture content with oil content for those sets.

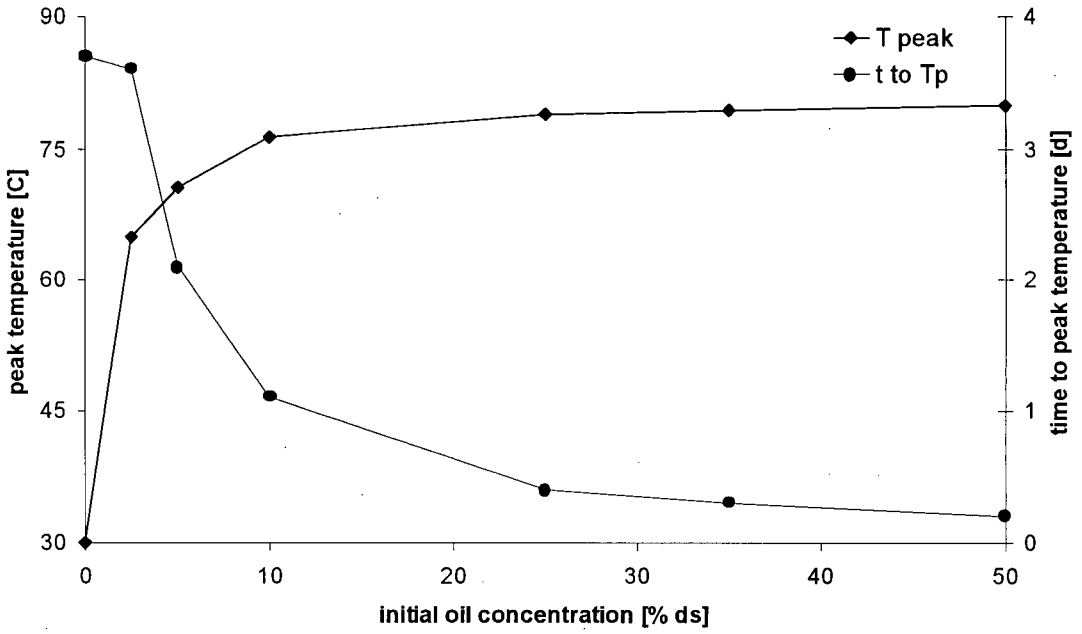

Figure 6.16 Effect of oil concentration on the peak temperature and the time to reach peak temperature.

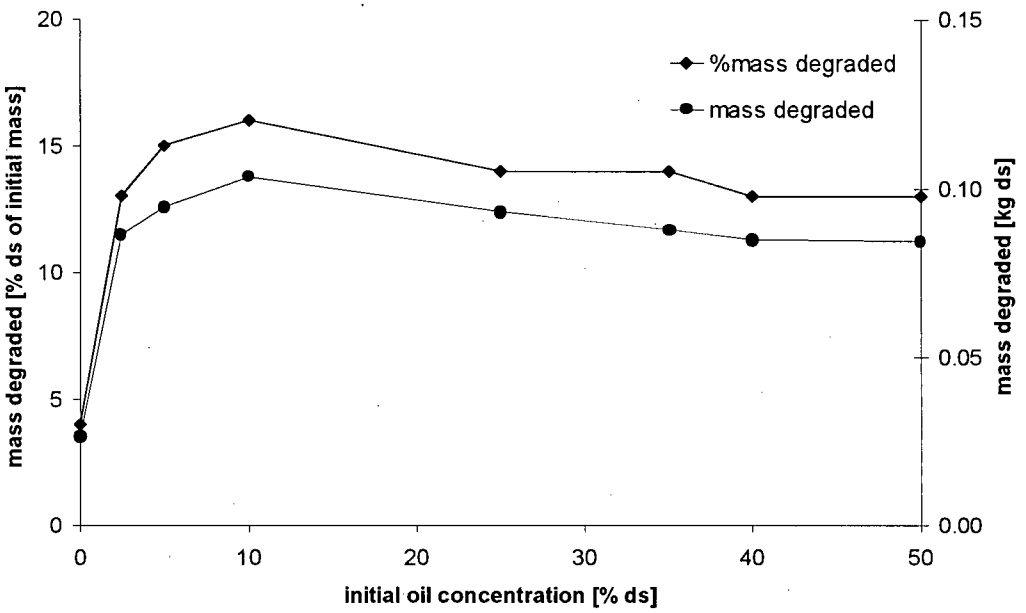

Figure 6.17 Effect of oil concentration on the amount of mass degraded.

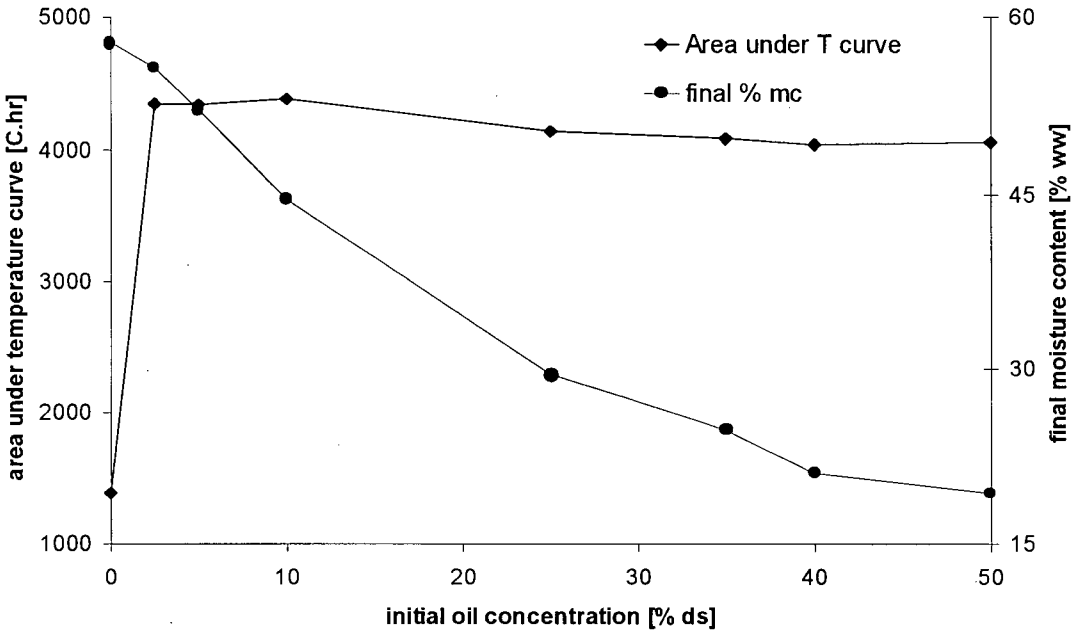

Figure 6.18 Effect of oil concentration on the area under the temperature profile and the final moisture content.

6.4.5 Model Sensitivity

The purpose of this section was to determine how the model behaves (i.e. how the predicted accuracy would be affected) due to variations of a choice of parameters, which have somewhat uncertain values used as inputs to the model. Model sensitivity evaluation was carried out by studying the effect of changing several kinetic and thermal parameters on the peak temperatures, time to reach peak temperature, and percentage mass degraded.

The effect of changing the biodegradation rate of dry solids, and lipids are shown in Figures 6.19 and 6.20. The parameters that had the largest impact in both peak temperature (T_p), and mass changes were the 'biodegradability of solids' (β_{solids}), followed by the biodegradation rate coefficient for solids (k_{solids}). The change in peak temperature and mass degradation seemed to be larger for a reduction of 50% in both biodegradability and biodegradation rate coefficient of solids (β_{solids} and k_{solids}).

The effect of changing the biodegradability and biodegradation rate coefficient of oil was less evident in the values of peak temperature and percentage mass degraded. This might be due to the small proportion of oil used (5% ds) for the model sensitivity analysis.

Figure 6.21 present the effect of changing three thermal parameters (the overall heat transfer multiplied by the area for heat transfer 'U^*A_s'; the heat of combustion of solids 'Q_{cs}'; and the heat of combustion of oil 'Q_{co}') on the peak temperature. The largest effect on the peak temperature is due to the changes in the parameter U^*A_s.

Once more, the effect of the 'solids' parameter (in this case Q_{cs}) was the largest in the peak temperature, as compared to the effect of changing 'Q_{co}'. The decrease of 'Q_{cs}' by 50% resulted in a sharp decrease in the peak temperature. It is fair to state that the largest effect of the 'solids' parameters is due to the largest proportions (95%) of solids in the composting mixture, when compared to the proportion of oil. Nonetheless, increasing the value of 'Q_{co}' resulted in a mild increase in the peak temperature, due to the larger (almost double) value of the heat of combustion of oil when compared to the heat of combustion of solids.

Figure 6.22 shows the effect of the deviation of FAS (free air space) from the assumed value of 30% as input to the model on the predicted peak temperature and the percent mass degraded. It is noticeable that the model reaches peak temperatures above 50°C only for a limited range of FAS (23 to 38%), which, according to Eq. 8c would correspond to initial moisture contents of 46 to 69% ww. This represents one limitation of the model, which might be attributed to the correlation between FAS and moisture content, as derived from data relevant to garbage and sludge co-composting (Haug 1993).

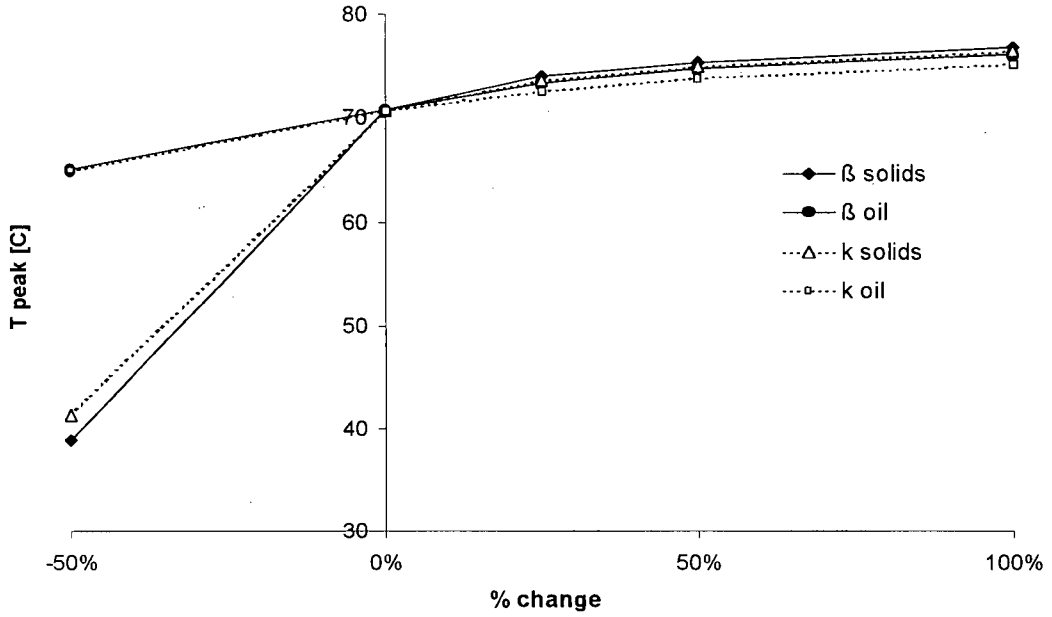

Figure 6.19. Model sensitivity. Effect of the biodegradability and biodegradation rate coefficient on the temperature peak.

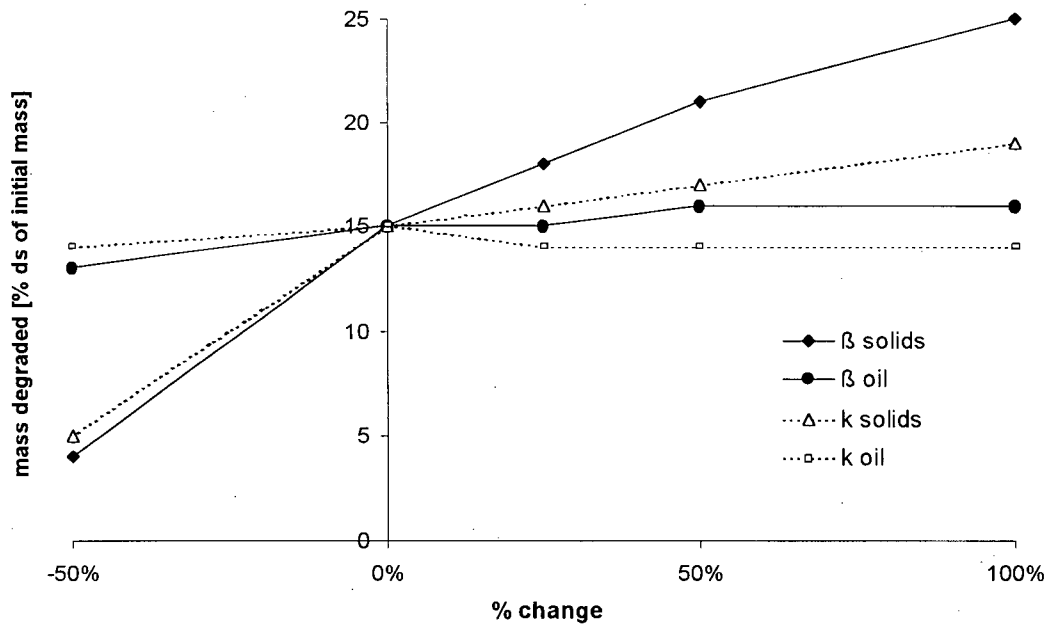

Figure 6.20. Model sensitivity. Effect of the biodegradability and biodegradation rate coefficient on the percentage mass degraded.

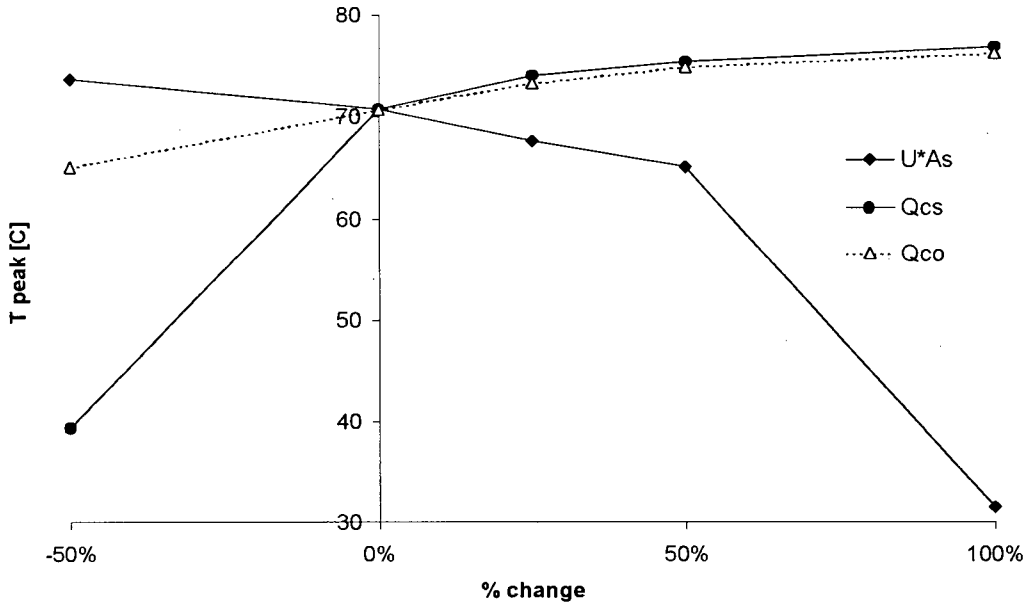

Figure 6.21 Model sensitivity. Effect of various thermal parameters on the temperature peak.

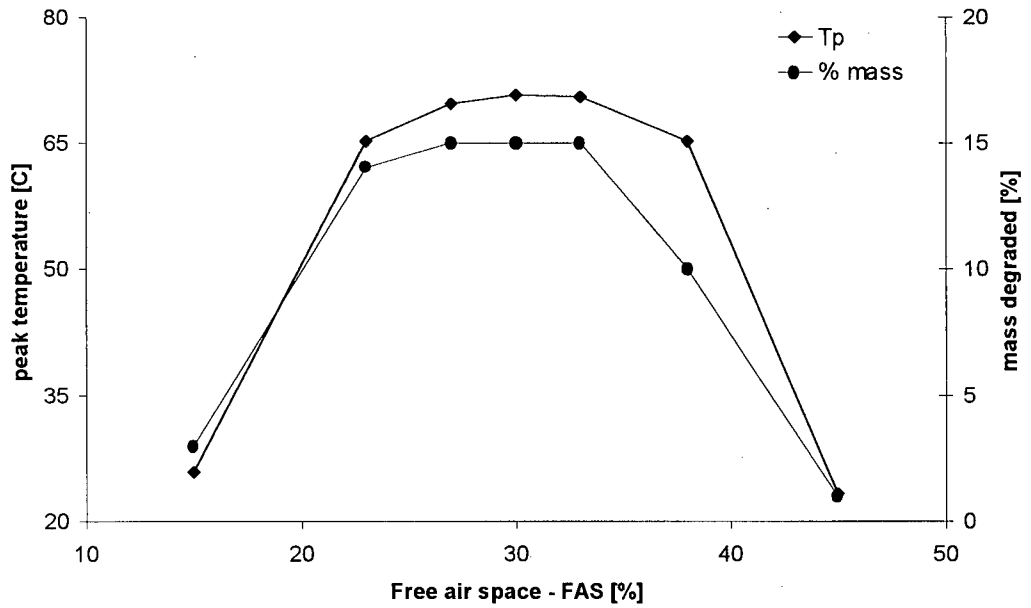

Figure 6.22 Model sensitivity. Effect of free air space (FAS) on the temperature peak and the percentage mass degraded.

6.5 CONCLUSIONS

A model to simulate the composting of food waste or yard trimmings, with lipid waste added was developed. Results of simulations are in general agreement with expected trends. The major findings were as follows:

a. A 'macrokinetic' model was developed using a dynamic modeling approach (using mass and energy balances with kinetic parameters). The model allowed for the inclusion of lipid wastes as an energy amendment. The model included provision for temperature control through aeration.
b. The use of an 'Excel®' spreadsheet proved to be useful, fast, and very practical for the solution of the model equations.
c. The model simulated fairly well the trends for the composting of yard trimmings mixtures in terms of the temperature profiles, but seemed to underestimate the area under the temperature curve for food wastes mixtures.
d. The values of the peak temperatures were accurately predicted, for either composting substrate (yard trimmings or food waste mixtures), whereas the predicted values for the time to reach the peak temperature were, at maximum, within 1 day (24 hours) of the actual measurements.
e. The simulated values for mass degradation were in general less than the actual measurements, probably due to the uncertainties in the estimation of the biodegradation rate coefficients used in the model.
f. The model was able to predict the trends for the final moisture content, with fairly accurate values for mixtures of yard trimmings or food wastes alike with grease trap sludge added.
g. The addition of oil into composting mixtures produces an increase in the peak temperature with increasing initial oil concentration values, which is in agreement with the actual peak temperatures data from the experimental treatments. However, it took shorter times to reach the peak temperature with increasing oil concentrations. This is opposite to the trend shown by the actual measurements, where the time to reach the peak temperatures increased with increasing oil concentrations. This might be explained by the fact that the 'macrokinetic' model does not have provisions to account for the changes produced in the composting matrix due to the addition of the lipid substrates.

h. According to the model, the sharpest changes in mass degradation take place in the range of initial oil concentration from 0 to 10% ds, with a slight decrease in mass degraded for higher concentrations. This conforms to the experimental data, reaffirming that in practice an addition of lipids up to 10% ds to composting mixtures might be recommended.

i. The model was very sensitive to changes in the 'solids' parameters - biodegradation rate coefficient, biodegradability, and heat of combustion. The greater sensitivity of the model paired with the solids parameters is justified by the fact that the 'solids' (non-oil materials) corresponded to the largest fraction in the composting mixture, in comparison with lipids.

j. Among the thermal parameters, the term representing the heat transfer coefficient and area for heat transfer, had the largest effect on the peak temperature, confirming that insulation of the composting reactor is key in achieving thermophilic temperatures.

k. Among the limitations of the model are, the use of only the biodegradation rate coefficients to describe the kinetics of the biological and chemical processes taking place during composting, and the estimation of the biodegradation rate coefficients values. Nevertheless, the model was able to make reasonably accurate predictions of the temperature profiles and mass changes versus the experimental data.

l. Another limitation of the model was the limited range of free air space (FAS) that would result in peak temperatures above 50°C. This might be attributed to the equation used to calculate FAS, which correspond to a moisture content range of 46 to 69%, and is based on a different substrate type (garbage and sludge).

m. It is strongly recommended, for further studies, to include the measurements of both oxygen consumption and carbon dioxide generation, in order to determine the biodegradation rate coefficient (and its change with time) with more precision.

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CHAPTER 7
CONCLUSIONS AND RECOMMENDATIONS

The aim of this thesis research was to evaluate the treatability of lipid-rich residues when composted under solid substrate aerobic conditions, using either yard trimmings or food waste as main substrates. Canola oil and grease trap sludge (mainly from restaurants) were used as lipid-rich substrates. The contents in this chapter are divided into the three aspects of composting investigated, which are: composting process performance, composting process environmental impact, and compost quality. In addition, this chapter contains the major conclusions about the composting simulation model developed in this study. Recommendations for further research, as well as suggestions for the practical application of the findings of this study, are also presented.

Composting Process Performance:

The addition of canola oil, as lipid-rich compound, presented an advantage in terms of temperature profiles, in view of the fact that the yard trimmings treatment without canola oil achieved a lower peak temperature, when compared with the yard trimmings treatments with canola oil added. However, the lower values for the biodegradability and biodegradation rate coefficient for the treatments with yard trimmings and canola oil added is an indication that the presence of lipids might have an inhibitory effect on organic matter degradation during composting.

For treatments with synthetic food waste (hereinafter referred to food waste), the addition of canola oil resulted in similar thermal performance, when compared with the treatments without canola oil added. However, the treatments with canola oil added practically fulfilled the time-temperature regulatory requirements for pathogen inactivation, whereas the treatments with food waste alone as substrate did not satisfy this requirement.

In the high-rate phase of composting, canola oil was degraded by 48 to 62% ds (dry solids) for the yard trimmings and canola oil treatments, and by 11 to 79% ds for the food waste and canola oil treatments. In contrast, when yard trimmings or food

waste was composted alone, the lipids naturally present were degraded by 75-96% ds, and by 29-61% ds, respectively.

With aeration rate as a tested parameter, doubling the standard aeration rate from 0.72 to 1.44 lpm/kg ds had virtually no effect on the thermal performance; hence the standard aeration rate should be preferred due to the higher blower and operating costs associated with higher aeration rates.

Experimental results demonstrated that, as inocula, either activated sludge or chicken litter worked very well. The difference between using a 1% or 5% ww inoculum concentration was not significant. However, a higher initial concentration of inoculum appeared to give rise to smoother temperature curves.

When grease trap sludge (GTS) was added to the main substrate, the lipids were degraded by 39 to 51% ds for the yard trimmings treatments, and by 10 to 27% ds for the food waste treatments during the high-rate phase of composting. However, after the curing phase the overall degradation of lipids was raised to 48-77% ds for the treatments with yard trimmings and GTS, and 55-43% ds for the ones with food waste and GTS. By comparison, the lipids naturally present in the yard trimmings showed an overall degradation of 96% ds; while the lipids naturally occurring in the food waste were degraded by 44% for the entire composting period.

The yard trimmings treatments with no grease trap sludge added were found to perform in the mesophilic regime with temperature peaking at 49°C. Treatments with grease trap sludge added at either 5 or 10% ds performed in the thermophilic regime with maximum temperatures between 61-67°C. Nevertheless, only the treatment with 10% ds grease trap sludge fulfilled the time-temperature regulatory requirements for pathogens reduction. Incorporating increased amounts of grease trap sludge to yard trimmings composting resulted in longer times to reach the peak temperatures, indicating that some inhibition of the composting process might have occurred at higher grease trap sludge concentrations.

For the food waste treatments, the treatment with 10% ds grease trap sludge added showed no temperature peak or plateau during the 168-hour composting period; this phenomenon is unique among most other temperature profiles in all the experimental sets. Moreover, the food waste treatment with no grease trap sludge added even outperformed the treatments with grease trap sludge added at 5 or 10% ds. Reduced porosity due to the presence of significant amount of lipids, considering the

fact that dry dog food already had a higher lipid content than yard trimmings, could be the reason for these observations versus the opposite trends in yard trimmings tests with grease trap sludge.

An original contribution of this study was the measurement of biodegradation rate coefficients, 'k', for lipid-rich wastes when these residues were added to composting mixes. In general, lipids degraded faster and more easily when compared with volatile solids biodegradation. This result is in agreement with the literature that affirms the readily degradable nature of lipids. For the high rate phase of composting, treatments with canola oil added resulted in biodegradation rate values for volatile solids (k_{vs}) of 0.009-0.039 day^{-1}, while grease trap sludge treatments resulted in k_{vs} values of 0.009-0.033 day^{-1}.

Composting Process Environmental Impact:

Preliminary tests showed that the addition of canola oil at 35% ds or more to yard trimmings composting caused the generation of an oily leachate, which is undesirable due to its requirement for collection and treatment. Hence, the upper limit of lipids concentration chosen for the grease trap sludge degradation testing was fixed at 10% ds to avoid leachate production. From here, only the gaseous emissions of the yard trimmings or food waste treatments, with or without grease trap sludge added, were studied as potential pollutants with impact on the environment.

For treatments with yard trimmings substrate, the addition of 10% ds grease trap sludge resulted in more nitrous oxide, more carbon dioxide, less ammonia, and similar odor emissions; when compared with the emissions from yard trimmings composting alone. Whereas, composting of yard trimmings with 5% ds grease trap sludge added had similar emissions (carbon dioxide, nitrous oxide, ammonia, and odor) as the control. More specifically, the yard trimmings treatment with 10% ds grease trap sludge resulted in 10 times less ammonia emitted when compared with the treatment with 5% ds grease trap sludge and control treatment, the comparative values being 0.3, 3.1, and 3.6% of total initial nitrogen, respectively.

The addition of grease trap sludge to food waste resulted in less ammonia and odor emissions when compared to food waste composting alone. Apparently, the lower emissions were due to the observed poor composting performances. Specifically, the

ammonia losses for the food waste treatments were relatively small, with values of 0.1, 0.4, and 0.8% of the total initial nitrogen corresponding to GTS added at 10% ds, 5% ds and 0% ds (the control). This may be attributed to the low pH values of the food waste mixtures.

Nitrous oxide emission for the yard trimmings treatment with 10% ds grease trap sludge added was consistently higher when compared to the treatment with 5% ds grease trap sludge, and its release was 100 times more when compared with the control treatment. The increase in nitrous oxide emissions whenever lipids were added at 10% ds might be an indication of the presence of anoxic/anaerobic pockets in the composting mix. The values of nitrous oxide emissions were similar to the findings reported in the literature, being 1.11, 0.01, and 0.02 % of total initial nitrogen for the treatments with 10% ds, 5% ds grease trap sludge, and control, respectively.

As for carbon dioxide emission, yard trimmings treatment with 10% ds grease trap sludge added resulted in more carbon dioxide emitted (10.3 CO_2-C as % of total initial carbon) when compared with the treatment with 5% ds grease trap sludge, and the control (5.7 and 6.1 CO_2-C as % of total initial carbon, respectively). Higher carbon dioxide emission would indicate enhanced microbial activity with the addition of lipids above 5% ds to the substrate. All the yard trimmings treatments (with or without grease trap sludge) produced similar methane emissions, with values close to the ambient level of 2 ppmv.

The trend of odor emission for the yard trimmings treatments followed that of the temperature profiles, with peak values for both parameters in the first 72 hours after the start of composting. Cumulative specific odor emission for the yard trimmings treatments had similar values (1.6, 1.2, and 1.5 x 10^{-5} ou/kg ds initial mass, for the treatments with 10% ds, and 5% ds grease trap sludge, and no grease trap sludge added, respectively). Cumulative specific odor emission for the food waste treatment with no grease trap sludge added was greater than that for the treatments with 10% or 5% ds grease trap sludge added (with values of 6.8, 3.5, and 3.9 x 10^{-4} ou/kg ds of initial mass, respectively). These values are one order of magnitude larger than the ones for the yard trimmings treatments.

Compost Quality:

Compost quality was measured for the yard trimmings or food waste treatments with and without grease trap sludge added, at the end of the entire composting period (high-rate phase and curing phase).

Yard trimmings treatments with grease trap sludge added at 5 or 10% ds had the largest overall volatile solids reductions, at 29 and 41% respectively, when compared with 15% volatile solids reduction associated with the control treatment. However, all the food waste treatments including the control had essentially the same extent of volatile solids reduction at 21-22%.

During the high-rate composting phase for yard trimmings, the addition of grease trap sludge at 10% ds resulted in 66% increase in nitrogen content, when compared to the 37% increase for the treatment with no grease trap sludge added. However, in terms of overall nitrogen changes, the yard trimmings treatment with 10% ds grease trap sludge added had an increase of only 34%, when compared to 76% for the treatment with no grease trap sludge added. The addition of 5% ds grease trap sludge to either food waste or yard trimmings had similar overall changes in nitrogen content when compared to the control treatments.

As for the phytotoxicity potential, which was measured via seed germination test, results indicated that the addition of lipids at 10% ds did not have a beneficial effect on curly cress seed germination. Treatments with 10% ds grease trap sludge added to either food waste or yard trimmings resulted in a germination index similar to the one for the treatment with distilled water alone; seed germination was significantly less when compared against treatments with 5% ds or no lipid added. Improved root lengths were also observed in the latter treatments. Yet, for germination test using radish seeds, root length, and germination index values were statistically similar for all treatments, suggesting that radish seeds were not affected by the different treatments examined.

Compost Model:

A 'macrokinetic' model was developed using a dynamic modeling approach (using mass and energy balances along with kinetic parameters) for the simulation of the composting process. The model allowed for the inclusion of lipid wastes as an energy amendment. It also included provision for temperature control through aeration.

The use of an 'Excel®' spreadsheet proved to be a useful, fast, and very practical means for solving the model equations.

The addition of oil into composting mixtures, according to the model, produces an increase in the peak temperature with increasing initial oil concentration values, which is in agreement with the actual peak temperature data from the experimental treatments. However, the model simulations give shorter times to reach the peak temperature with increasing oil concentrations; this presents an opposite trend to the actual measurements. This might be explained by the fact that the 'macrokinetic' model does not have provisions to account for the transient changes that occur in the composting matrix due to the addition of the lipid substrates.

The predicted values of mass degradation were in general less than the measured values, probably due to the uncertainties in the estimated biodegradation rate coefficients used as inputs to the model. The computer model was able to predict the trends for the final moisture content, yielding fairly accurate values for mixtures of yard trimmings or food wastes alike with grease trap sludge added. According to the model, the sharpest changes in mass degradation take place in the range of initial oil concentration from 0 to 10% ds, with a slight decrease in mass degraded at higher concentrations. This conforms to the experimental data, reaffirming that in practice an addition of lipids up to 10% ds to composting mixtures might be desirable.

The model was very sensitive to changes in the 'solids' parameters - biodegradation rate coefficient, biodegradability, and heat of combustion. The greater sensitivity of the model paired with the solids parameters is justified by the fact that the 'solids' (non-oil materials) represented the largest fraction in the composting mixture, in comparison with lipids. Among the thermal parameters, the term representing the heat transfer coefficient and area for heat transfer, had the largest effect on the peak temperature, confirming that insulation of the lab-scale composting reactor or a critical composting mass is key in achieving thermophilic temperatures.

Among the limitations of the model are, the use of only the biodegradation rate coefficients to describe the kinetics of the biological and chemical processes taking place during composting, and the estimation of the biodegradation rate coefficients values. Furthermore, the model was limited by the range of free air space (FAS) that would result in peak temperatures above 50°C. This might be attributed to the equation used to calculate FAS, which is based on a different type of substrate, and on a limited

range of moisture content values (46-69%) that would result in an optimal FAS (with values around 30%). Nevertheless, the model was able to make reasonably accurate predictions of the temperature profiles and mass changes versus the experimental data.

Recommendations:

It is strongly recommended, for further studies, to include the measurements of both oxygen consumption and carbon dioxide generation, in order to determine the change of biodegradation rate coefficients with time and with more precision. To achieve this objective, several reactors can be run in parallel, with the same treatment conditions. Sampling for mass degradation shall be performed at regular intervals with an aim to obtain data for relating different thermal and kinetic parameters over time.

Pilot-scale composting studies involving lipid-rich wastes and various major substrate should be performed, and preferably using the windrow composting technology, in order to evaluate the impact of adding lipid-rich wastes to open composting systems. It is also recommended to conduct actual plant growth trials with soils having various amounts of cured compost applied.

Further studies shall also be performed with other types of lipid-rich waste, such as fish processing wastes, which might save disposal costs for the producers, and favor the environment of British Columbia.

Practical Recommendations:

As a practical recommendation, yard trimmings composting with grease trap sludge added at 5% ds would result in enhanced thermal performance, improved rate and extent of biodegradation of solids and lipids, greater overall reduction in wet mass and water content, when compared with the composting of yard trimmings alone. Higher biodegradation rates would save the capital investment and operating costs for a composting facility (for more details see Appendix K on the economics of composting grease trap sludge).

APPENDIX A
NOTATION

% FAS	=	free air space, in [%]
% $Oil_{contribution}$	=	percentage oil contribution to total heat produced
% $RH_{exhaust}$	=	relative humidity of the exhaust gases, in [%]
% RH_{in}	=	relative humidity of the air inlet, in [%]
a,b,c,d	=	coefficients of the substrate chemical formula $C_aH_bO_cN_d$
A_{curve}	=	area under the temperature curve, in [°C.hr]
A_s	=	surface area for heat transfer, in [m^2]
BOD	=	biological oxygen demand
BVS	=	biodegradable volatile solids, in [kg ds]
C	=	canola oil in experimental treatments nomenclature
C	=	carbon content
C:N	=	carbon-to-nitrogen ratio, also as C/N
CH_4-C	=	carbon from methane
CO_2-C	=	carbon from carbon dioxide
COD	=	chemical oxygen demand
C_{pc}	=	specific heat of compost, in [$MJ.kg^{-1}.°C^{-1}$]
C_{po}	=	specific heat of oil, in [$MJ.kg^{-1}.°C^{-1}$]
C_{pw}	=	specific heat of water, in [$MJ.kg^{-1}.°C^{-1}$]
db	=	dry basis
ds	=	dry solids
F	=	food waste in experimental treatments nomenclature
F1-Fn	=	mathematical functions for physico-chemical parameters affecting 'k'
FOG	=	fats, oil, and grease residues
F_T	=	temperature correction factor for the biodegradation rate coefficient
FW	=	food waste
G	=	germination, in phytotoxicity test
G	=	grease trap sludge in experimental treatments nomenclature
GI	=	germination index, in phytotoxicity test
GTS	=	grease trap sludge

h_1 and h_2	=	enthalpies of inlet and outlet air, in [$kJ.kg^{-1}$]
HEM	=	hexane extractable materials
k	=	biodegradation rate coefficient, in [day^{-1}]
k_{20}	=	biodegradation rate coefficient at reference temperature 20°C
k_m	=	maximum biodegradation rate coefficient at a given temperature
k_{T1}	=	biodegradation rate coefficient at reference temperature, T1
L	=	liters
lpm	=	liters per minute
M	=	composting mass, in [kg ww]
m_a	=	mass flowrate of air passing through the composting materials, [kg/s]
M_{BVS}	=	mass of biodegradable volatile solids, in [kg ds]
mc	=	moisture content, in [decimal]
min	=	minute
M_o	=	mass of oil or lipids in, in [kg ds]
M_s	=	mass of solids, in [kg ds]
M_{total}	=	total composting mass, in [kg ww]
M_{vs}	=	mass of volatile solids, in [kg ds]
M_w	=	mass of water, in [kg ww]
MW	=	molecular weight, in [kg/kg mol]
N	=	nitrogen content
n	=	number of replicates
N_2O-N	=	nitrogen from nitrous oxide
NH_3-N	=	nitrogen from ammonia
oil	=	oil content, in [decimal]
ou	=	odor units
P	=	atmospheric pressure, in [kPa]
P_s	=	partial pressure at saturation, in [kPa]
Pw	=	partial pressure of moist air, in [kPa]
Q	=	heat, in [MJ]
Q_a	=	heat accumulated in the composting mass, in [MJ]
Q_c	=	heat of combustion, in [$kJ.kg^{-1}$]
Q_{co}	=	heat of combustion of oil, in [$kJ.kg^{-1}$]
Q_{cs}	=	heat of combustion of solids, in [$kJ.kg^{-1}$]

Q_l	=	heat loss by conduction/convection, in [MJ]
Q_p	=	heat produced, in [MJ]
RH	=	relative humidity, in [%]
RL	=	root length, in [mm], in phytotoxicity test
s	=	solids content, in [decimal]
SD	=	standard deviation
T	=	Temperature, in [°C]
t	=	time, in [hr]
t_{55}	=	time that the composting mix is at temperature ≥ 55°C, in [hr]
T_{amb}	=	ambient temperature, in [°C], also as T_o
T_{avg}	=	average temperature, in [°C]
T_{db}	=	dry bulb temperature, in [K]
TLW	=	trucked liquid wastes, at Iona Island WWTP
T_o	=	ambient temperature, in [°C], also as T_{amb}
T_p	=	temperature peak, in [°C]
t_p	=	time to temperature peak, in [hour]
T_{set}	=	temperature set point for process control, in [°C]
U	=	overall heat transfer coefficient, in [$W.m^{-2}.°C^{-1}$]
vs	=	volatile solids content, in [decimal]
W	=	humidity ratio, in [kg water vapor/kg dry air]
wb	=	wet basis
ww	=	wet weight
WWTP	=	waste water treatment plant
Y	=	yard trimmings in experimental treatments nomenclature
YW	=	yard trimmings
β	=	biodegradability or biodegradation extent
ρ_{air}	=	density of air, [1.23 $g.L^{-1}$]
ρ_b	=	bulk density of the composting mix, in [$kg.m^{-3}$]

Subindices

a	=	accumulated
biodegraded	=	mass biodegraded, same as degraded

BVS	=	biodegradable volatile solids
change	=	parameter change from initial to final conditions
degraded	=	mass degraded, same as biodegraded
i	=	any component in the composting mix, out f 'n' components
l	=	lost
lipids	=	used interchangeably with oil
o	=	oil, also lipids
oil	=	used interchangeably with lipids
p	=	produced
s	=	solids
solids	=	solids
t	=	current time step
vs	=	volatile solids
w	=	water

APPENDIX B
COMPARISON BETWEEN THE MEASURED AND CALCULATED CARBON CONTENT

Following is a compilation of measured and calculated carbon content values for several composting mixes samples. The carbon concentration was measured using a CN Carlo Erba NA-1500 Analyzer (accuracy ± 0.3%). Calculated carbon content values were obtained from the measured volatiles solids values (n=3, obtained by gravimetric analysis and ash content - ignition at 550°C for 2 hours, test accuracy ± 6.5%). The carbon content was calculated using the 'New Zealand' formula:

$$\%\,Carbon = \frac{100 - \%\,Ash}{1.8} = \frac{\%\,Volatile\ Solids}{1.8}$$

The difference between the measured and calculated values was: **-10.5 ± 12.5 %**

(n = 19)

Treatment - % initial lipids	Composting time (days)	Measured carbon content, n=3 (% ds)	Calculated carbon content, based on VS values (% ds)	Error between measured and calculated C values
YW + 10%	0	42.50	52.97	-24.63
YW + 10%	0	42.50	51.74	-21.74
FW + 5%	0	62.25	51.59	17.13
FW Control	0	62.24	52.26	16.04
FW + 5%	0	62.25	52.85	15.09
YW Control	0	44.59	51.94	-16.49
YW + 5%	7	45.92	52.20	-13.67
YW + 10%	7	47.61	51.31	-7.76
YW Control	7	44.87	51.17	-14.04
YW + 5%	7	46.24	51.22	-10.78
YW + 10%	7	46.10	50.58	-9.72
FW Control	7	47.68	51.36	-7.71
YW + 5%	7	47.44	51.35	-8.25
YW + 10%	133	43.65	52.94	-21.27
YW Control	133	44.16	52.42	-18.71
YW + 5%	133	44.13	51.88	-17.56
YW + 10%	133	46.09	51.22	-11.14
FW Control	133	43.95	51.06	-16.19
FW + 5%	133	41.01	50.73	-23.70

YW: yard trimmings, FW: food waste, lipids added were GTS: grease trap sludge, Control: No lipid added, VS: volatile solids.

APPENDIX C
MEASUREMENT OF OIL AND MOISTURE IN GREASE TRAP SLUDGE

The measurement of oil content in greased trap sludge (GTS) was performed by modifying the AOAC Official Method 926.12 "Moisture and Volatile Matter in Oils and Fats" (Section 41.1.02, AOAC 2000) used for measurement of oil in food products. This was necessary since the traditional methods of drying (conventional oven, APHA 1995) will also consume the oil in the GTS sample, thus producing a false reading for moisture content.

The developed protocol used vacuum drying for extracting the water in the GTS sample. This was achieved using a vacuum oven set at 28 psi vacuum. Lipids were measured by the USEPA (1998) protocol on the GTS sample after vacuum drying. Ash was measured according to APHA (1995), and nitrogen was measured using a LECO analyzer (FP 228, Leco Corp., St. Joseph, MI) on the dry sample.

The measurements were performed in 7 replicates to ensure result accuracy. The moisture content protocol details are as follows:

1. Weigh ca. 30.0000 g wet weight of grease trap sludge in 100 ml beaker.
2. Cover with clean, weighed watch glass.
3. Dry in vacuum oven at 28 psi vacuum (oven temperature ~40°C), for 4 hours or until bubbling has reduced to a minimum.
4. Weigh beaker residue and watch glass residue.
5. Calculate the difference between initial weight and residue. This corresponds to the moisture content of the sample.
6. Weigh 5 g of the residue and follow the Sohxlet extraction procedure according to USEPA (1998).

In order to cross check the values obtained by the previous protocol, grease trap sludge samples were frozen at 4°C for 24 hours, after that the grease layer was separated manually and weighed. Results from this procedure were in accordance with the values measured by vacuum drying and oil measurement, with lipid values of approximately 30% ww.

The results for the analysis of grease trap sludge samples (n = 7) were:

Parameter	Value
Moisture content	61 ± 10% ww
Lipids	31 ± 26% ww
Nitrogen	0.31 ± 0.1 % ds
Ash	1.4 ± 0.2 % ds

The measured values found for the different chemical parameters in grease trap sludge are in accordance with the values suggested by Plante and Voroney (1998), namely moisture content 80-95%, ash ~ 0%, lipid content 35%, and negligible nitrogen content.

Fernandes et al. (1988) found that flotation foams had a moisture content of 45-60%, lipid content of 60-65% ds, 3-6% ash, and 0.8-1.2% ds nitrogen. In contrast, slaughterhouse wastes had 88-86% moisture content, 25-55% lipids, 6-7% ash, and 3-4% ds nitrogen.

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APPENDIX D
COMPOSTING RECIPE CALCULATION WORKSHEET

RAW MATERIALS	% C db	% N db	% MC wb	% Oil wb	Assumed Wet mass in kg	%Prop wb	%Prop db
Grass clippings	49.99	4.67	72.8	0.7	**0.828**	53	33
Grease trap sludge	54.80	0.31	95.0	35	**0.204**	13	2
Sawdust	55.24	0.37	7.7	0.9	**0.240**	15	33
Woodchips	52.85	0.38	11.8	1.2	**0.204**	13	27
Chicken manure	43.05	3.08	53.0	0.4	**0.084**	5	6
Urea	55.47	46.00	1.2	0	**0.000**	0	0
Water	0	0	100	0	**0.000**	0	0
Total mass composting mix					**1.56**	100	100
Initial carbon-to-nitrogen ratio (C:N)					**26.6**		
Initial moisture content					**56.6**		

% inoculum	**5.4**
% oil ww	**5.3**
% oil db	**6.4**

Airflow calculation: Using 0.72 L/min per kg dry matter

Total dry mass to be composted (kg db):	**0.68 kg db**
Airflow - 100% standard (33% duty cycle)	0.49 L/min

APPENDIX E
SAMPLE CALCULATIONS FOR COMPOST MASS CHANGES (CHAPTER 3)

Following is an example of the calculations for the compost mass changes presented in Chapter 3. The initial and final mass values for treatment FC6 were as follows:

	Initial	Final
Total mass [kg ww]	1.406	0.927
Moisture content [% ww]	55.0	42.8
Volatile solids content [% ds]	97.7	54.7
Lipids content [% ds]	10.6	2.2
Mass of solids: $M_s = (1-mc) * M_{total}$	0.633	0.530
Mass of volatile solids: $M_{vs} = vs * M_s$	0.618	0.507
Mass of oil: $M_o = oil * M_s$	0.067	0.020
Mass of water: $M_w = mc * M_{total}$	0.773	0.397

Biodegradation rate coefficient for lipids [d^{-1}]:

$$k_{lipids} = \frac{\ln\left(\frac{M_{oil,\ t=168\ hr}}{M_{oil,\ t=0\ hr}}\right)}{168\ hr} = \frac{\ln\left(\frac{0.020}{0.067}\right)}{168} = -0.0072\frac{1}{hr} * \frac{24\ hr}{1\ d} = -0.173\ d^{-1}$$

Biodegradation rate coefficient for volatile solids [d^{-1}]:

$$k_{vs} = \frac{\ln\left(\frac{M_{vs,\,t=168\,hr}}{M_{vs,\,t=0\,hr}}\right)}{168\,hr} = \frac{\ln\left(\frac{0.507}{0.618}\right)}{168} = -0.0012\frac{1}{hr} * \frac{24\,hr}{1d} = -0.029\,d^{-1}$$

Biodegradability of lipids [%]:

$$\beta_{lipids} = \left(\frac{M_{oil,\,t=0} - M_{oil,\,t=168}}{M_{oil,\,t=0}}\right) * 100 = \left(\frac{0.067 - 0.020}{0.067}\right) * 100 = 70\%$$

Biodegradability of volatile solids [%]:

$$\beta_{vs} = \left(\frac{M_{vs,\,t=0} - M_{vs,\,t=168}}{M_{vs,\,t=0}}\right) * 100 = \left(\frac{0.618 - 0.507}{0.618}\right) * 100 = 18\%$$

Change in mass total [as % of initial mass total]:

$$M_{total,\,change} = \left(\frac{M_{total,\,t=0} - M_{total,\,t=168}}{M_{total,\,t=0}}\right) * 100 = \left(\frac{1.406 - 0.927}{1.406}\right) * 100 = 34.1\%$$

Change in mass of water [as % of initial water]:

$$M_{water,\,change} = \left(\frac{M_{water,\,t=0} - M_{water,\,t=168}}{M_{water,\,t=0}}\right) * 100 = \left(\frac{0.773 - 0.397}{0.773}\right) * 100 = 48.6\%$$

Amount of lipids degraded [kg lipids]:

$$M_{oil,\,degraded} = M_{oil,\,t=0} - M_{oil,\,t=168} = 0.067 - 0.020 = 0.047$$

Amount of volatile solids degraded [kg vs]:

$$M_{vs,\,degraded} = M_{vs,\,t=0} - M_{vs,\,t=168} = 0.618 - 0.507 = 0.111$$

Amount of 'non oil' volatile solids degraded [kg vs]:

$$M_{non\,oil\,vs,\,degraded} = M_{vs,\,degraded} - M_{oil,\,degraded} = 0.111 - 0.047 = 0.064$$

APPENDIX F
OXYGEN DEMANDS FOR DIFFERENT SUBSTRATES

FOOD WASTE:

$$C_{18}H_{26}O_{10}N + 18.75 * O_2 \rightarrow 18 * CO_2 + 11.5 * H_2O + 1 * NH_3$$

417 g 600 g

oxygen demand for food waste: $\frac{600}{417} = 1.4 \frac{g\,O_2}{g\,\text{food waste}}$

YARD TRIMMINGS:

$$C_{27}H_{38}O_{16}N + 27.75 * O_2 \rightarrow 27 * CO_2 + 17.5 * H_2O + 1 * NH_3$$

632 g 888 g

oxygen demand for food waste: $\frac{888}{632} = 1.4 \frac{g\,O_2}{g\,\text{yard trimmings}}$

CANOLA OIL:

$$C_{18}H_{33}O_2 + 25.25 * O_2 \rightarrow 18 * CO_2 + 16.5 * H_2O$$

281 g 808 g

oxygen demand for canola oil: $\frac{808}{201} = 2.9 \frac{g\,O_2}{g\,\text{canola oil}}$

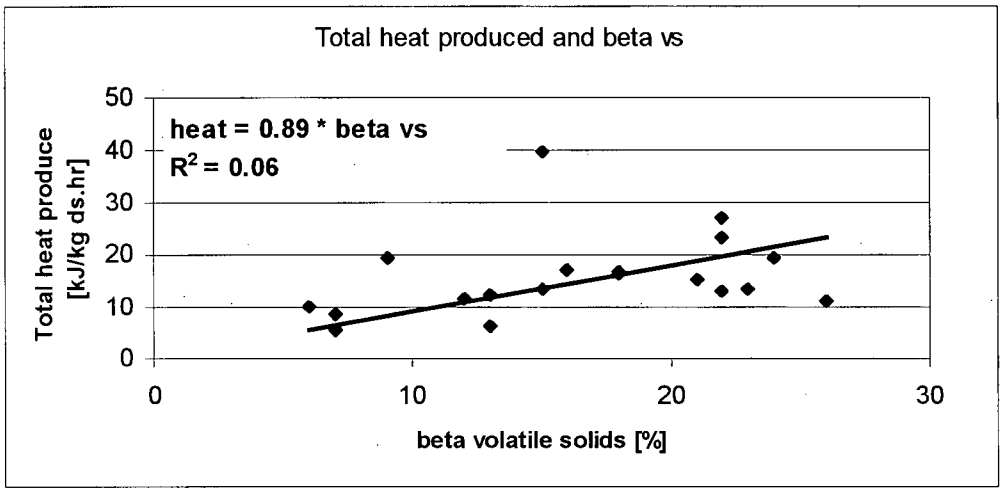
FATS AND OILS:

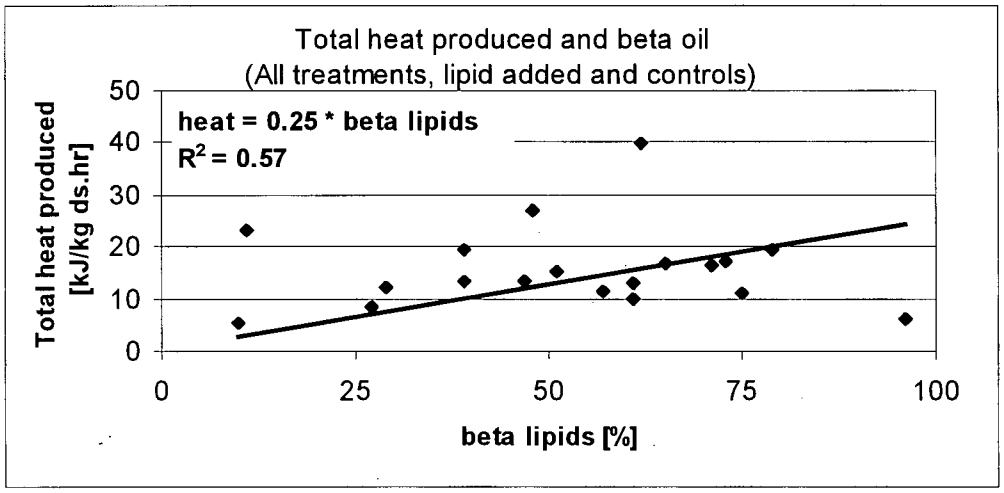
$$C_{50}H_{90}O_6 + 69.5 * O_2 \rightarrow 50 * CO_2 + 45 * H_2O$$

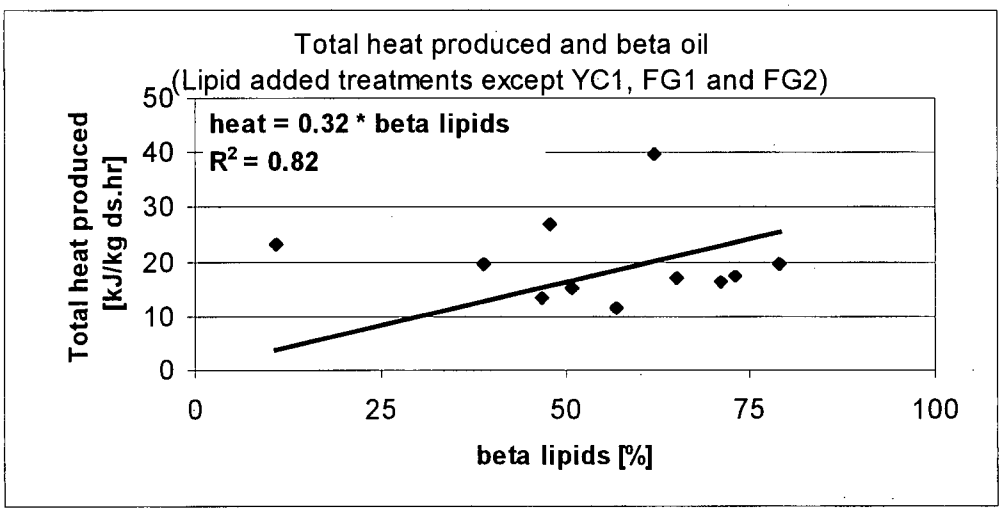
786 g 2224 g

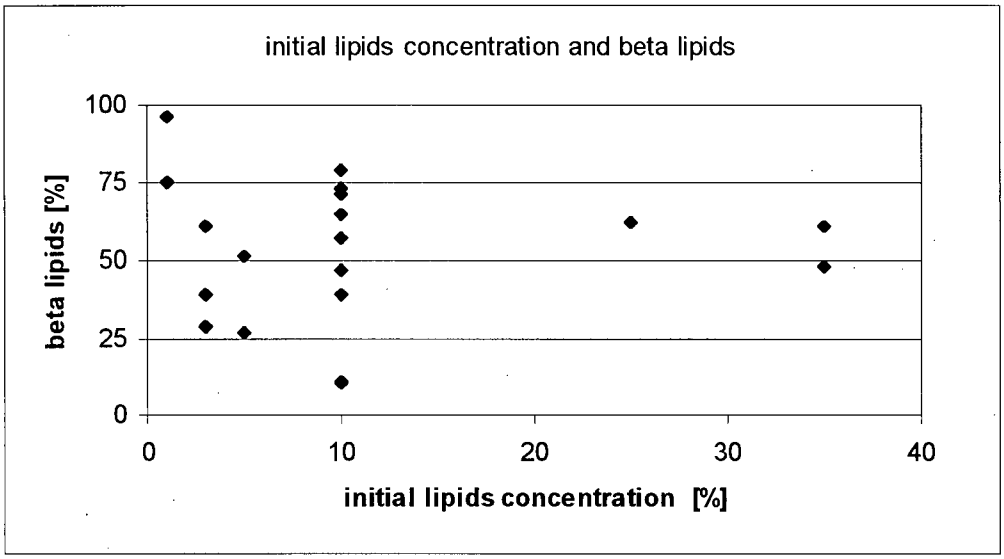
oxygen demand for food waste: $\frac{2224}{786} = 2.8 \frac{g\,O_2}{g\,\text{fats and oil}}$

APPENDIX G
CORRELATIONS BETWEEN TOTAL HEAT PRODUCED AND VOLATILES SOLIDS OR LIPIDS DEGRADED








initial lipids concentration and beta lipids
beta lipids [%]
100
75
50
25
0
0
10
20
30
40
initial lipids concentration [%]

APPENDIX H
HEAT TRANSFER COEFFICIENT AND HEAT TRANSFER AREA CALCULATIONS

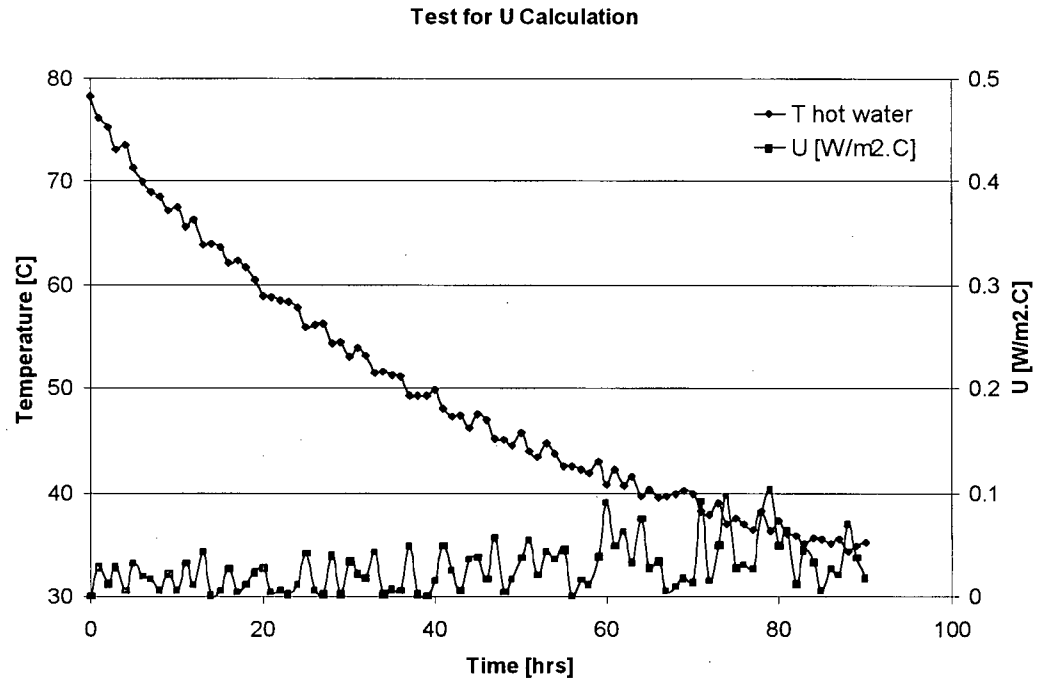
The area for heat transfer by conduction/convection was the outer surface area of the adiabatic box. Calculations are as follows:

Sides: 4 sides * 70 cm wide * 50 cm height * $1m^2/10000\ cm^2$ = 1.4 m^2 total sides

Top: 50 cm * 50 cm * $1m^2/10000\ cm^2$ = 0.25 m^2 total top

A = Total area = 1.4 + 0.25 = 1.65 m^2

The overall heat transfer coefficient (U) was measured by testing the laboratory bench-scale set up with hot water. The temperature decrease over time is shown in the following graph. The value of 'U' was calculated as the average of the hourly 'U' values.



Mass water	4 kg
Cp water	1.0006 kJ/kg.C
A box	1.65 m2
U (n = 91)	**0.028 ± 0.002 W/(m^2.°C)**

APPENDIX I
CORRELATION BETWEEN 'k empirical' AND TEMPERATURE

Graphs I.1 and I.2 show the correlations between the values of the empirical biodegradation rate coefficient ($k_{empirical}$), and the temperature calculated from the chosen 'alpha factor' (α), assuming that the 'alpha' factor follows the same equation reported by Haug (1993) for the temperature factors. Note that both values, $k_{empirical}$ and the temperature (derived from the chosen α value) were measured/chosen independently; nevertheless their relationship when plotted as the ln(k) vs. 1/T still follows the Arrhenius equation trend.

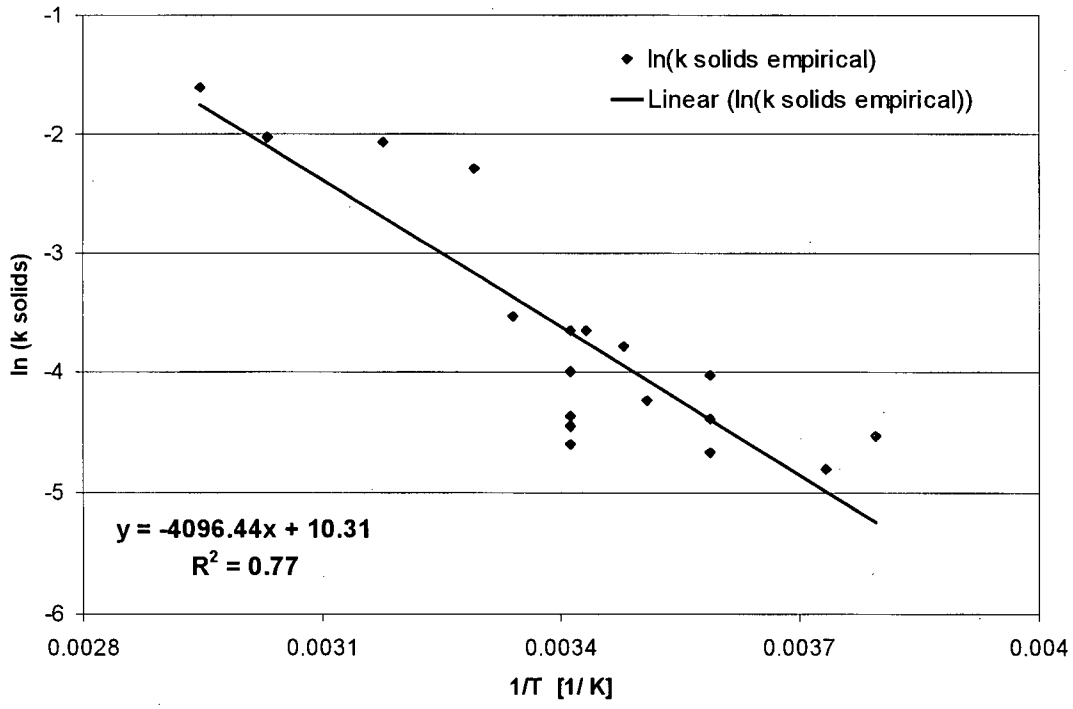


Figure I.1 Relationship between the empirical biodegradation rate coefficient for solids and the arbitrarily chosen temperature.

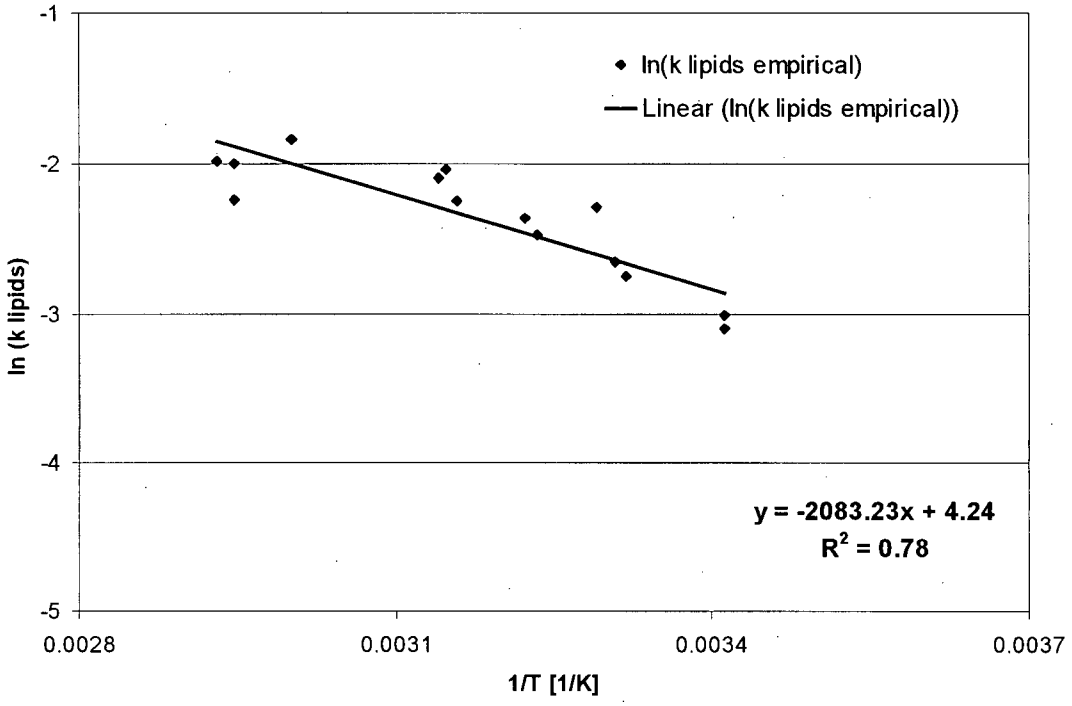


Figure I.2 Relationship between the empirical biodegradation rate coefficient for lipids and the arbitrarily chosen temperature.

APPENDIX J
SAMPLE CALCULATIONS FOR THE BIODEGRADATION COMPOST MODEL (CHAPTER 6)

Following is an example of the calculations for the biodegradation compost model presented in Chapter 6. The input values for treatment YC2 modeling were as follows:

INPUTS		INPUTS	
Time step	3 min	U	0.03 $W/m^2.°C$
HRT	7 days	As	1.65 m^2
k empirical solid	0.100 day^{-1}	Cp solids	0.00065 MJ/kg.°C
k empirical oil	0.106 day^{-1}	Cp water	0.00418 MJ/kg.°C
α solids	0.50	Cp oil	0.00190 MJ/kg.°C
α oil	0.06		
C1	1.07		Yard trimmings Oil
C2	1.21	C (a)	27 18
Initial mass total	1.65 kg ww	H (b)	38 33
Initial mc	55 % ww	O (c)	16 2
Initial FAS	32 %	N (d)	1 0
Initial % Oil	35 % ds	MW	632 281
Initial % Oil	24.3 % ww		
β solids	22 % ds	Airflow	0.53 lpm
β oil	48 % oil	ρ_{air}	1.23 g/L
T_{amb}	23.9 °C	ma 33%	0.0007 kg air
T_{set}	65 °C	ma 100%	0.0020 kg air
Qco	38.33 MJ/kg	RH air in	25 %
Qcs	15.1 MJ/kg	RH air out	100 %

Coefficients		
	a_1	17.684
	a_2	7.0622
	a_3	23.675
	a_4	3.4945
	a_5	1.0375
	a_6	1.525
	a_7	Food waste: 0.50 Yard trimmings: 0.50
	a_8	1.06

Time step, dt:

$$dt = \frac{1}{480} = 0.0021 \text{ [day]}$$

Initial composting mix temperature, T:

$$T = T_{amb} = 23.9 \quad [^{o}C]$$

Initial mass of solids, M_s:

$$M_s = \left(1 - \frac{\%mc}{100} - \frac{\%oil}{100}\right) * M = \left(1 - \frac{55}{100} - \frac{24.3}{100}\right) * 1.65 = 0.3416 \quad \text{[kg solids]}$$

Initial mass of water, M_w:

$$M_w = \frac{\%mc}{100} * M = \frac{55}{100} * 1.65 = 0.9075 \text{ [kg water]}$$

Initial mass of oil, M_o:

$$M_o = \frac{\%oil}{100} * M = \frac{24.3}{100} * 1.65 = 0.4009 \text{ [kg oil]}$$

Initial mass of BVS solids, $M_{BVS\ solids}$:

$$M_{BVS\ solids} = \frac{\%\beta_{solids}}{100} * M_s = \frac{22}{100} * 0.3416 = 0.0751 \quad \text{[kg ds]}$$

Initial mass of BVS oil, $M_{BVS\ oil}$:

$$M_{BVS\ oil} = \frac{\%\beta_{oil}}{100} * M_o = \frac{48}{100} * 0.4009 = 0.1924 \quad \text{[kg oil]}$$

Biodegradation rate coefficient of solids at T=20°C:

$$k_{20,\ solids} = \alpha * k_{empirical,\ solids} = 0.5 * 0.100 = 0.050 \quad [day^{-1}]$$

Biodegradation rate coefficient of oil at T=20°C:

$$k_{20,\ oil} = \alpha * k_{empirical,\ oil} = 0.06 * 0.106 = 0.006 \text{ [day}^{-1}\text{]}$$

Biodegradation rate coefficient maximum solids, $k_{m,\ solids}$ with T effect:

$$k_{m,\ solids} = k_{20,\ solids} * (C1^{T-20} - C2^{T-60})$$

$$k_{m,\ solids} = 0.05 * (1.066^{23.9-20} - 1.21^{23.9-60}) = 0.064 \quad [day^{-1}]$$

Biodegradation rate coefficient maximum oil, $k_{m,\ oil}$ with T effect:

$$k_{m,\ oil} = k_{20,\ oil} * (C1^{T-20} - C2^{T-60})$$

$$k_{m,\ oil} = 0.006 * (1.066^{23.9-20} - 1.21^{23.9-60}) = 0.0075 \quad [day^{-1}]$$

F1, moisture content effect on the biodegradation rate coefficient, in [decimal]:

$$F1 = \frac{1}{EXP(-a_1 * mc + a_2) + 1} = \frac{1}{EXP\left(-17.684 * \frac{55}{100} + 7.0622\right) + 1} = 0.935$$

Free air space, % FAS:

$$\%FAS = \frac{\left(a_5 - \frac{\%mc}{100}\right)}{a_6} * 100 = \frac{\left(1.0375 - \frac{55}{100}\right)}{1.525} * 100 = 32 \ [\%]$$

F2, free air space effect on the biodegradation rate coefficient, in [decimal]:

$$F2 = \frac{1}{EXP(-a_3 * FAS + a_4) + 1} = \frac{1}{EXP\left(-23.675 * \frac{32}{100} + 3.4945\right) + 1} = 0.983$$

Biodegradation rate coefficient solids, k_{solids}:

$$k_{solids} = k_{m,\ solids} * F1 * F2 = 0.064 * 0.935 * 0.983 = 0.0588 \ [day^{-1}]$$

Biodegradation rate coefficient oil, k_{oil}:

$$k_{oil} = k_{m,\ oil} * F1 * F2 = 0.0075 * 0.935 * 0.983 = 0.0069 \quad [day^{-1}]$$

Amount of BVS solids degraded:

$$M_{BVS,\ solids\ degraded} = M_{BVS\ solids} - M_{BVS,\ solids} * EXP(-k_{solids} * dt)$$

$$M_{BVS,\ solids\ degraded} = 0.0751 - 0.0751 * EXP(-0.0588 * 0.0021) = 9.275E-06 \quad [kg\ ds]$$

Amount of BVS oil degraded:

$$M_{BVS,\ oil\ degraded} = M_{BVS\ oil} - M_{BVS,\ oil} * EXP(-k_{oil} * dt)$$

$$M_{BVS,\ oil\ degraded} = 0.1924 - 0.1924 * EXP(-0.0069 * 0.0021) = 2.779E-06 \quad [kg\ oil]$$

Heat produced, Q_p:

$$Q_p = M_{BVS,\ solids\ degraded} * Q_{cs} + M_{BVS,\ oil\ degraded} * Q_{co}$$

$$Q_p = 9.275E-06 * 15.1 + 2.779E-06 * 38.33 = 2.466E-04 \quad [MJ]$$

Heat loss by convection, $Q_{l,\ convection}$:

$$Q_{l,\ convection} = UA_s * (T - T_{amb}) = 0.03 * 1.65 * (23.9 - 23.9) = 0 \quad [MJ]$$

Aeration mass, m_a for $T < T_{set}$, in [kg air]:

$$m_{a,\ intermittent} = (M_s + M_o) * 0.72 * \rho_{air} * 0.33 * dt$$

$$m_a = (0.3416 + 0.4009) * (0.72 * 60 * 24) * \left(\frac{1.23}{1000}\right) * 0.33 * 0.0021 = 0.00066$$

Heat loss in the exhaust gases, $Q_{l,\ exhaust}$, in [MJ]:

$$Q_{l,\ exhaust} = m_a * (h_2 - h_1) = 0.00066 * (71.452 - 35.620) * \frac{1}{1000} = 2.35E-05$$

Heat loss, Q_l, in [MJ]:

$$Q_l = Q_{l,\ convection} + Q_{l,\ exhaust} = 0 + 2.35E-05 = 2.35E-05$$

Heat accumulated, Q_a, in [MJ]:

$$Q_a = Q_p - Q_l = (2.466E-04) - (2.35E-05) = 2.23E-04$$

Water produced, $M_{w,p}$, in [kg water]:

$$M_{w,p} = a_7 * M_{BVS,\ solids\ degraded} + a_8 * M_{BVS,\ oil\ degraded}$$

$$M_{w,p} = 0.50 * (9.275E-06) + 1.06 * (2.779E-06) = 7.56E-06$$

Water losses, $M_{w,l}$, in [kg water]:

$$M_{w,l} = m_a (W_2 - W_1) = 0.00066 * (0.019 - 0.005) = 9.25E-06$$

Mass solids, t+dt, in [kg solids]:

$$M_{s,\ t+dt} = M_{s,\ t} - M_{BVS,\ solids\ degraded} = 0.3416 - (9.275E-06) = 0.34159$$

Mass solids, t+dt, in [kg solids]:

$$M_{s,\ t+dt} = M_{s,\ t} - M_{BVS,\ solids\ degraded} = 0.3416 - (9.275E-06) = 0.34159$$

Mass water, t+dt, in [kg water]:

$$M_{w,\ t+dt} = M_{w,\ t} + M_{w,\ p} - M_{w,\ l} = 0.9075 + (7.56E-06) - 9.255E-06 = 0.9074$$

Mass oil, t+dt, in [kg oil]:

$$M_{o,\ t+dt} = M_{o,\ t} - M_{BVS,\ oil\ degraded} = 0.4009 - (2.779E-06) = 0.40089$$

Mass total, t+dt, in [kg ww]:

$$M = M_{water} + M_{solids} + M_{oil} = 0.9074 + 0.34159 + 0.40089 = 1.649$$

Moisture content, t+dt, in [% ww]:

$$\%mc_{t+dt} = \frac{M_{w,\ t+dt}}{M_{total,\ t+dt}} = \frac{0.9074}{1.649} * 100 = 55$$

Mass average per component by the specific heat, M*Cp, in [MJ/°C]:

$$(M * C_p)_t = \sum_{i=1}^{n} \left(\frac{M_t + M_{t+dt}}{2} * C_p \right)_i$$

$$(M * C_p)_{t+dt} = \left(\frac{0.9075 + 0.9074}{2}\right) * 0.00418 + \left(\frac{0.3416 + 0.34159}{2}\right) * 0.00065 + \left(\frac{0.4009 + 0.40089}{2}\right) * 0.00190 = 0.0048$$

Temperature, t+dt, in [°C]:

$$T_{t+dt} = T_t + \frac{Q_{a,t}}{(M * C_p)_t} = 23.9 + \frac{2.23E-04}{0.0048} = 23.93$$

Psychometric Subroutine Equations:

Partial pressure at saturation for the inlet air, in [kPa]:

$$P_{s1} = EXP\left(52.58 - \frac{6790.5}{T_{amb} + 273} - 5.028\ln(T_{amb} + 273)\right)$$

$$P_{s1} = EXP\left(52.58 - \frac{6790.5}{23.9 + 273} - 5.028\ln(23.9 + 273)\right) = 2.948$$

Partial pressure at saturation for the exhaust gases, in [kPa]:

$$P_{s2} = EXP\left(52.58 - \frac{6790.5}{T + 273} - 5.028\ln(T + 273)\right)$$

$$P_{s2} = EXP\left(52.58 - \frac{6790.5}{23.9 + 273} - 5.028\ln(23.9 + 273)\right) = 2.948$$

Partial pressure for the moist inlet air, in [kPa]:

$$P_{w1} = \frac{\%RH_{inlet}}{100} * P_{s1} = \frac{25}{100} * 2.948 = 0.737$$

Partial pressure for the moist air exhaust, in [kPa]:

$$P_{w2} = \frac{\%RH_{exhaust}}{100} * P_{s2} = \frac{100}{100} * 2.948 = 2.948$$

Humidity ratio for the inlet air, in [kg water vapor/kg air]:

$$W_1 = 0.622 * \frac{P_{w1}}{(P_{amb} - P_{w1})} = 0.622 * \frac{0.737}{(101.3 - 0.737)} = 0.005$$

Humidity ratio for the exhaust gases, in [kg water vapor/kg air]:

$$W_2 = 0.622 * \frac{P_{w2}}{(P_{ambient} - P_{w2})} = 0.622 * \frac{2.948}{(101.3 - 2.948)} = 0.0186$$

Enthalpy of the inlet air, in [kJ/kg dry air]:

$$h_1 = 1.006 * T_{amb} + W_1 * (2501 + 1.805 * T_{amb})$$

$$h_1 = 1.006 * 23.9 + 0.005 * (2501 + 1.805 * 23.9) = 36.76$$

Enthalpy of the exhaust air, in [kJ/kg dry air]:

$$h_2 = 1.006 * T + W_2 * (2501 + 1.805 * T)$$

$$h_2 = 1.006 * 23.9 + 0.0186 * (2501 + 1.805 * 23.9) = 71.364$$

APPENDIX J - CHAPTER 6 CALCULATIONS EXAMPLE

Continues in the next 2 pages

COMPOST MODEL - CHAPTER 6

Treatment: **YC2** Updated: **11-Feb-03**

INPUTS

Time step	3	min
HRT	7	days

k empirical solid	0.100	1/d
k empirical oil	0.106	1/d
alpha solids	0.50	
alpha oil	0.06	
k20 solid	0.050	1/d
k20 oil	0.006	1/d
C1	1.07	
C2	1.21	

Initial mass total	1.65	kg ww
Initial mc	55	% ww
Initial FAS	32	%
Initial % Oil	35	% ds
Beta solids	22	% ds
Beta oil	48	% oil

INPUTS

U	0.03	W/m2.C
As	1.65	m2
Cp compost	0.00065	MJ/kg.C
Cp water	0.00418	MJ/kg.C
Cp oil	0.00190	MJ/kg.C

	Yard trimmings	Oil
C (a)	27	18
H (b)	38	33
O (c)	16	2
N (d)	1	0
Mole Wt	632	281

T amb	23.9	C
T set point	65	C
Heat comb Oil	38.33	MJ/kg
Heat comb YT	15.1	MJ/kg

Airflow (Std)	0.53	lpm
ma 33%	0.0007	kg air
ma 100%	0.0020	kg air
RH air in	25	%
RH air out	100	%

OUTPUTS

Mass biodegraded	**0.1083**	kg ww
% Mass degraded	**15**	% of initial mass
Solid degraded	**0.1326**	kg solids
Oil degraded	**0.0268**	kg oil

Total heat produced	**2617**	kJ
Heat per unit mass	**24156**	kJ/kg VS
T max	**75**	C
T average	**51**	C
Area under curve	**4474**	C.hr
final mc	**37.2**	% ww

Time t [day]	Temp T(t) [deg C]	Tot mass M(t) [kg ww]	M Solids Ms(t) [kg ds]	M Water Mw(t) [kg ww]	M Oil Mo(t) [kg oil]	M BVS BVS(t) sol [kg ds]	M BVS BVS(t) oil [kg oil]	%MC mc(t) [%]	km, sol with T effect [1/d]	km, oil with T effect [1/d]	F1 mc effect [decimal]	FAS [%]	F2 FAS [decimal]
0.000	23.9	1.65	0.342	0.908	0.401	0.075	0.192	55.0	0.0645	0.0075	0.935	32.0	0.983
0.002	23.9	1.65	0.342	0.907	0.401	0.075	0.192	55.0	0.0646	0.0076	0.935	32.0	0.983
0.004	24.0	1.65	0.342	0.907	0.401	0.075	0.192	55.0	0.0648	0.0076	0.935	32.0	0.983
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6.99	26.5	1.01	0.275	0.376	0.359	0.009	0.150	37.2	0.0762	0.0089	0.383	43.6	0.999
7.00	26.5	1.01	0.275	0.376	0.359	0.009	0.150	37.2	0.0762	0.0089	0.383	43.6	0.999

APPENDIX J - CHAPTER 6 CALCULATIONS EXAMPLE...Cont.

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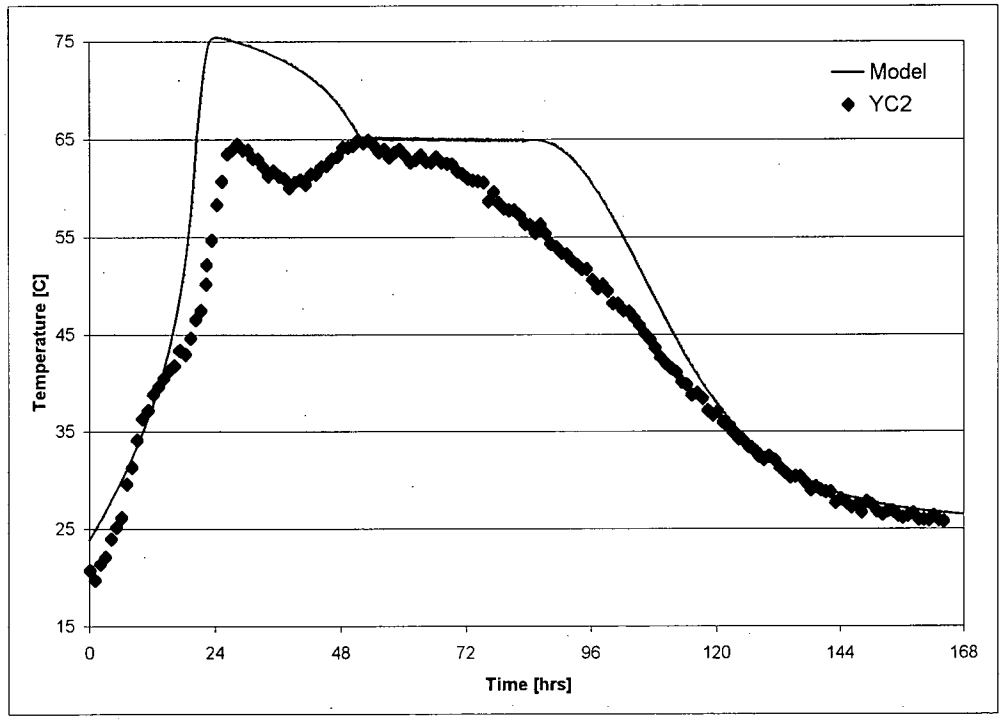

k solids [1/d]	k oil [1/d]	BVS used solids [kg ds]	BVS used oil [kg ww]	Heat Pro [MJ]	Heat loss Convec [MJ]	Aeration mass [kg]	Heat loss exhaust [MJ]	Heat Acc [MJ]	Wat prod [kg]	Water loss [kg]	M solids (Ms t+dt) [kg ds]	M water (Mw t+dt) [kg ww]	M oil (Mo t+dt) [kg oil]
0.059	0.007	9.275E-06	2.779E-06	2.466E-04	0.000E+00	0.00066	2.354E-05	2.230E-04	7.560E-06	9.252E-06	0.3415	0.91	0.401
0.059	0.007	9.301E-06	2.787E-06	2.473E-04	4.160E-07	0.00066	2.366E-05	2.232E-04	7.582E-06	9.288E-06	0.3415	0.91	0.401
0.060	0.007	9.328E-06	2.796E-06	2.480E-04	8.323E-07	0.00066	2.378E-05	2.234E-04	7.604E-06	9.323E-06	0.3415	0.91	0.401
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0.029	0.003	5.494E-07	1.068E-06	4.922E-05	2.332E-05	0.00066	3.077E-05	-4.875E-06	1.402E-06	1.139E-05	0.2755	0.38	0.359
0.029	0.003	5.493E-07	1.067E-06	4.921E-05	2.330E-05	0.00066	3.077E-05	-4.861E-06	1.402E-06	1.139E-05	0.2755	0.38	0.359

APPENDIX J - CHAPTER 6 CALCULATIONS EXAMPLE...Cont.

M total (M t+dt) [kg ww]	% MC (mc t+dt) [%]	Mavg*Cp each comp [MJ/C]	Temp (T t+dt) [deg C]	Psycrometric for air inlet & outlet							
				Ps1 [kPa]	Ps2 [kPa]	Pw1 [kPa]	Pw2 [kPa]	W1 [kg w/kg	W2 [kg w/kg	h1 [kJ/kg]	h2 [kJ/kg]
1.65	55.0	0.0048	23.93	2.948	2.948	0.737	2.948	0.005	0.019	35.620	71.452
1.65	55.0	0.0048	23.97	2.948	2.956	0.737	2.956	0.005	0.019	35.620	71.638
1.65	55.0	0.0048	24.02	2.948	2.964	0.737	2.964	0.005	0.019	35.620	71.824
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.	.	.	.	.	.	.	.	.	.	.	.
.	.	.	.	.	.	.	.	.	.	.	.
1.01	37.2	0.0024	26.50	2.948	3.445	0.737	3.445	0.005	0.022	35.620	82.464
1.01	37.2	0.0024	26.49	2.948	3.444	0.737	3.444	0.005	0.022	35.620	82.455

APPENDIX K
ECONOMICS OF COMPOSTING GREASE TRAP SLUDGE

Economics of Grease Trap Sludge Treatment

The economic comparison of treating grease trap sludge either by wastewater treatment (as currently done) and by aerobic composting was based in three categories: (1) treatment process efficiency, (2) processing cost, and (3) value and impact of end product/by products generated.

The efficiency of the treatment process is based on the process capability of degrading the grease trap sludge. Costs analysis includes processing cost, avoided costs, and revenues generated by end product (compost) sales. Processing costs does not include collection and transportation costs, because it is estimated that these costs will be about the same for both treatment alternatives.

The purpose of the treatment process sustainability analysis is an attempt to include the advantages or disadvantages of the treatment process mainly on an environmental basis.

The economics of grease trap sludge treatment were analyzed under three different categories; details are as follows:

1. Treatment Process Efficiency:

The average treatment efficiency for lipid materials at the wastewater treatment plant (Iona Island) is about 50% (GVRD 1999). For my experiments, the degradation efficiency of GTS was about 77%, for the high-degradation phase and the curing phase of composting studied (7 days and 126 days, respectively).

According to the literature reviewed, longer periods of composting would lead to higher degradation rates. According to the new Provincial Regulation (BCMWLAP, Organic Matter Recycling Regulation, B.C. Reg. 18/2002), there should be a minimum composting time of 14 days plus 21 days of curing, this longer composting time will likely result in more GTS degraded, thus the efficiency of GTS degradation during composting is expected to exceed the current efficiency at the wastewater treatment plant.

2. GTS Processing Costs:

The price charged for disposal of this waste at the wastewater treatment plant is calculated to be a full cost recovery price including both capital costs and operation and maintenance cost. This cost is currently set at $61.62 per m^3 (non-domestic liquid waste).

According to the BC Ministry of Agriculture, Food and Fisheries (1996), composting cost ranged from $36.00 to $70.00 per tonne of material treated. The composting facility (turned windrow system) located at the Vancouver Landfill reported a cost of yard trimmings composting of $40.45 per tonne for 2000 (City of Vancouver 2000). The tipping fees charged at the Vancouver Landfill for disposing yard trimmings is, on average, $42.50 (rate changes from $35.00 to $50.00 per tonne depending of weight disposed) (City of Vancouver 2002).

Assuming that treating grease trap sludge at the Vancouver Landfill composting facility will not demand major operational or equipment changes, then an estimate could be made using the tipping fees for the wastewater treatment plant and the composting facility, meaning a tipping fee of $61.62 per tonne (assuming water density) vs. $42.50 per tonne at the composting facility. This ballpark estimation shows an economic competitive difference if GTS were to be treated at the yard waste composting facility.

Another component of the processing cost estimation is the 'avoided costs'. Avoided costs are usually included in composting plant economic estimates as the savings produced by 'avoided' tipping fees for landfill disposal. In the case of composting grease trap sludge, the avoided costs would have three components:

1. Avoided disposal charges at the wastewater treatment plant.
2. Avoided sewer maintenance and cleaning costs needed due to clogging. Wastewater treatment process upsets due to the presence of GTS, and reduced treatment efficiency (adding the wastewater scum to the mesophilic digesters) due to the inhibition in anaerobic digestion as a result of high concentrations of long-chain fatty acids.
3. Avoided environmental costs due to a higher processing efficiency during composting, as compared to discharging the effluent of a relatively low treatment process into the ocean.

The last component to be considered for the processing cost estimation is the possibility of generating revenues out of the treatment process product. In the case of wastewater treatment of GTS, the only product related might be biosolids, which might have a market value.

In contrast, in the case of composting the end product might be sold (revenue), and also since the addition of grease trap sludge to yard waste to composting seemed to have a 'conservation' effect on the composting mass (more GTS was degraded versus the overall volatile solids), thus it might be possible to produce 'more' compost when adding GTS, as compared to not using GTS.

3. Treatment Process Sustainability

There are advantages and disadvantages in both treatment processes. For the wastewater treatment the GTS is relatively easy to manage since it is in liquid form, and the excess liquid might produce undesirable results in a compost facility (e.g. excess moisture content, leachate production).

Odor production might be a cause of concern for composting grease trap sludge in open windrows. However, according to Alpert et al. (2001) the only source of odor when using grease trap sludge was associated with the storage of this material. GTS was no longer odorous when combined with the biosolids and bulking agent. In addition, stricter environmental regulations point at the direction of enhanced odor controls in practically all waste treatment operations.

Regarding the potential benefits of producing compost using grease trap sludge, it has been reported that the presence of lipids in soil has a positive effect on soil aggregation and aggregate stability, and a negative effect on water retention. Furthermore, lipid content appears to be high in soils rich in humus, and inversely, low in soils poor in humus.

Plante and Voroney (1998) found that adding oily food waste to soils resulted in increased soil microbial biomass, and this increased the soil wet aggregate stability. Structural stability is a determining factor in soil aeration, water infiltration, and water retention, all of which are important to plant growth.

Forest soils have lipid content between 1.6-5.3 g per 100 g of oven dry soil. British Columbia soils have lower lipid content compared with forest soils with a concentration between 0.24-1.3 g per 100 g of oven-dry soil (Dinel et al. 1990). The

addition of soil conditioners rich in lipid compounds might be of benefit for British Columbia soils

Conclusion

The economics estimation of treating GTS under two different alternative systems, suggest there are improved economical and environmental benefits when treating GTS by aerobic composting. This is mainly due to the 'avoided' costs (wastewater treatment cost and maintenance avoided), and to the potential beneficial and sustainable environmental impacts of using GTS in composting (reduced NH_3 emissions, more compost product generated, and benefits to soils when adding compost rich in lipid materials).

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