UNIVERSITY OF SOUTHAMPTON

FACULTY OF ENGINEERING, SCIENCE AND MATHEMATICS
School of Civil Engineering and the Environment

Evaluation of Methods to Determine Biodegradable Municipal Waste Diversion from Landfill

by

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ABSTRACT

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EVALUATION OF METHODS TO DETERMINE BIODEGRADABLE MUNICIPAL WASTE DIVERSION FROM LANDFILL

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The biological tests of biochemical methane potential (BMP) and dynamic respiration index (DRI) are recommended by the EA monitoring guidance (Environment Agency, 2005) to evaluate the performance of Waste Disposal Authorities (WDAs) in diverting biodegradable municipal waste (BMW) from landfills by means of mechanical and biological treatment (MBT). Because the biological tests are complex and time-consuming to conduct, chemical tests may provide rapid surrogate measurement of biodegradability. The relative contents of the cellulose, hemicellulose and lignin determined by fibre analysis have been used to assess the degree of decomposition in landfills. However, their relationship with biodegradability has not been well investigated and understood, particularly for mixed BMW.

In this study, BMP and DRI, gravimetric test and chemical tests (fibre analysis and TC, TN) were conducted in parallel using a variety of MBT waste samples collected from anaerobic and aerobic degradation (composting) experiments in which real MSW or BMW was used. Two laboratory scale composting reactors were built where four batches of composting experiments were conducted.

The results of these tests were compared to identify the correlations between them, especially the correlations between the biological tests and fibre analysis. Evaluations are made based on this study for tests of BMP, DRI4, gravimetric test and TC/TN ratio. The main findings are: (1) Based on the good correlation observed between fibre analysis and biological tests for the treated waste, three linear model equations are proposed to predict the biodegradability upon (C+H)/L ratio or C/L ratio for the BMW during or after the MBT process, thereafter to predict the amount of BMW diverted by MBT in terms of anaerobic biogas potential; (2) The cellulose and hemicellulose contents or the (C+H)/L ratio of the untreated BMW (fresh mechanically pretreated BMW) were not found to correlate directly with biodegradability. This is consistent with the former findings on the specific waste components (untreated waste) by Eleazer *et al.* (1997) and Godley *et al.* (2005). It is suggested that untreated BMW and treated BMW need to be investigated separately in examining the relationships between relative fibre content and biodegradability.

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DECLARATION OF AUTHORSHIP

I, Bing Zheng,

declare that the thesis entitled *Evaluation of methods to determine biodegradable municipal waste diversion from landfill* and the work presented in the thesis are both my own, and have been generated by me as the result of my own original research. I confirm that:

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- parts of this work have been published as:

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ABBREVIATIONS

AD: Anaerobic Digestion

ADF: acid detergent fibre

ADL: acid detergent lignin

BMP: anaerobic biochemical methane potential test

BMW: biodegradable municipal waste

BOD: biological oxygen demand

(C+H)/L: cellulose plus hemicellulose to lignin ratio

C/L: cellulose to lignin ratio

COD: chemical oxygen demand

DEFRA: Department for Environment, Food and Rural Affairs

DM: dry matter

DRI: dynamic respiration index

EA: Environment Agency

LATS: Landfill Allowance Trading Scheme

LOI: loss of ignition

MBT: mechanical biological treatment

MSW: municipal solid waste

NDF: neutral detergent fibre

RDF: energy-rich refuse derived fuel

RLS: rate limiting step

TC/TN: total carbon to total nitrogen content ratio

TOC: total organic carbon

US EPA: United States Environmental Protection Agency

VFAs: volatile fatty acids

VOCs: volatile organic carbon compounds

WDAs: Waste Disposal Authorities

CHAPTER 1 INTRODUCTION

1.1 Background

Municipal solid waste (MSW) is made up of household waste (domestic waste) and other wastes collected by waste collection authorities or its contractors, such as municipal parks waste, beach cleansing waste and any commercial or industrial waste for which the collection authorities take responsibility (Waste Strategy for England 2007). About 29.1 million tonnes of municipal waste were collected in England in 2006/07 (DEFRA MSM survey, 2007). The best overall estimates available up to now show that household waste was estimated to be 89% of the total, of which 20% was garden waste, 18% paper and board, 17% kitchen waste, glass 7%, textile 3%, etc. (Strategy Unit, 2002). Paper/board, yard and food waste can be classified in a broad category known as organic or biodegradable municipal waste (BMW), which is estimated to be 60-70% of municipal waste in England (Guidance on LAS, 2006).

Landfill has been the dominant MSW disposal method. The proportion of municipal waste being disposed of at landfill in England was about 62% (17.9 million tonnes) in 2005/06 and 58% (16.9 million tonnes) in 2006/07 (DEFRA MSM survey, 2007). Landfilling untreated MSW results in long-term methane and leachate emissions, causing adverse ecological effects and giving rise to high costs for landfill aftercare (Strategy Unit, 2002). In order to prevent or reduce the negative environmental impacts of landfills, the EU Landfill Directive (1999/31/EC) requires that the BMW that may be disposed of in landfills be diverted from landfills in large and increasing quantities. Under the EU Landfill Directive, in the UK the volume of BMW sent to landfill must be reduced to 75% of the total amount (by weight) of BMW landfilled in 1995 by 2010, to 50% of 1995 levels by 2013, and to 35% of 1995 levels by

2020 (Strategy Unit, 2002). The typical flow of BMW is illustrated schematically in Figure 1.1. The diversion of BMW can be achieved by treatment techniques (e.g. composting) prior to the MSW disposed into landfills.

Mechanical biological treatment (MBT) of MSW is increasingly being applied in the UK prior to landfilling as it helps in reducing the amount of BMW disposed in landfill, and thus assists Waste Disposal Authorities (WDAs) meeting the Landfill Directive BMW diversion targets (CIWEM PPS, 2006; Environment Agency, 2007). Options for MSW biological treatment are anaerobic (anaerobic digestion), aerobic (composting) or a combination of the two. One major objective of either anaerobic or aerobic biological pre-treatment of MSW prior to landfilling is to degrade most of the organic components of MSW in a short period (such as 3 to 6 weeks) under controlled conditions so as to reduce the biodegradability of the waste stream for final disposal to landfills (Komilis *et al.*, 1999; Soyez and Plickert, 2002). For example, the degradation can reach approximately 50% in terms of the BMW amount in 4-6 weeks in a tunnel composting system (Müller and Bulson, 2005). As a consequence, future landfill needs will increasingly focus on the acceptance of MSW residues after the waste treatment processes (Strategy Unit, 2002).

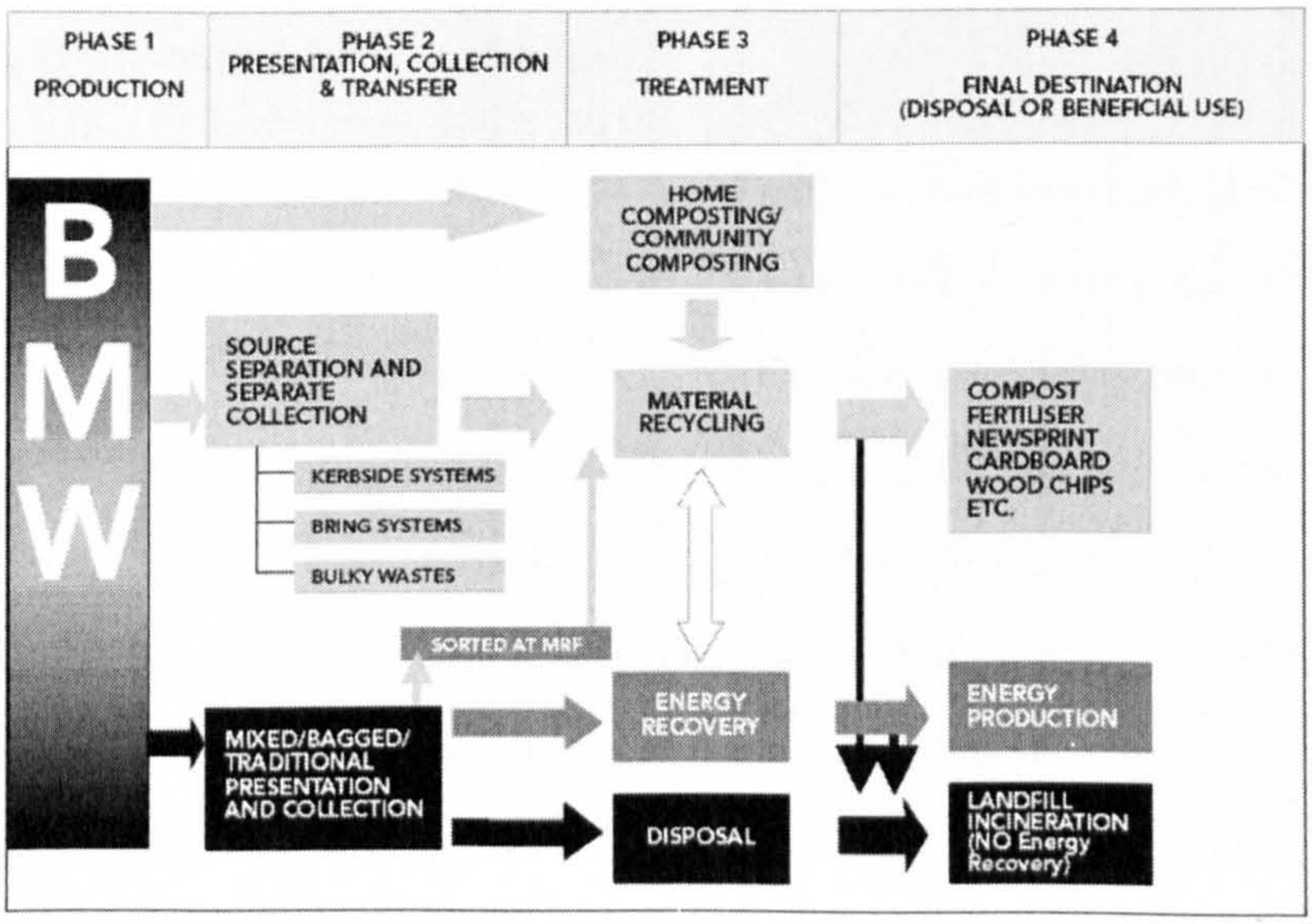


Figure 1.1. Summary flow chart for BMW diversion (Crowe *et al.*, 2002, European Environment Agency)

1.2 Research Justification

The Waste and Emissions Trading Act (2003) provides the legal framework for the allocation of tradable landfill allowances to each WDA in England for the purpose of progressive diversion of millions of tonnes BMW from landfills by the year 2020. The Landfill Allowance Trading Scheme (LATS) was launched in April 2005 which set allowances for WDAs to reduce the amount of BMW sent to landfill, thereby to enable England to meet its BMW diversion targets under the UK targets. The Environment Agency (EA) is responsible for monitoring the LATS and the performance of the WDAs in meeting the targets for diverting BMW from landfill. Therefore, for the waste subjected to MBT or other treatments prior to landfilling, characterization methodologies are needed to determine the contribution of the treatments processes, i.e., the amount of BMW diverted from landfill, the extent to which readily biodegradable organic matter has decomposed, as well as the amount of BMW sent to landfills. This may be achieved by determining the change in biodegradability between the input MSW and its MBT output, or by directly measuring the degradable fraction prior to and then following treatment (Brauer *et al.*, 2005; Environment Agency, 2007).

Godley *et al.* (2005) assessed a number of physical, chemical and biological test methods (Table 1.1) to determine their suitability in providing a measure of the biodegradability of waste. They recommended that biodegradability measured using the anaerobic biochemical methane potential (BMP) and the aerobic dynamic respiration index (DRI) methods showed the greatest promise in providing a reliable and consistent measure of the biodegradability potential of organic wastes. Of the non-biological methods evaluated, it was suggested that gravimetric determination of dry matter (DM) and loss of ignition (LOI) contents is basic and essential because LOI generally represents the total organic solid of the test material and the biodegradability is recommended to report on the basis of LOI or DM of the test

material. The chemical tests were found to provide some useful information on the waste composition, but more investigation work and evaluation were required to find out if these tests can provide a reliable indication of the waste biodegradability.

Table 1.1. Test methods evaluated by Godley *et al.* (2005)

Parameter
Moisture content, dry matter content, Loss on ignition, ash content
Total organic carbon and total nitrogen
ADF, cellulose, lignin
Water leachable dissolved organic carbon (DOC), chemical oxygen demand (COD) and biochemical oxygen demand (BOD)
Dynamic Respiration Index (DRI)
Specific oxygen uptake rate (SOUR)
Anaerobic BMP
Cellulase hydrolysis

In the guidance on monitoring the MBT processes (Environment Agency, 2005), the 100 day anaerobic biogas potential test (BM100) and the 4 day aerobic dynamic respiration test (DRI4) are specified for the monitoring purpose. However, as biological tests BMP and DRI tests involve the use of inoculum and optimal incubation conditions, which may not be easy to control to obtain consistent effects. BMP test is time-consuming to conduct, requiring several weeks or more before results are obtained. DRI test is relatively rapid, but in some cases it underestimate the overall biodegradation potential in a short test time, such as 4 or 7 days (Godley *et al.*, 2005, 2007a). According to the 'Consultation on Revised Guidance for Monitoring MBT Plant Under Landfill Allowance Schemes' (Environment Agency, 2007), the estimated monitoring cost using BMP or DRI test is about 250 to 300

English pounds per test, excluding the sample preparation cost. Due to their extended time requirements and/or expense, there is a need to look for a rapid chemical method which is of lower cost and able to provide a surrogate assessment of BMW biodegradability (Brauer, *et al.*, 2005; Environment Agency, 2007; Wagland, *et al.*, 2007).

Cellulose and hemicellulose are the major organic biodegradable components of MSW and lignin is also a major organic compound commonly found in MSW. The cellulose plus hemicellulose fraction was found making up for over 90% of MSW methane potential in landfills (Bookter and Ham, 1982; Barlaz *et al.*, 1989b). As cellulose and hemicellulose are slowly degradable materials and lignin is difficult to degrade, their relative contents (such as, the cellulose to lignin ratio) have been used to assess the degree of decomposition in landfills at various stages (Bookter and Ham, 1982; Wang *et al.*, 1994; Baldwin *et al.*, 1998) and the changes in the cellulose, hemicellulose and lignin contents were reported to be a good index of the degree of maturity for a compost (Keller, 1961; Komilis and Ham, 2003). Therefore, the determination of the waste cellulose, hemicellulose and lignin content may provide a non-biological test of assessing biodegradability.

There were some analytical methods of fibre analysis, the determination of cellulose, hemicellulose and lignin contents. Van Soest (1963a, b, 1967) developed the procedures of fibre analysis, which make use of chemical detergents and three tests (acid detergent fibre (ADF), neutral detergent fibre (NDF) and acid detergent lignin (ADL)) to partition substrate constituents, which are cell soluble matter, cellulose, hemicellulose and lignin. These procedures were used for estimating feedstuff digestibility with great success. Chandler et al. (1980) first investigated whether the developed by Van Soest (1963a, b, 1967) may aid the evaluation of substrate biodegradability and found that the biodegradable fraction of organic solids in anaerobic conditions was directly proportional to the lignin contents of the studied material. Kitcherside *et al.* (2000) introduced the FibreCap system, an improved procedure based on Van Soest's procedures for the analysis of fibre in samples of

feeding stuffs, which has been demonstrated to produce higher quality data. This procedure of fibre analysis is quicker in operation (e.g. 1 hour for digestion in acid detergent in ADF test) and more repeatable than conventional methods. However, this procedure has not been used to characterize the composition of MSW samples in the relevant published researches, such as, Wang *et al.* (1994), Eleazer *et al.* (1997) and Godley *et al.* (2005). It is worth investigating, as a cost-effective and rapid chemical analysis method, for the purpose of assessment of the waste biodegradability and further determination of BMW diversion from landfill.

On the one hand, in the relevant reported studies on the relationship between cellulose, hemicellulose and lignin contents and the biodegradability, the specific MSW components were used for investigation and, to date, no reliable quantitive correlation was identified between the biodegradability and the cellulose plus hemicellulose to lignin ratio ((C+H)/L) (Barlaz *et al.*, 1997; Eleazer *et al.*, 1997) or the cellulose and lignin content (Godley *et al.*, 2005). The possible reasons are (1) the procedure or technique used for the fibre measurements, which can be laborious and involve several filtration processes that may introduce errors through the loss of the retained mass of the waste sample (Kitcherside *et al.*, 2000); (2) studying the specific waste components instead of the real mixed MSW.

On the other hand, few attempts have been made to identify the correlation between the change of fibre contents and the degree of stability of MSW during (and after) MBT. As far as BMW diversion is concerned, the investigation of whole BMW biodegradation, especially the biologically treated BMW, is of more significance than investigation of the untreated individual MSW component. Therefore, in this research, real mixed MSW or BMW before, during and after MBT were used to investigate the relationship between fibre contents of waste and the corresponding biodegradability, on which no research or results had been reported. This experimental investigation aims to identify fibre analysis, a rapid and cost- effective analysis method, for assessment of the waste biodegradability, thereby to reduce the

cost of monitoring BMW diversion performance, compared to the cost of BM100 or DRI4.

1.3 Research Objectives

The emphasis of this study is to investigate whether the use of the fibre analysis method can be a surrogate for traditional biological tests (BMP and DRI) in the characterization of the MSW biodegradability, and in the assessment of the degree of degradation which takes place during MBT prior to that waste being landfilled. The specific objectives include:

- To track the change of cellulose, hemicellulose and lignin contents in BMW during biodegradation;
- To investigate any correlation between the tests of BMP, DRI and fibre analysis. Thereby to find out whether the fibre analysis, as a rapid chemical test, could provide a surrogate for measurement of biodegradability;
- To identify the correlation between the change in the contents of cellulose, hemicellulose and lignin and the degree of stability of BMW following MBT.
- To evaluate the tests used in this study.

1.4 Structure of Thesis

This thesis consists of eight chapters. Chapter 2 presents an overview of the main MSW disposal options in the UK, including landfill and MBT. It reviews the processes and operation of MSW degradation in landfill, mechanical and biological treatment of MSW, as well as the roles they play under the EU Landfill Directive.

Chapter 3 reviews MSW characterization in terms of waste composition, organic constituents, and biodegradability. A series of essential and/or promising tests for

determining the waste composition and the stability of treated waste are also reviewed. The non-biological tests include loss on ignition, carbon to nitrogen ratio, and fibre analysis. The two biological tests are the biochemical methane potential test and dynamic respiration index test, which had been proposed for testing waste degradability. These tests are the principle methods used and evaluated throughout this study.

Chapter 4 presents the results from a preliminary experimental study on the anaerobic degradation of both the mechanically and aerobically treated MSW, particularly using the FibreCap method. The biodegradation of test waste material was quantified by the biogas yield and qualified through the ratio of cellulose plus hemicellulose to lignin ((C+H)/L ratio) of the non-degraded material. The correlation between the change of (C+H)/L ratio and biogas production was also studied. Additionally, the composition data on the mechanically and/or biologically treated MSW and its components is presented.

Chapter 5 explains the research methodology used, including fibre analysis by FibreCap method, BMP test and DRI measurement in parallel using a variety of fresh BMW and aerobically treated BMW samples. It describes the setup and operation of the laboratory-scale composting reactors which are used to simulate the aerobic treatment process for BMW and allow the DRI to be measured during composting. The sampling procedures, methods of analysis and tests, and the material used are also presented.

In Chapter 6, the operation and related tests results of the four batches (B1-B4) composting experiments are described and discussed. The results are presented in terms of temperature and respiratory activity changes, the change of solid composition during composting and the resultant reduction in biogas production potential after aerobic treatment. The composting processes and waste degradation in all the four batches are also compared and summarized in this chapter.

In Chapter 7, the results from the tests in the four batches are compared to determine any correlations between the tests. In particular, the results of fibre analysis are correlated with those of the biological tests, including anaerobic biogas potential and oxygen consumption.

Chapter 8 summarizes the conclusions drawn from the experimental results and discussions in this study, as well as presents future research recommendations.

CHAPTER 2 MSW DISPOSAL OPTIONS

Landfill has been the dominant MSW disposal method in the UK. The EU Landfill Directive requires that the volume of biodegradable municipal waste (BMW) disposed in landfills is reduced gradually in the following years. This will affect the waste management systems in the UK because of its reliance on landfilling for waste disposal (Strategy Unit, 2002). Mechanical and biological treatment (MBT) is playing an increasingly important role in the reduction of the BMW sent to landfill. This Chapter reviews the processes and operation of MSW degradation in landfills, mechanical and biological treatment, as well as the roles they play in waste management.

2.1 Disposal of MSW in Landfill

2.1.1 Biochemical and Microbial Aspects of MSW Degradation in Landfill

When waste is disposed in a landfill, as oxygen is present in the void space in waste, aerobic decomposition occurs first and the easily biodegradable organic materials react quickly with oxygen to form simpler hydrocarbons, carbon dioxide, water and other by-products (e.g. bacterial cells). As oxygen becomes depleted and anaerobic microorganisms presents, the anaerobic decomposition phase starts, which is the dominant phase in the process of MSW degradation in landfill and of more significance from the perspective of biogas production (El-Fadel *et al.*, 1997).

From the biochemical and microbial point of view, in the anaerobic phase large organic molecules are broken down to simple polymers, which are ultimately converted into methane and carbon dioxide by the action of microorganisms. The general anaerobic degradation of solid waste can be described by means of Equation 2.1 (Tchobanoglous *et al.*, 1993). At the microscopic level hundreds of potential intermediary reactions and compounds are involved in the breakdown process of organic material (Evans, 2001). The main four stages of the anaerobic degradation include hydrolysis, acidogenesis, acetogenesis and methanogenesis (Mosey, 1983; Archer and Robertson, 1986), which are illustrated in Figure 2.1 and reviewed as following.

$$\text{Organic matter} + H_2O + \text{nutrients} \longrightarrow \text{new cells} + \text{resistant organic matter} + CO_2 + CH_4 + NH_3 + H_2S \quad (2.1)$$

Hydrolysis: In the first stage of hydrolysis, or liquefaction, complex and/or insoluble organic polymers, such as carbohydrates, cellulose, proteins and fats, are broken down and liquefied by the extracellular enzymes produced by hydrolytic bacteria. In general, these conversions in landfill include (Evans, 2001):

Lipids → Fatty Acids

Polysaccharides → Monosaccharides

Protein → Amino Acids

Nucleic Acids → Purines & Pyrimidines

These conversions make the original organic matter more easily available for use by the acidogenic bacteria of the next stage (Evans, 2001; Parkin and Owen, 1986). From a chemical point of view, hydrolysis means the breakdown of long-chain biomolecules under reactions with water. Biologically, hydrolysis works through the influence of enzymes (Fox and Pohland, 1994). The hydrolysis of complex compounds, e.g. cellulose, to simple, soluble substances is often the rate limiting step (RLS) in anaerobic degradation, because the stabilization and methane

fermentation of complex organics cannot occur unless this initial hydrolysis step is functioning properly (Noike *et al.*, 1985; Parkin and Owen, 1986; Boone *et al.*, 1993).The rate of hydrolysis is governed by substrate availability, bacterial population density, temperature and pH (Evans, 2001).

The major group of bacteria involved in this step is the hydrolytic bacteria (Barlaz, 1989a) which excrete the hydrolytic extracellular enzymes to break down complex and/or insoluble organic polymers into a size and form that can pass through bacterial cell walls for use as energy or nutrient sources.

Acidogenesis: This second stage of acidogenesis is characterised by production of acetic acid and other volatile fatty acids (VFAs) which are fermented from the product released in the preceding stage. The pH falls as the levels of these compounds increase. The by-products of carbon dioxide and hydrogen are also present as a result of the catabolism of carbohydrates. The proportion of the different by-products produced depends on the particular bacteria species present and the environmental conditions (Evans, 2001).

The population of bacteria responsible for acid production is facultative and/or obligate anaerobic bacteria that are often identified as 'acidogens' or 'acid formers' (Parkin and Owen, 1986; Tchobanoglous *et al.*, 1993). This group of bacteria ferments the breakdown products from the first stage to simple organic acids, such as acetic, propionic, butyric acid, lactic acid, alcohols and ammonia (from amino acids). For example:

$$C_6H_{12}O_6 + 2H_2O \longrightarrow 2CH_3COOH \text{ (acetic acid)} + 4H_2 + 2CO_2 \quad (2.2)$$

$$C_6H_{12}O_6 + 2H_2 \longrightarrow 2CH_3CH_2COOH \text{ (propionic acid)} + 2H_2O \quad (2.3)$$

$$C_6H_{12}O_6 \longrightarrow CH_3CH_2CH_2COOH \text{ (butyric acid)} + 2CO_2 + H_2 \quad (2.4)$$

Acetogenesis: In this third stage, the long-chain fatty acids from acidogenesis are further digested by acetogenic bacteria to produce carbon dioxide, hydrogen and

mainly acetic acid. The long-chain fatty acids are degraded by subtraction of carbon fragments. For example:

$$CH_3CH_2CH_2COOH + 2H_2O \longrightarrow 2CH_3COOH + 2H_2 \quad (2.5)$$

$$CH_3CH_2CH_2\ CH_2COOH + 2H_2O \longrightarrow CH_3COOH + CH_3CH_2COOH + 2H_2 \quad (2.6)$$

$$CH_3\ CH_2COOH + 2H_2O \longrightarrow CH_3COOH + CO_2 + 3H_2 \quad (2.7)$$

Methanogenesis: Methanogenesis involves the production of methane from a number of simple substrates (acetic acid, methanol or carbon dioxide and hydrogen) produced in the previous stage (Evans, 2001). Around 75% of the methane produced is derived from acetic acid and the closely related acetate according to the equation below:

$$CH_3COOH \longrightarrow CH_4 + CO_2 \quad (2.8)$$

Methanol may also be used as an energy source by methane-forming bacteria.

$$CH_3OH + H_2 \longrightarrow CH_4 + H_2O \quad (2.9)$$

Methane is also produced through carbon dioxide reduction using hydrogen as the energy source, by CO_2-reducing methanogens (Equation 2.10). In a landfill environment, methane generation in this way is often limited as most of the hydrogen is consumed by sulphate reducers (El-Fadel *et al.*, 1997).

$$CO_2 + 4H_2 \longrightarrow CH_4 + 2H_2O \quad (2.10)$$

The group of microorganisms responsible for the production of methane in this stage is called the methanogenic bacteria or methanogens, which are pH sensitive with the required range being mildly acidic (6.6-7.0) (Evans, 2001). If the initial fermentation in the landfill is too rapid, the increase in VFA concentration (and thus a decrease in pH) may inhibit the development of methanogenic bacteria and the required stabilization of the waste will not be achieved properly. In this sense, hydrogen produced in the preceding stage can be an indicator of the balance between hydrogen-producing acetogenic bacteria and the hydrogen-utilising

methanogens. An active population of hydrogen-utilising methanogens can ensure that the acid degradation is not inhibited (Parkin and Owen, 1986).

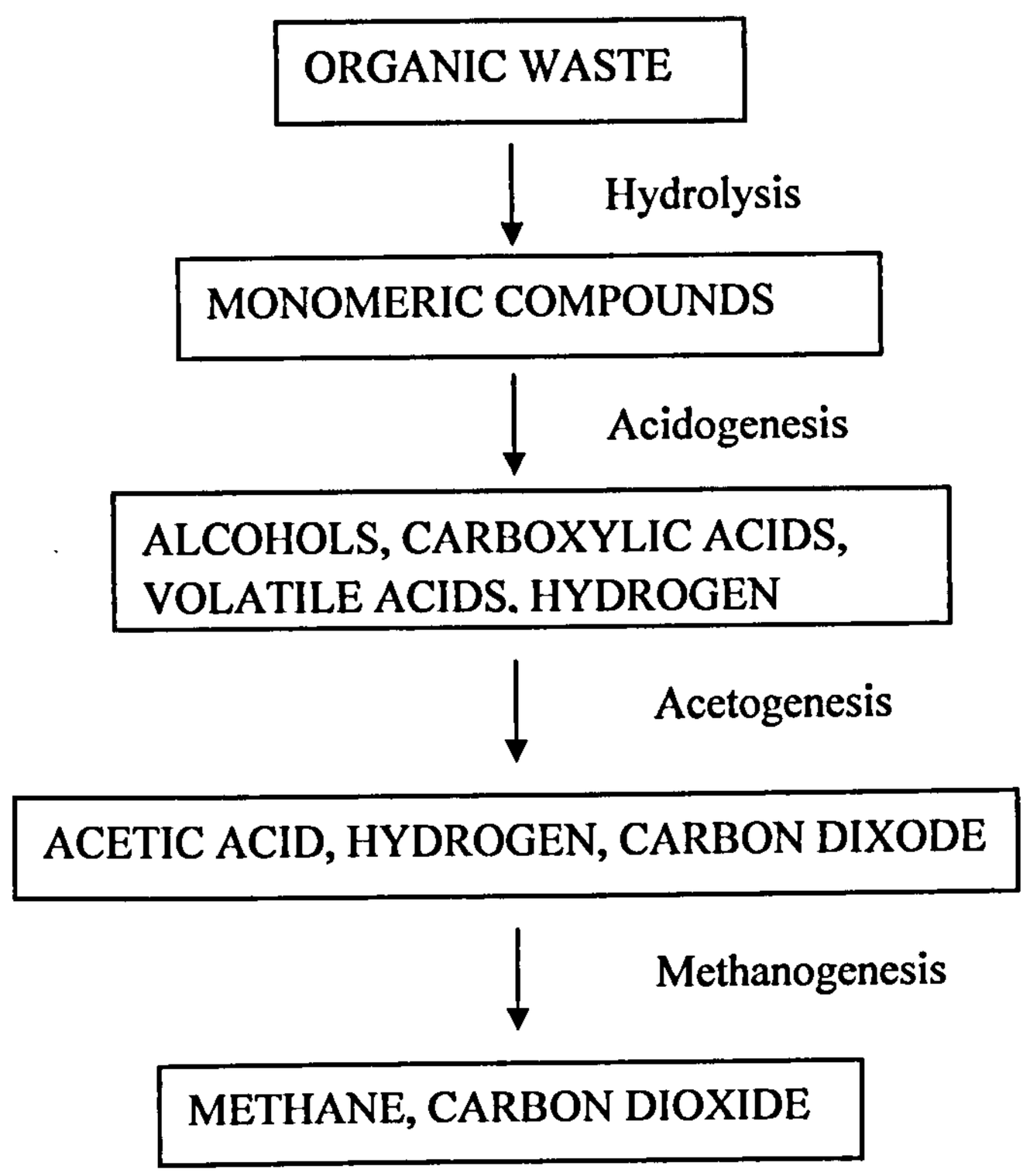


Figure 2.1 Major degradation steps during the anaerobic digestion phase (El-Fadel *et al.*, 1997)

2.1.2 Environmental Impact and Role of landfill

In general, the gas produced in landfills is composed of methane, carbon dioxide, water, and various trace components such as ammonia, sulphide, and non-methane volatile organic carbon compounds (VOCs) e.g. benzene and chlorobenzene (Vesilind *et al.*, 2002). The quality of landfill gas is highly dependent on the degradation stage within the landfill (Barlaz *et al.*, 1989a). Under a stabilized methanogenic condition, which is of interest in the aspect of a beneficial recovery of

methane, methane and carbon dioxide are the two principle components of landfill gas, consisting of more than 90% of the total gas generated (El-Fadel *et al.*, 1997). On one hand, the gas can pose an environment threat because methane is a powerful green house gas, and many of the VOCs are odorous and/or toxic. On the other hand, the gas has a high energy content of methane and can be used for power, steam, or heat generation if treated and collected properly (Vesilind *et al.*, 2002).

Leachate is formed in landfill as a result of the removal of soluble compounds by the percolation of water through the wastes. The characteristics of the leachate generated are highly dependent on the stage of degradation in the landfill, waste composition, operational procedures, climate and site hydrogeology (Harmsen, 1983; Vesilind *et al.*, 2002). Many chemicals (e.g. metals, aliphatics, acyclics, terpenes, and aromatics) have been detected in landfill leachate, which can move towards the groundwater and the surrounding environment (El-Fadel, *et al.*, 1997).

The environmental impacts of landfills arise primarily due to gas and leachate formation and were summarized by El-Fadel, *et al.*, (1997), which include fire and explosions, unpleasant odour, landfill settlement, groundwater pollution, local air pollution and global warming. Although methane gas is increasingly being captured from landfill sites (Strategy Unit, 2002), it is also reported that landfills are a major contributor to the production of the potent greenhouse gas methane, which will need to be progressively reduced over the period up to 2020 (DEFRA, 2004). Another issue concerning landfilling has arisen, as landfill sites are becoming increasingly scarce in the South East and North West of England due to other pressures on land use (Strategy Unit, 2002). Therefore, the volumes of waste sent to landfill need to be reduced.

To prevent or minimize the negative effects of landfilling, the European Union (EU) Landfill Directive (1999/31/EC) sets demanding targets to reduce the amount of BMW sent to landfill. The Landfill (England and Wales) Regulations 2002 came into force in June 2002 and implemented the Landfill Directive through measures

such as separate collection and recycling, or pre-treatment of MSW before landfilling (DEFRA, 2004; Pan and Voulvoulis, 2007). These measures contribute to reducing the amount of BMW which will be disposed of in landfill. However, they are not final disposal measures and there are still residues left after treatments, which can not be further reused or recycled economically. Therefore, landfilling is expected to continue to play an important role as part of an integrated solution of waste management, which will become increasingly focussed on processed MSW residues.

2.2 Aerobic Composting

2.2.1 Process Description

In the aerobic composting process, aerobic microorganisms extract energy from the organic matter through a series of exothermic reactions that break down the material to simpler and more stable materials. The basic aerobic degradation process can be expressed as following (Tchobanoglous *et al.*, 1993; Vesilind *et al.*, 2002):

$$\text{Complex organics} + O_2 \xrightarrow[\text{microorganisms}]{\text{Aerobic}}$$

$$\text{Compost} + CO_2 + H_2O + NO_3^- + SO_4^{2-} + \text{Heat} \qquad (2.11)$$

The composting process occurs in two major phases. In the first stage, microorganisms decompose the feedstock into simpler compounds, producing heat as a result of their metabolic activities. The size of the composting pile is reduced during this stage. In the second stage, the compost product is "cured" or finished. Microorganisms deplete the supply of readily available nutrients in the compost which, in turn, slows their activity. As a result, heat generation gradually diminishes and the compost becomes dry and crumbly in texture. When the curing stage is

complete, the compost is considered 'stabilized' or 'mature'. Any further microbial decomposition will occur very slowly (US EPA, 1994; Xi, *et al.*, 2005).

A rapid succession of mixed microbial populations is involved in the dynamic process of composting. The main groups of microorganism are bacteria, including actinomycetes, and fungi (Golueke, 1991). Different types of microorganisms are active at different phases of composting (US EPA, 1994). At the beginning of composting, mesophilic bacteria predominate. But after the temperature increases to over 40°C, thermophilic bacteria take over and thermophilic fungi also appear in the compost. When the temperature exceeds 60°C, microbial activity decreases dramatically. After the compost has cooled mesophilic bacteria and actinomycetes dominate again (McKinley and Vestal, 1985; Strom, 1985).

Composting is the most commonly used biological process for the conversion of the BMW to *compost*, a stable humus-like material (Tchobanoglous *et al.*, 1993). Humus is a material comprising many organic substances and represents a stable status following decomposition of organic materials. It is a semi-finite state of decomposition (the finite state is ash) and is characterized by a slow decomposition rate (Epstein, 1997). The general objectives of composting are (Tchobanoglous *et al.*, 1993):

- To stabilize the biodegradable organic material and reduce the original volume of waste;
- To destroy pathogens, insect eggs and parasites;
- To produce a product that can be used for soil amendment and to support plant growth.

2.2.2 Critical Parameters in Composting Operation

The microbiology of all aerobic composting processes is similar (Tchobanoglous *et al.*, 1993). Microbial activity is influenced by oxygen levels, particle size of the

feedstock, nutrient levels and balance (indicated by the carbon-to-nitrogen (C/N) ratio), moisture content, temperature, and acidity/alkalinity (pH) (US EPA, 1994). Among them, moisture content, C/N ratio and temperature are the critical parameters in the control of the composting processes (Tchobanoglous *et al.*, 1993). These factors and their interrelationships are reviewed briefly below.

Moisture Content: In compost, a majority of the micro-organisms typically grow in or below a liquid film on the surface of compost particles (Golueke, 1972). If the mixture is too dry the microbial activity would be seriously limited or the micro-organisms may not survive, and composting efficiently stops. If there is too much moisture present, the oxygen from the air is not able to penetrate to where the micro-organisms are, and the mixture becomes anaerobic (Vesilind *et al.*, 2002). The optimum moisture content in terms of the total mass is in the range 50 to 60 percent during the composting process. Moisture can be adjusted by blending components and by the addition of water (Tchobanoglous *et al.*, 1993). If the moisture level drops below about 40 to 45 %, the nutrients are no longer in an aqueous medium and easily available to the microorganisms. Their microbial activity decreases and the composting process slows. Below a moisture of 20 %, very little microbial activity occurs (Haug, 1980).

Temperature: During composting, temperature is both the cause and effect of the microbial activity (MacGregor *et al.*, 1981). Temperature determines the rate of microbial activity while at the same time it is a function of biologically-generated heat (Stringfellow, 1998).

Typically, a batch composting operation is characterized by two stages: (1) a short active phase (10-18 days) with a rapid rise in temperature, followed by cooling and (2) a longer maturation period (lasting up to 3-6 months) comprising slower degradation at low temperature (Figure 2.2, Gray *et al.*, 1971). As the reaction develops, the early decomposers are mesophilic bacteria followed after about a week

by thermophilic bacteria, actinomycetes, and thermophilic fungi (Golueke, 1972). In most well-operated composting operations the temperature can increase to about 70 °C. Above 70 °C, spore-forming bacteria predominate. As the decomposition slows, the temperature drops and mesophilic bacteria and fungi reappear (Vesilind *et al.*, 2002). The maximum temperature achieved in a composting pile depends on the type of microbes present and process variables such as aeration, particle size, heap volume, moisture content and nutrient availability (Gray *et al.*, 1971).

The temperature which results in the optimum rate of composting may vary depending on the substrate and microorganisms present (Stringfellow, 1998). Golueke (1972) found that the optimal temperature range for the composting process as a whole is broad, from 35 to 55°C, since a range of microorganisms are involved in the decomposition of organic matter. Bach *et al.* (1984) found that the optimal temperature for sewage sludge composting was around 60°C as observed from the CO_2 evolution rate.

Nutrients: Nutrients, especially carbon and nitrogen, play an important part in the composting process as they are essential for microbial growth and activity. Carbon is needed as the principle energy source, and nitrogen is needed for cell synthesis. Phosphorus, potassium and sulphur are also important for cell growth and metabolic processes, and trace elements such as copper, zinc, cobalt, manganese and iron are necessary for enzymatic functions, but little is known about their importance to the composting process (Epstein, 1997).

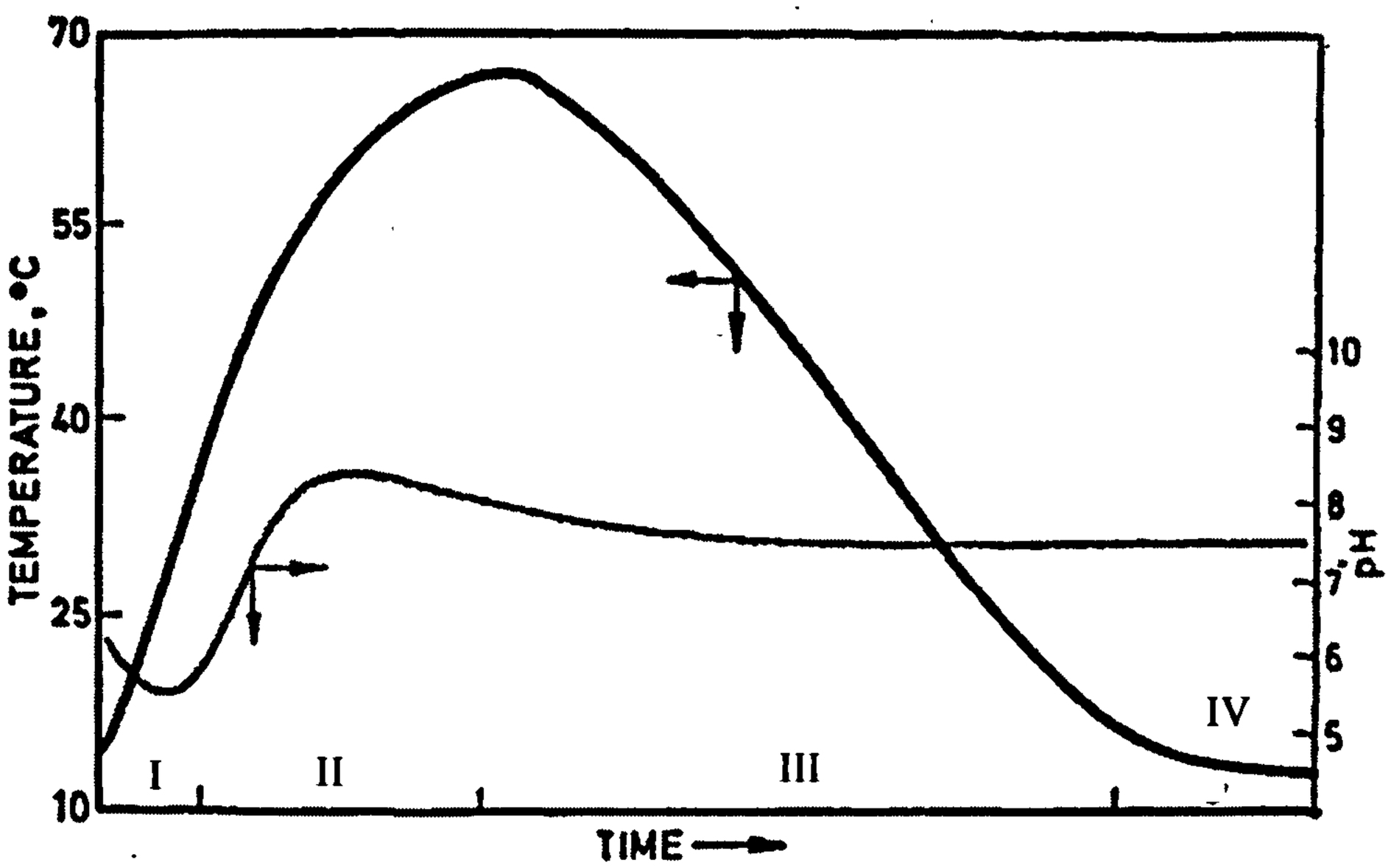


Figure 2.2. Temperature and pH variation with time indicating the phases of microbial activity (Gray *et al.*, 1971)

I = mesophilic, II = thermophilic, III = cooling, IV = maturing.

de Bertoldi *et al.* (1983) and Golueke (1991) showed that the practical optimum C/N ratio for composting is approximately 25 to 30. This range is obtained from theoretical determinations of the C/N ratio according to the microbial energy requirement and synthesis of new cells during active aerobic composting, and is confirmed by field experiments (Haug, 1993). A C/N ratio greater than 30 can increase the time to maturity. At C/N ratios greater than about 50:1, nitrogen can become limiting and the composting process slows due to the depletion of available nitrogen which results in the decrease of cellular growth. Low C/N ratio leads to nitrogen volatilization in the form of ammonia as microbial activity eliminates nitrogen which cannot be converted into protoplasm because of lack of energy-rich carbon (Epstein, 1997).

In general, all of the organic nitrogen present in most organic compounds will become available, whereas not all of the organic carbon will be biodegradable. Depending on the particular waste material, the C/N ratio computed on the basis of total carbon and nitrogen content can be quite misleading, especially in those cases

where all the available nitrogen is biodegradable, but only a portion of organic carbon is biodegradable e.g. lignin in waste paper (Kayhanian and Tchobanoglous, 1992). Blending of a waste high in carbon and low in nitrogen (e.g. newsprint) with a waste that is high in nitrogen (e.g. yard wastes) is often used to achieve optimum C/N ratios for composting (Tchobanoglous *et al.*, 1993).

Aeration: Air must be supplied to a composting material for three basic purposes: to satisfy oxygen demand for organic decomposition (stoichiometric demand), to remove heat (heat removal demand), and to remove water from the wet substrate (drying demand) (Haug, 1993). Microorganisms important to the composting process require oxygen to break down the organic compounds in the composting feedstock. Oxygen can be provided by mixing or turning the pile, or by using forced aeration systems. The amount of oxygen that needs to be supplied during composting depends on (US EPA, 1994):

- The *stage of the process* - Oxygen generally needs to be supplied in the initial stages of composting; it is usually not as needed during curing;
- *The type of feedstock* - Dense, nitrogen-rich materials (e.g. grass clippings) will require more oxygen;
- *The particle* size *of the feedstock* - Feedstock materials of small particle size (e.g., less than 1 or 2 inches in diameter) will compact, reducing void spaces and inhibiting the movement of oxygen;
- *The moisture content of the feedstock* - Materials with high moisture content (e.g., food scraps, garden trimmings) will require more oxygen.

The oxygen required for metabolism (stoichiometric demand) may be calculated from the composition of the organic fraction of the feedstock (Tchobanoglous *et al.*, 1993):

$$C_aH_bO_cN_d + \left(\frac{4a+b-2c+3d}{4}\right)O_2 \rightarrow aCO_2 + \left(\frac{b-3d}{2}\right)H_2O + dNH_3 \qquad (2.12)$$

Aeration is also widely used to restrict temperature rise and remove moisture by encouraging vaporisation (Stringfellow, 1998). In addition to the moisture already present in the sludge, more water is produced during degradation; for each gram of metabolised material, composting produces 0.5-0.6g of water (Miller, 1996). Air needs to be supplied at nine times the stoichiometric rate to achieve adequate loss of moisture and to control temperature (MacGregor *et al.*, 1981).

pH: Composting is a flexible system in many ways, and organic matter within a pH range from 3 to 11 can be composted (de Bertoldi *et al.*, 1983). However, most decomposition takes place between pH 5.5 and 9 (Rynk, 1992; Gray *et al.*, 197l). Bacteria prefer a pH between 6 and 7.5. Fungi thrive in a wider range of pH levels than bacteria, in general preferring a pH between 5.5 and 8 (Boyd, 1984). If the pH drops below 6, microorganisms, especially bacteria, will die off and decomposition rate will slow (Wiley, 1956). If the pH reaches 9, nitrogen is converted to ammonia and becomes unavailable to organisms (Rynk, 1992). This too will slow the decomposition process.

Figure 2.2 shows the progression of pH over time in a composting pile. During the start of the composting process, organic acids are formed and the composting materials usually become acidic with a pH of about 5. At this point, the acid-tolerating fungi play a significant role in decomposition. Microorganisms soon break down the acids, however, and the pH levels gradually rise to a more neutral range, or even as high as 8.5. The role of bacteria in composting increases in predominance again as pH levels rise. If the pH does not rise, this could be an indication that the compost product is not fully matured or cured (US EPA, 1994).

2.3 Practice and Role of MBT

2.3.1 Description of MBT Practice

Mechanical biological treatment (MBT) is primarily a volume-reducing process recovering recyclable materials from MSW and biologically treating the biodegradable component of the waste. MBT of waste has been used across the EU over the last 10 years or so, particularly in Germany and Austria (Assurre, 2006). It has risen to prominence in waste management over recent years because MBT can assist in meeting Landfill Directive targets for reduction of BMW sent to landfill under the Landfill Allowance Trading Scheme (LATS) (CIWEM PPS, 2006).

The term MBT is used generally to describe treatment systems consisting of a mechanical sorting system with an adjacent biological treatment facility (either aerobic or anaerobic, or a combination of the two). MBT is primarily dealing with the residues of mixed MSW once the dry-recyclable fraction (i.e. paper, card, plastics cans, glass etc. and to some extent garden waste) has been reduced and largely removed in some cases through separate collection systems from households. MBT systems can vary in terms of the combination and the degree of mechanical sorting and the type of biological process applied (Assurre, 2006, DEFRA, 2005). Figure 2.3 provides some options used in MBT systems. Generally, the mechanical sorting phase is placed at the front of the MBT process although some systems operate end-of-process sorting (Soyez and Plickert, 2002).

The part of mechanical treatment process can include shredding, screening, and both dry and wet processes, the application of which depends on what the plant wants to achieve. For example, the MBT plant can be designed to produce a solid derived fuel (SDF) coming from the dry stabilized material after removing the non-combustible material, such as glass; it can also aim to produce a refuse derived fuel

(RDF) by separating the components with high calorific value comprising paper, plastics and other combustible fractions, which have not been treated (Soyez and Plickert, 2002; Assurre, 2006). As far as the purpose of reducing the amount and biodegradability of BMW by MBT prior to landfilling is concerned, the purpose of the mechanical processes, for example, mixing, homogenising, grinding, agglomeration, sorting, separating, sizing, sieving etc., is to separate and/or recycle the substances which are unsuitable for biological treatment, and to remove interfering substances and pollutants such as metals, glass, stones, plastics, batteries etc., as well as to optimise the organic-rich fraction of the remaining degradable wastes that is suitable for composting or anaerobic digestion by increasing both availability and homogeneity (Raninger and Nelles, 1997; Assurre, 2006; CIWEM PPS, 2006).

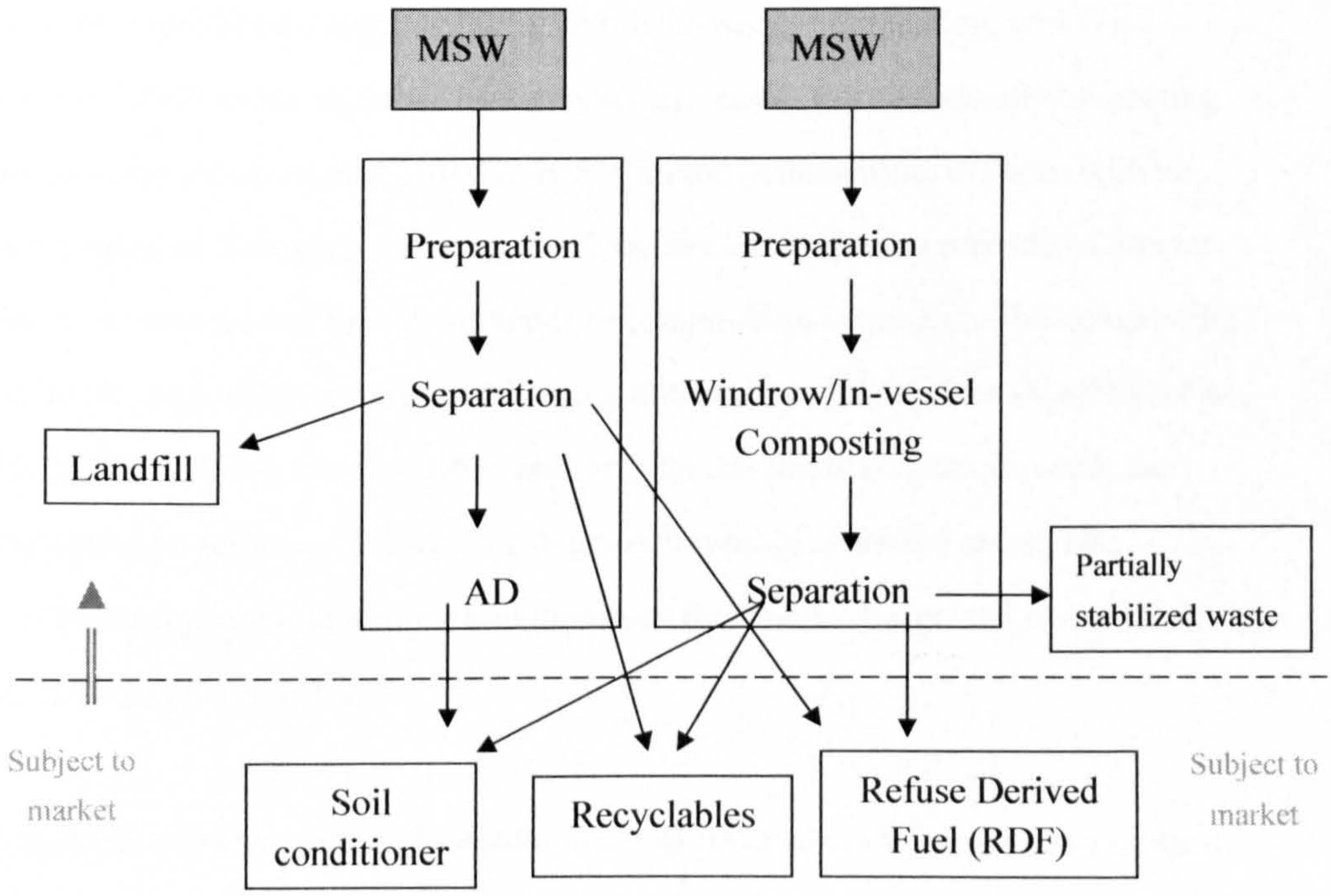


Figure 2.3. The typical options used in MBT systems (adapted from DEFRA, 2005)

Biological treatment processes are to handle the remaining organic substances, such as kitchen and yard waste, paper, cardboard etc.. The purpose of biological treatment is to reduce and convert the biodegradable substances of the incoming wastes to the greatest possible extent by the application of either aerobic or anaerobic treatment, or a combination of the two. The aerobic process is used widely in forms more frequently associated with composting. Composting systems range from the very simple to the very sophisticated. The most common and simplest form of composting is open–air windrow composting. Typically, windrows are constructed of shredded and /or screened refuse, which is placed in long parallel piles. These windrows are monitored for optimal temperature and moisture levels and are periodically aerated by turning the material around. The composting process can be accomplished in three to four weeks or more (Tchobanoglous, *et al.*, 1993; Vesilind *et al.*, 2002).

A more sophisticated composting system is in-vessel composting, which is accomplished inside an enclosed container or vessel. In an in-vessel composting process, the shredded and sorted refuse is mixed in an aerobic digester with air being injected through hollow augers. This contained method provides a greater degree of control and the rate of waste decomposition is quicker. The composting period is short, often as little as 24 hours, but usually about 5 days (Vesilind *et al.*, 2002). While these systems differ primarily in the aeration methods used, the biological principles of the aerobic degradation process are the same. The biochemical process and important variables for system design and operation are described in Section 2.2.

Anaerobic biodegradation of organic material proceeds in the absence of oxygen and in the presence of anaerobic microorganisms. The anaerobic process is most associated with water treatment plants and is described as Anaerobic Digestion (AD). It has been used for treatment of the degradable fraction of MSW because of the opportunity to recover methane and the digested material is similar to the

compost produced aerobically (Tchobanoglous, 1993). Another advantage of anaerobic pre-treatment is that minor odour problems occur in the process (Soyez and Plickert, 2002). The principles and processes of anaerobic degradation are described in Section 2.1.1.

The mechanically and biologically treated waste is characterised by an obvious reduction in amount (reduction up to 80wt. % of dry matter), water content and gas formation potential, as well as by a significant improvement in leaching and settlement behaviour in the landfill (Raninger and Nelles, 1997; Scheelhaase and Bidlingmaier, 1997; Pichler and Kögel-Knabner, 1999). For example, the biogas production potential of the MSW was found to be reduced by 90% to 19 L/kg DM after about 6 months of aerobic treatment (Lornage *et al.*, 2007). The environmental issues associated with MBT are bio-aerosols, pathogens, air and water pollution, and heavy metal content.

Although the biodegradability of the waste is reduced via the MBT process, the residue left may not be classified as inert (Strategy Unit, 2002). As the residue left by MBT should still be landfilled, emission from such landfills should be monitored and controlled for decades or more, as with other landfilled waste (Inanc *et al.*, 2005). Additionally, in order to help local authorities meet the EU landfill directive targets, criteria should be established for acceptance of these residues in landfill.

2.3.2 Effect of Aerobic Pre-treatment

As described in Sections 2.2.1 and 2.3.1, composting of the organic fraction of solid wastes can lead to the production of a soil amendment by-product and/or can be used as a pre-treatment technique prior to landfilling. In comparison to anaerobic degradation, aerobic biodegradation rates are more rapid and therefore can potentially decrease the time of stabilization of BMW mass.

An important aspect of composting is the reduction in the volume of waste (Whitney and Lynch, 1996). Additionally, the potential reasons behind the positive effects of aerobic pre-treatment on landfill behaviour are summarized by Komilis *et al.* (1999):

- Aerobic pre-treatment removes readily degradable matter. This prevents rapid hydrolysis rates thereby limiting consequent rapid acid accumulations once the waste is landfilled. Thus, a balance between acidogenic and methanogenic stages can be achieved quickly and little or no inhibition of methanogens takes place (Stegmann and Spendlin, 1987).
- Aerobic pre-treatment increases the temperature of the wastes which are placed in a landfill afterwards so that methanogenic conditions are enhanced by the increased temperature.

Stegmann and Spendlin (1987) also concluded that landfilling of aerobically pretreated MSW results in rapid establishment of methanogenesis.

The results of Brinkmann *et al.* (1995) showed that anaerobic degradation within a landfill environment required more time to reach the same organic material reduction as that achieved by aerobic treatment. Compounds with high lignin content are a major class of organic compounds in MSW that are slowly to negligibly degraded under anaerobic conditions (Komilis *et al.*, 1999; Whitney and Lynch, 1996). However, significant lignin degradation appears to be possible during aerobic composting (Hammouda and Adams, 1989; Tomati *et al.*, 1995; Howarth *et al.*, 1995). Aerobic composting involves the complete or partial degradation of all of the lignocellulosic compounds by a consortium of microorganisms (Whitney and Lynch, 1996). Lignin degraders in nature are mainly white-rot fungi, which is one of the main groups of microorganisms involved in the composting process (Tuomela, *et al.*, 2000). Since lignin is intimately associated with both cellulose and hemicellulose, partial decomposition of lignin during aerobic conditions makes cellulose and hemicellulose (the primary substrates during anaerobiosis) more readily available for methanogenic conversion (Komilis *et al.*, 1999). However, the

specific effect of lignin degradation under aerobic conditions on the biodegradability of the remaining carbon has not been extensively researched.

The effect of partial composting on the performance and efficiency of anaerobic digestion is also of significance as far as the valuable energy recovery of anaerobic digestion is concerned. ten Brummeler and Koster (1990) reported that the start-up of the dry anaerobic batch digestion (BIOCEL process) of the organic fraction of MSW at 30°C can be accelerated by partial aerobic composting for a period of 2 weeks. They observed that methanogenic decomposition was significantly enhanced by an initial composting pre-treatment step compared to the cases without pre-treatment, where very small amounts of methane were produced over a period of 180 days. A major drawback, however, was the loss of 40% of the potential methane yield during composting.

2.3.3 The Role of Anaerobic Treatment

It is generally recognized that anaerobic digestion is a more controlled and sustainable way of treating organic waste as compared to other disposal routes (i.e., landfilling or composting) because it provides an energy recovery alternative to less sustainable solid waste processing methods and also provides an option for the biological treatment technique. However, anaerobic processes are inherently slower than aerobic processes and a major class of organic compounds found in MSW, namely lignin-contained compounds, are difficult to degrade under anaerobic conditions. Biomass containing plant fibres has a limited digestibility due to the shielding effect that lignin provides to otherwise digestible cellulosic components (Colberg, 1988). The degradation of plants fibres under both aerobic and anaerobic conditions is described in detail in Section 3.2.2.

Industrial anaerobic digestion facilities processing waste biomass have been forced to accept the limited digestibility of MSW and other biomass wastes and

consequently provide a short-time (15~20 day) digestion phase for the readily digestible components, with a reliance on a followed composting step for stabilization of the remaining non-digested solids (Mata-Alvarez *et al.*, 1993; Leikam and Stegmann, 1997). Therefore, additional treatments are required to remove the protecting shield of lignin-hemicellulose to enhance the biodegradability of cellulose, hemicellulose and lignin in waste in anaerobic digestion (Komilis *et al.*, 1999).

2.4 Summary

The migration of gas and leachate away from the landfill boundaries and their release to the surrounding environment can cause serious environmental concerns. Mechanical and biological treatment can separate recyclable material and/or recover energy from MSW, and reduce the amount of BMW that contributes to gas and leachate emission when landfilled. Landfill continues to be an important option for containment and further stabilization of waste residues left by other more sustainable options, such as recycling and MBT.

As for biological treatment processes, both composting and anaerobic digestion have potential roles for the diversion of BMW from landfills. Aerobic biodegradation rates are more rapid and could potentially decrease the time required for stabilization of MSW in comparison to anaerobic digestion and direct landfilling.

CHAPTER 3 CHARACTERIZATION OF MSW

MBT plants can produce treated waste material with its biodegradable content reduced to a level that, when landfilled, will allow the waste disposal authority to meet its BMW allowance under the Landfill Directive (CIWEM PPS, 2006). However, there remain a number of obstacles and uncertainties within the use of MBT. A primary uncertainty relates to whether the end product delivers the reductions required by BMW landfill allowance of the Waste Disposal Authority (WDA). MBT processes that cannot demonstrate a reduction in the biological content of the waste to the satisfaction of the Environment Agency will not be acceptable to Local Authorities (CIWEM PPS, 2006).

Characterization of MSW is of great importance for MSW management. In order to monitor the performance of the WDAs in meeting the targets for diverting BMW from landfills, waste characterization should be performed regularly at different stages of disposal and treatment. It includes sorting the raw, untreated MSW into different categories, and a series of physical, chemical and biological tests to determine the mass of waste and BMW, as well as the stability of the treated waste. There are some test methods available (Table 1.1) to measure the waste biodegradability. These methods have different principles and the application depends on the specific purposes or circumstances. The test methods reviewed in this chapter include three non-biological tests (loss on ignition (LOI), carbon to nitrogen ratio, and the fibre test) and two biological tests (BMP and DRI)), which are regarded as essential and/or promising for characterization of waste degradability (Soyez and Plickert, 2002; Environment Agency, 2005; Godley *et al.*, 2005).

3.1 Classification of Untreated MSW

3.1.1 Classification of Untreated MSW

Dixon and Langer (2005) reviewed and discussed a number of existing MSW classification systems, which differ in a range of criteria, such as degradability, shape, strength, size, etc.. Whatever the classification system, the starting point is identification of the initial waste groups or categories. The waste composition is defined by measuring the weight percentage of each group or component in the surveyed sample (Tchobanoglous *et al.*, 1993). Because of the large variety of components present in MSW, a practical approach is to identify major groups of material (Dixon and Langer, 2005).

It is well known that the composition of raw MSW varies substantially with economic and cultural conditions, location, season, waste collection and disposal methods, etc. (Bonomo and Higginson, 1988; Tchobanoglous *et al.*, 1993). The variety and complexity of MSW hinders the establishment of standard category/group of waste components for classification of MSW samples and different descriptions are used in experimental programmes (Barlaz, *et al.*, 1990; El-Fadel, 1997; Dixon and Langer, 2005). For example, the English Municipal Waste Survey in 2000/2001 used the following main groups: garden waste, paper and board, kitchen waste, general household sweepings, glass, wood, metal, plastic, textiles, nappies and soil (Strategy Unit, 2002). An American waste composition survey done by the Department of Environmental Quality (1998) used the following main groups: organic, paper, wood, polymer/plastics, metal, soil-like, ceramic, glass, inerts and rubber (Dixon and Langer, 2005).

Components that typically make up the residential portion of MSW and their distribution are given in Table 3.1, which was summarized by El-Fadel *et al.* (1997)

and Burnley (2007). The waste categories in Table 3.1 were selected because they are easily identifiable and have proven adequate for the characterization of MSW for most applications (Tchobanoglous *et al.*, 1993).

Table 3.1. Typical MSW composition

Waste Category	European Community[a]	United States[a]	UK[b]
Paper/cardboard	20-42	28-50	23-25
Food waste	20-50	6-18	35-38
Yard waste	12-18	5-20	
Plastics	3-8	4-10	8-10
Glass	4-12	4-12	6-7
Metal	3-13	3-13	3-5
Wood/rubber/leather/textiles	2-14	1-12	-
Inerts/inorganics (Dirt, ash, etc.)	1-20	0-6	-

Unit used is % weight.

[a] Summarized by El-Fadel *et al.* (1997); [b] Summarized by Burnley (2007).

For the purpose of monitoring BMW diversion from landfills, the Environment Agency produced a guidance document on how to monitor the performance of MBT and similar processes (Environment Agency, 2005). In this monitoring protocol, raw, untreated MSW are sorted into five categories of waste composition, including:

- Metals (cans, wire, engine blocks etc.);
- Glass (whole, broken or cullet);
- Plastics (sheets, bottles, caps high density polyethylene (HDPE), polyethylene (PE) etc.);
- Non-combustibles (other non-biodegradable materials e.g. stone, ceramic, slate);
- BMW (All potentially biodegradable material; includes yard waste, kitchen waste, paper, cardboard, textiles, and any fines that are not recognised in other categories).

All BMW is categorized into a whole group because they are the monitored objects as far as the performance of WDAs is concerned, and it is convenient for carrying out regular checks/analyses.

3.1.2 Major Constituents in BMW

In terms of chemical constituents, seven major organic complexes and compounds constitute organic matter in BMW according to Epstein (1997). They are carbohydrates and sugars, protein, fats, hemicelluloses, cellulose, lignin and mineral matter. The first three constituents are readily biodegradable whereas hemicellulose and cellulose are slower to biodegrade. Here cellulose and hemicellulose are not included in the category of carbohydrates because of differences in their degradation rates, although some researchers (e.g. Brauer *et al.*, 2005) have categorized them as carbohydrates. Lignin is much more resistant to biodegradation and mineral matter does not biodegrade (Epstein, 1997).

The constituents of the organic matter in BMW vary with source because of the variety of waste composition. For example, generally, food wastes contain considerably lower amounts of cellulose and lignin than wood and wheat straw. Even among the same category of sources the constituents may also vary. For example, Table 3.2 shows the percentage range for the various constituents for both plant and manure. According to Gray *et al.* (1971), the constituents of manure depend on the type of animal and its feed; the variation in plant constituents relates to the species (e.g., straw or grass), age (fresh or aged), and the environment. El-Fadel *et al.* (1997) summarized the organic constituents in MSW (Table 3.3) by referring to Pfeffer and Khan (1976), Pfeffer (1976) and Barlaz and Ham (1993).

Separation and quantifying some of these constituents can provide more detailed characterization of BMW with regards to biodegradation, such as readily biodegradable or slowly biodegradable. Furthermore, some chemical indicators

derived from the concentration of these constituents, such as cellulose/lignin ratio, lipids and protein index, or humic stability index (humic acid / fulvic acid), have been identified and used to indicate stability of the studied material (Brauer *et al.*, 2005). The partition of cellulose, hemicellulose and lignin in BMW, as well as the study of their relative contents in the aspect of biodegradability, is reviewed in Section 3.2.2 and 3.3.3.

Table 3.2. Constituents of plant and manure

Constituents	Percent Dry Weight	
	Plants	Manure
Hot and cold water soluble- sugars, starches, amino acids, aliphatic acids, urea and ammonium salts	5-30	2-20
Ether/alcohol soluble-fats, oils, waxes, resins	5-15	1-3
Proteins	5-40	5-30
Hemicellulose	10-30	15-25
Cellulose	15-60	15-30
Lignin	5-30	10-25
Inorganic (ash)	1-13	5-20

From Gray *et al.*, 1971.

Table 3.3. Constituents of solid waste

Constituent	Range (Percent Dry Weight)	Average (Percent Dry Weight)
Cellulose, sugar, starch	52-64.5	58
Hemicellulose	11.9	11.9
Lignin	5.4-15.2	11.2
Lipids	5.7	5.7
Protein	2.6-4.2	3.4

From El-Fadel *et al.*, 1997.

3.2 Characteristics of Lignocellulose

MSW comprises approximately 40-50% cellulose, 12% hemicellulose and 10-15% lignin on a dry weight basis (Wang *et al.* 1994). The cellulose plus hemicellulose

fraction makes up for over 90% of MSW methane potential (Bookter and Ham, 1982; Barlaz *et al.*, 1989b). According to Lynch (1993), cellulose, hemicellulose and lignin are the principle providers of the carbon and energy for microorganisms in the production of stabilized compost, which may also be true for the long-term biodegradation of BMW.

3.2.1 Lignocellulose Structure

Cellulose, hemicellulose and lignin, which are collectively termed as lignocellulose, are the three major components of plant vascular tissue. Figure 3.1 shows the structural relationship between the cellulose, hemicelluloses and lignin parts of a plant cell wall. Spiral layers of cellulose are located around the inside lumen part, which give flexibility and strength to the cell-wall structure. The hemicellulose surrounds the cellulose elements and provides both a link and even chemical bonding between the cellulose and the lignin which is located further outside. This complexity of compounds forms a unique material. The inside parts of a cell wall are highly flexible due to the linear hydrophobic chains of cellulose; meanwhile the outside parts are highly resistant to microbial degradation due to the amorphous but also hydrophobic lignin, (Colberg, 1988; Pekka, 2000).

Cellulose: '*Cellulose is the main polymeric component of the plant cell wall and is the most abundant natural polysaccharide on earth*' (Pekka, 2000). A single cellulose chain is represented in Figure 3.2. It is unbranched with several D-glucose units joined via ß-1, 4-glycosidic linkages and is insoluble in water. Cellobiose is a disaccharide product from partial hydrolysis of cellulose and contains the ß-1, 4-bond, while glucose is released as a result of complete hydrolysis (Colberg, 1988).

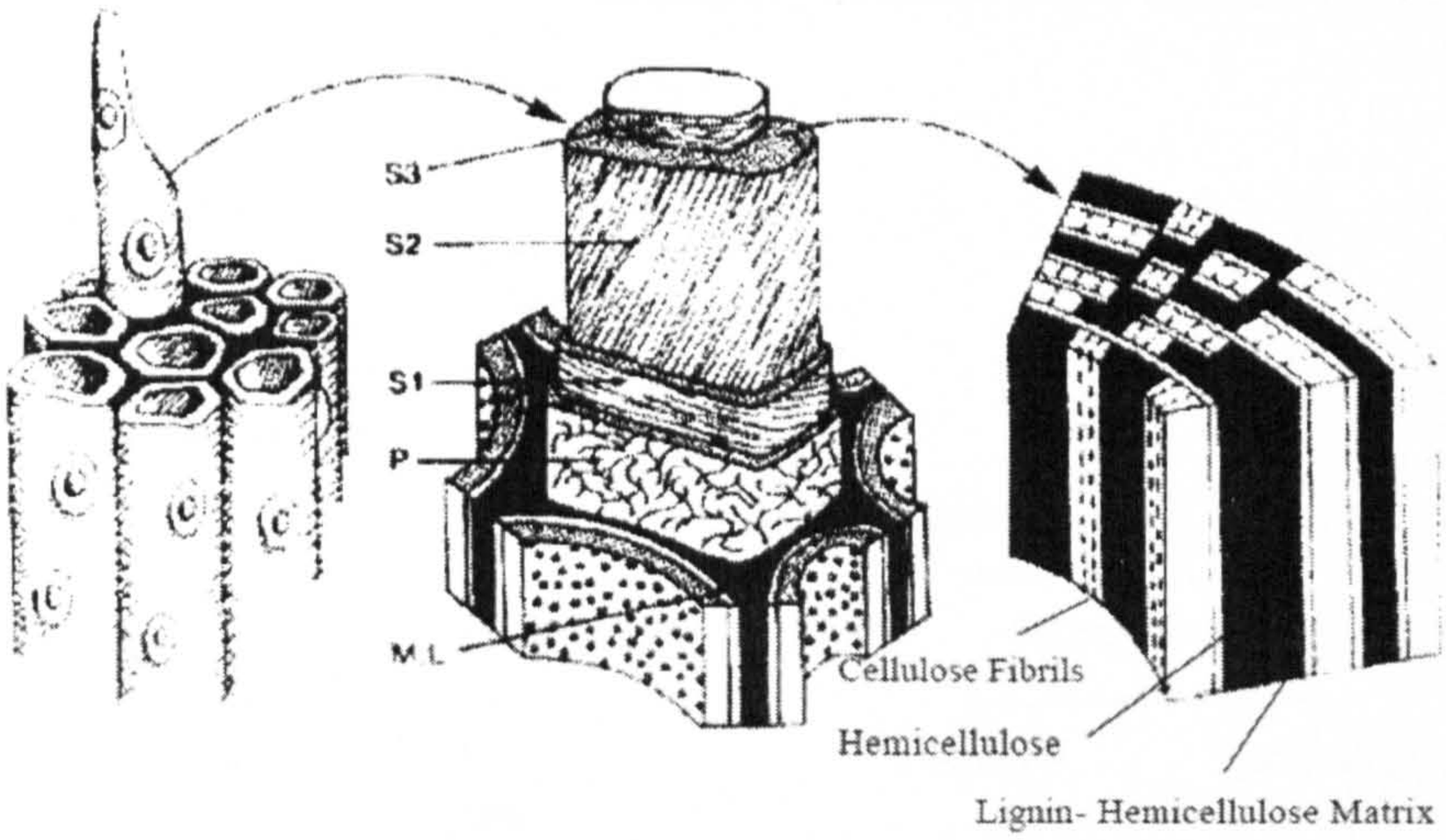


Figure 3.1. Schematic illustration of the morphology of the tracheids, secondary wall layers and the relationship of the lignin, hemicelluloses, and cellulose in the secondary wall of a tracheid (Pekka, 2000)

(S1-S3: secondary cell wall layers; P: primary wall; M.L.: middle lamella)

Hemicellulose: Hemicellulose consists of several different sugar units (xylose, arabinose, galactose, mannose, and glucose) and substituted side chains in the form of linear or branched polymer with a low molecular weight. Hemicelluloses bind bundles of cellulose fibrils to form microfibrils, which enhance the stability of the cell wall. They are also cross linked with lignin, creating a complex web of bonds which provides structural strength and is also resistant to microbial degradation (Ladisch *et al.*, 1983; Lynch, 1993).

Lignin: Lignin ranks second only to cellulose as the most abundant naturally occurring polymer on earth. Lignin is a three dimensional macromolecule with covalent bonds and a very high molecular weight. It is a branched, aromatic polymer composed of phenylpropane sub-units, which are randomly linked by a variety of carbon-carbon and ether bonds. A portion of "average lignin" is illustrated in Figure 3.2. The most common intermonomeric link in lignin is the arylglycerol-ß-aryl ether bond, located between aromatic rings 2-3, 4-6, and 7-9. Lignin is hydrophobic and

acts like a "water-proof" cover surrounding the cellulose fibrils in terms of chemical structure and composition (Colberg, 1988).

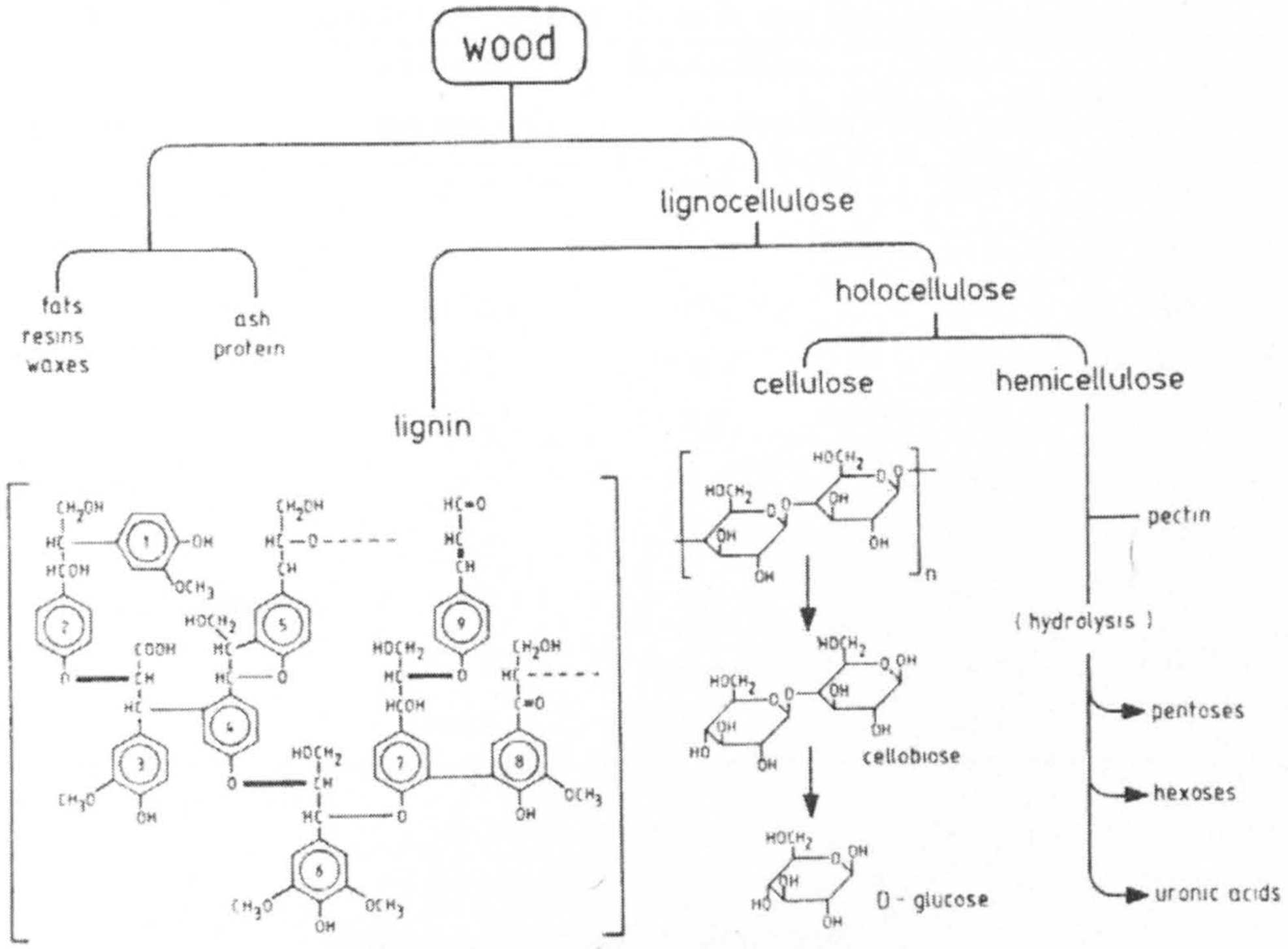


Figure 3.2. Major components of wood, including the chemical structure of lignin (model), cellulose, and its hydrolysis products, cellobiose and D-glucose (Colberg, 1988).

Table 3.4 presents the test results of the content of cellulose, hemicellulose and lignin in MSW components and one MSW sample in the study of Eleazer, *et al.* (1997), where the contents of lignocellulose were determined by acid hydrolysis of the sample. The analysis methods are detailed in Section 3.3.3. The cellulose plus hemicellulose to lignin ratios ((C+H)/L) were in a wide range of 0.6 - 41.7. Komilis and Ham (2003) analysed cellulose, hemicellulose and lignin in food wastes, mixed paper, yard wastes, grass, leaves, branches and mixtures of the first three groups of wastes using the same fibre analysis method of acid hydrolysis. The (C+H)/L ratios were also in a wide range from 0.4 (leaves) to 11.5 (office paper). Chandler *et al.* (1980) tested a number of substrates which offered a wide range of cell wall and

lignin content. They used the procedure developed by Van Soest (1963a, b, 1967) for fibre analysis and the results are shown in Table 3.5.

Table 3.4. Lignocellulose content of MSW and the components

Material/ Component	Cellulose (% Dry wt.)	Hemicellulose (% Dry wt.)	Lignin (%Dry wt.)	(C+H)/L
Grass-1	26.5	10.2	28.4	1.3
Grass-2	25.6	14.8	21.6	1.9
Leaves	15.3	10.5	43.8	0.6
Branch	35.4	18.4	32.6	1.7
Food	55.4	7.2	11.4	5.5
Coated paper	42.3	9.4	15.0	3.4
Old newsprint	48.5	9.0	23.9	2.4
Old corrugated containers	57.3	9.9	20.8	3.2
Office paper	87.4	8.4	2.3	41.7
MSW	28.8	9.0	23.1	1.6

Source: Eleazer *et al.* (1997)

Table 3.5. Substrate composition

Material	Lignin (%VS)	(C+H) (%VS)	(C+H)/L
Wheat straw	8.9	68.2	7.7
Corn stalks	3.9	45.7	11.7
Corn leaves	3.8	55.5	14.6
Cattails	8.5	55.0	6.5
Treated kelp	6.0	27.2	4.5
Water hyacinth	8.7	51.4	5.9
Corn meal	2.0	19.6	9.8
Newsprint	20.9	67.8	3.2
Elephant manure	10.4	67.0	6.4
Chicken manure	3.4	41.8	12.3
Pig manure	2.2	38.3	17.4
Cow manure R1	8.1	49.0	6.0
Cow manure R2	7.9	44.4	5.6
Cow manure R3	10.1	52.8	5.2

Source: Chandler *et al.* (1980)

VS: volatile solids; (C+H): (Cellulose + Hemicellulose).

3.2.2 Decomposition of Lignocellulose

The simplicity of the cellulosic structure, using repeated identical bonds, means that only a small number of enzymes are required to degrade this material. Cows and other ruminants create an environment in their rumen which encourages the microbial degradation of cellulose, converting the cellulose to volatile fatty acids and microbial biomass which the ruminant can then digest and use (Richard, 1996). An important feature of cellulose, relatively unusual for a polysaccharide material, is its crystalline structure. Approximately 30 individual cellulose molecules are assembled into larger units known as elementary fibrils (protofibrils), which are packed into larger units called microfibrils, and these are in turn assembled into the familiar cellulose fibres (Lynd *et al.*, 2002). The structural features of cellulose considered as hydrolysis rate –impacting factors include crystallinity index (RCI; the most commonly used estimate of degree of interchain hydrogen bonding), degree of polymerization and accessible area (Zhang and Lynd, 2004). Though the relationship between structural features of cellulose and rates of enzymatic hydrolysis has been extensively studied and reviewed (Cowling and Kirk, 1976; Lynd *et al.*, 2002), it is still not completely understood (Zhang and Lynd, 2004). For example, Fan *et al.* (1981) and Gharpuray *et al.* (1983) reported that the crystallinity index is the most important factor affecting cellulose digestibility. Whilst, Fierobe *et al.* (2002) concluded that accessibility of cellulose is a more important factor than crystallinity index in determining the hydrolysis rate by comparing the hydrolysis rates on various sources of model cellulosic substrates.

The presence of lignin can interfere greatly with cellulose and hemicellulose degradation (Micales and Skog, 1997). The ability of lignin to resist biological and chemical degradation allows it to protect cellulose within lignocellulose materials. It has been estimated that at least 18% of the cellulose and hemicellulose in refuse is resistant to degradation because of its close association with lignin. For example, newsprint, a major portion of the paper in MSW, mainly consists of cellulose (about

48%), hemicellulose (9-18%), lignin (22-25%) and some minor components, such as lipids, protein, and ash (Stinson and Ham, 1995; Eleazer *et al.*, 1997; Wu *et al.*, 2001). Among the several factors contributing to the slow rate of newsprint degradation in landfills, the association between lignin and cellulose/hemicellulose is especially important because either physical blockage or the presence of resistant chemical bonds can hinder the microbial degradation of cellulose and hemicellulose in the lignocellulosic materials (Benner *et al.*, 1984; Stinson and Ham, 1995). The effect of lignin on the bioavailability of other cell wall components is thought to be largely a physical restriction, with lignin molecules reducing the surface area available to enzymatic penetration and activity (Haug, 1993). By investigating cellulose decomposition in lignocellulosic materials through the use of BMP experiments, Stinson and Ham (1995) also found that the inhibition of cellulose decomposition in newspaper is not due to chemical effects of lignin (e.g. enzyme adsorption) but is mainly due to the physical association of lignin and cellulose.

Colberg (1988) suggested that refuse must be considered as a lignocellulosic substrate. The lignin concentration has been related to both the rate and extent of cellulose plus hemicellulose degradation. However, the degree to which this relationship can be modelled varies between studies. Dehority and Johnson (1961) reported that cellulose digestibility was not directly correlated to the lignin concentration. In their study, legume cellulose was digested to a lesser extent than the cellulose in four types of grass, although the grasses and legumes had similar lignin concentrations. Chandler *et al.* (1980) formulated an equation for the anaerobic degradation of different organic materials (r^2=0.94), in which VS destruction is directly proportional to the lignin content of the material:

$$B=0.830\ \ 0.028X, \qquad (3.1)$$

where B is biodegradable fraction of VS (<0B<1) and X% is the lignin content of the VS. The data were collected from the anaerobic fermentation process using a

wide range of lignocellulosic materials (Chandler *et al.*, 1980). They also concluded that lignin was the predominant factor in determining the extent of organic substrate degradation in anaerobic conditions.

The complexity of the lignin structure has proven to be as resistant to detailed biochemical characterization as it is to microbial degradation, which greatly impedes the understanding of its effects on degradation of lignocellulose. Micales and Skog (1997) reviewed the decomposition of lignocellulose in landfills. Cellulose is hydrolyzed into glucose and cellobiose in the landfill. These compounds are then fermented into other compounds, including carbon dioxide, hydrogen, ethanol, and acetic, propionic, butyric, valeric, and caproic acids (Rees, 1980). Hemicellulose, which coats the cellulose fibrils in wood and paper, is also metabolized by landfill bacteria (Ghosh *et al.*, 1985). Lignin is extremely resistant to chemical and enzymic degradation because of the three dimensional network that the polymers form (Burla, 1995). Colberg (1988) has reviewed the anaerobic microbial degradation of lignin compounds and concluded that the intermediate metabolic products called oligolignols, released during aerobic degradation, may be partially degraded to CO_2 and CH_4 by anaerobic microorganisms. Also, polymeric lignin is mineralized in anoxic sediments at a slow rate. Benner and Hodson (1985) reported that an elevated temperature of 55°C enhances the anaerobic degradation of lignin. It has also been shown that anaerobic rumen microorganisms are capable of degrading plant fibre cell walls (Kuhad *et al.*, 1997). In many studies, however, lignin was found to be not metabolized by anaerobic bacteria and not significantly decomposed in landfills (Young and Frazer, 1987; Barlaz *et al.*, 1989a, b, 1990; Suflita *et al.*, 1992; Wang *et al.* 1994, Micales and Skog, 1997). In optimized laboratory studies of anaerobic degradation, 71% of cellulose and 77% of hemicellulose from typical landfill refuse was degraded; lignin degradation was negligible, even under the most ideal conditions (Barlaz *et al.*, 1989 b).

Significant lignin degradation appears possible during aerobic composting (Hammouda and Adams, 1989; Tomati *et al.*, 1995; Howarth *et al.*, 1995). Aerobic microorganisms are primarily lignin degraders in most environments. Some organisms, particularly fungi, have developed the necessary enzymes to rapidly break lignin apart. The initial reactions are mediated by extracellular lignin and manganese peroxidases, primarily produced by white-rot fungi (Kirk and Farrell, 1987). Tuomela *et al.* (2000) reviewed the biodegradation of lignocellulosic materials, especially lignin in a compost environment. Their main emphasis is placed on thermophilic fungi because they occur frequently in compost and fungi, especially white-rot fungi, are also the most important group of lignin biodegraders in nature. The review showed that very little is known about lignin degradation in compost, although lignin degradation by white-rot fungi had been extensively studied in recent years.

Komilis and Ham (2003) investigated the biodegradation process of seven solid waste components and seven combinations of these components under aerobic conditions. A linear equation (B=0.85 0.01X) was derived that correlate degradation extent (B, as indicated by the volatile solids reduction) to initial lignin contents (X%). In comparing the linear equation to the similar equation developed for anaerobic environments by Chandler *et al.* (1980), lignin was found to be less inhibitory to the overall substrate decomposition in aerobic environments compared to anaerobic ones.

The impact of lignin degradation on the biodegradability of the remaining carbon has not been extensively researched (Richard, 1996). In one of the few studies which might provide an insight, Latham (1979) measured a 5 to 11% increase in anaerobic digestibility of barley straw after 3 to 4 week aerobic incubation at 30°C with various pure cultures of white-rot fungal species. Komilis *et al.* (1999) concluded that partial decomposition of lignin during aerobic conditions makes cellulose and hemicellulose, the primary substrates during anaerobiosis, more

readily available for methanogenic conversion. However, they did not present the data / reasons from which these conclusions could be drawn.

3.3 Non-biological Indicators for stability testing

In order to monitor and assess the reduction in biodegradability in the outputs from MBT and other pre-treatments, the EA guidance recommends a suite of preferred tests, of which the non-biological tests are dry matter (DM) content, loss on ignition (LOI), total organic carbon (TOC), and total nitrogen (TN) (Environment Agency, 2005).

Godley *et al.* (2005) reviewed and evaluated several biological and non-biological methods for characterizing municipal organic wastes (Table 1.1). The non-biological methods evaluated are: DM, LOI, TOC, TN, water leachable dissolved organic carbon, biological oxygen demand (BOD), chemical oxygen demand (COD), lignin and 'cellulose' (sum of cellulose and hemicellulose here) content (by acid detergent fibre methods), and cellulose hydrolysis. In this review, tests of DM, LOI, TC and TN content, and fibre content are considered because they were recommended by a number of researchers (e.g. Chandler, *et al.*, 1980; Epstein, 1997; Brauer, *et al.*, 2005; Godley *et al.*, 2005) either for characterizing waste composition or for indicating the stability of test material. These tests are also used in this research.

3.3.1 Moisture, DM, LOI and Ash Content

The measurements of moisture, DM, LOI content and ash content are gravimetric procedures performed on solid samples representative of the whole waste. The weight loss on ignition (LOI) is also called 'volatile solids (VS)', which generally represents the total organic solid of the material and has been used as an integral criterion to characterise the mineralization of the wastes (Laine-Ylijoki *et al.*, 2004,

Soyez and Plickert, 2002). The moisture and DM content are determined at 105°C, and LOI and ash content at 550 °C.

The Guidance on Landfill Completion (2003), which replaces WMP 26A (1993), recommends that generally, samples with a LOI content greater than 10 % dry weight have the potential to generate significant amounts of methane. In the standards set by the Austrian Landfill Directive (1997) for acceptance of treated waste in landfill, the waste with LOI ≤ 8 % DM is allowed to be landfilled; however, this limit cannot be met by MBT and is only possible using incineration (Binner, 2002). Depending on the ratio of readily degradable material to total LOI content, there may be some wastes with LOI content between 25-35% DM or higher which will not generate significant amounts of methane, especially in the case of waste after MBT.

In some studies, LOI is not regarded as a suitable criterion to describe the potential biological reactivity and may lead to incorrect conclusions from the test results (Soyez and Plickert, 2002; Laine-Ylijoki *et al.*, 2004). The main reason is that LOI also includes organic parts which do not degrade and will not contribute to any biochemical reaction. However, the results of Godley *et al.* (2005) shows that this gravimetric method of waste characterization is very reproducible and can be used for general waste characterization. Additionally, since the composition of waste and some other tests (e.g. BMP and DRI) are based on DM and/or LOI, general characterization in terms of DM and LOI content is essential. The LOI test is also proposed as a measure for reduction of biodegradability across MBT processes in the ' Consultation on Revised Guidance' (Environment Agency, 2007).

3.3.2 TC/TN Ratio

The TC/TN ratio is used to demonstrate suitability of the material for composting; it has also been used as a chemical analysis method to assess the stability or maturity of compost, because the TC/TN ratio changes with the aging of the compost,

reaching some values which are characteristics of a stable organic material (Morel *et al.*, 1985). Ratios below 20 were assumed to be indicative of stable compost by Chayansak and Kubota (1981). This may be misleading because of the variety of the start TC/TN ratio for different material. For example, mixing BMW with sewage sludge, which is rich in nitrogen, may lower the TC/TN values to those corresponding to mature compost even though the BMW has not degraded yet. Therefore, Morel *et al.* (1985) proposed that if the TC/TN ratio is a good indicator of the biological stability of compost, it would be essential to interpret the final TC/TN according to the initial characteristics of the material.

3.3.3 Fibre Analysis

The measurement of cellulose, hemicellulose and lignin has been described and can be determined by several acid detergent fibre (ADF) methods (Effland, 1977; Pettersen and Schwandt, 1991; Kitcherside *et al.*, 2000). One basic procedure is the hydrolysis of a solid sample in 72% (wt/vol) H_2SO_4 followed by a secondary hydrolysis in 3% (wt/vol) H_2SO_4. The hydrolysis converts cellulose and hemicellulose to their respective monomers. Typically, HPLC equipped with a pulsed ampiometric detector (Petterson and Schwandt, 1991) is used to quantify the concentrations of glucose, xylose, mannose, arabinose, and galactose in the acid hydrolysate. The glucose originates from cellulose and the other sugars from hemicellulose. The solids remaining after acid hydrolysis are combusted at 550°C, and the lignin content is calculated from the weight loss of the dried solids after combustion. This procedure was widely used to characterize the biodegradability of refuse components (Barlaz *et al.*, 1989a, b; Ham *et al.* 1993; Wang *et al.*, 1994, 1997; Eleazer *et al.*, 1997; Hossain, 2003). However, these procedures are laborious and involve several filtration steps, which may introduce errors due to losses in the test sample (Kitcherside *et al.*, 2000).

Van Soest introduced the use of detergents in feed analyses to determine cell wall components (Van Soest, 1963a, 1963b; Van Soest and Wine, 1967; Van Soest *et al.*,

1991). This methodology has gained wide acceptance for feed characterization and is now used all over the world. One procedure is an ADF method (Goering and Van Soest, 1970) using heat treatment of the sample with 0.5 m sulphuric acid containing 2% cetyltrimethyl – ammonium bromide (CTAB). The CTAB dissolves nearly all the nitrogenous constituents, and the acid hydrolyses the starch. ADF contains mainly cellulose, lignin and variable amounts of silica (Equation 3.2). The lignin content is generally determined by permanganate oxidation and the resulting residue gives the cellulose content. ADF is an AOAC-approved method of analysis (AOAC, 1984).

A second procedure is a neutral detergent fibre (NDF) method, which is based on extraction of the feed with a hot neutral solution of sodium lauryl sulphate (Van Soest and Wine, 1967). This detergent extracts lipids, sugars, organic acids and other water soluble components as well as pectin, non-protein nitrogen (NPN) compounds, soluble protein and some of the silica and tannin. NDF is the insoluble residue made up of cellulose, hemicellulose, lignin, lignin-bound nitrogen, some protein, minerals and cuticle (Equation 3.3). NDF % is the weight of the residue expressed as a percentage of the original sample. The NDF determination can be considered to be a rapid way of estimating the cell wall content. The difference between NDF and ADF value provides an estimate of the content of non-cellulosic polysaccharides, which is the hemicellulose. Although widely used for fibre analysis of ruminant feeds, the NDF procedure is not an official AOAC approved method.

One of the major methods for lignin determination utilizes strong mineral acids to hydrolyze the other cell-wall components, leaving lignin as a residue to be measured gravimetrically. The other major class of methods employ oxidizing agents (such as $KMnO_4$) to remove the lignin selectively, which was used by Godley *et al.* (2005). In the second class of methods, lignin is estimated either by the loss in mass of the sample or through a photometric assay for lignin oxidation products. Jung *et al.* (1997) compared the acid detergent lignin (ADL) and Klason lignin methods for

their correlation with forage digestibility. The ADL method was first developed by Van Soest (1963b) and has thereafter been modified. The ADL procedure of Van Soest (1967) is most commonly employed by animal scientists and agronomists for the analysis of forages. The Klason method is the standard method for determining the lignin content of wood (Swift *et al.*, 1979). The ADL and Klason lignin procedures are similar in concept, but differ in order of reaction conditions. In the ADL method, the sample is first subjected to a dilute acid detergent solution consisting of CTAB at an elevated temperature (100℃) during the ADF step and then to concentrated acid (72% H_2SO_4) at a lower temperature; solubilized matter is removed by filtration between the two acid steps. ADL is the insoluble residue made up of lignin and minerals (Equation 3.4). In contrast, in the Klason lignin procedure, the sample is first treated with concentrated acid followed by dilute acid at a high temperature without a filtration step in between. These differences of the order of acid strength used, the inclusion of detergent in the ADF step and addition of the filtration step to the ADL procedure account for the difference in lignin values as measured by the ADL and Klason lignin methods (Kondo *et al.*, 1987; Lowry *et al.*, 1994).

ADF = Cellulose + Lignin + Mineral Ash (3.2)

NDF = Cellulose + Hemicellulose + Lignin + Mineral Ash (3.3)

ADL = Lignin + Mineral Ash (3.4)

Cellulose content = ADF-ADL (3.5)

Hemicellulose content =NDF – ADF (3.6)

Lignin =ADL –Ash (3.7)

Using the ADF, NDF and ADL tests, the cellulose, hemicellulose and lignin contents of the waste material can be determined (Equation 3.5 to 3.7). Jung (1997) discussed some of the major methods used in forage fibre analysis (ADF, NDF and ADL) and the strengths and weaknesses of each method. On the basis of the conventional procedures for analysis of NDF described by Van Soest *et al.* (1991)

and the Modified Acid-Detergent Fibre (MADF) by Clancy and Wilson (1966), Kitcherside *et al.* (2000) introduced an improved procedure for the analysis of fibre in feed samples. The FibreCap procedure uses a cylindrical capsule to contain the prepared waste sample through the various stages of the procedure and has been designed to speed up the analysis, limit the solvents used and more importantly to reduce the variability and systematic errors associated with extraction and filtration of the sample. The capsule walls are porous; they have the same filtration characteristics as Whatman 541 filter paper and are made from a hydrophilic material to facilitate the movement of solvents. The capsule lid is fabricated from a hydrophobic material to facilitate gas exchange. The FibreCap system is consistent with statutory requirements for fibre analysis, is more repeatable than conventional methods and is quicker in operation.

As reviewed in the Section 3.2, the major biodegradable components of refuse are cellulose and hemicellulose; lignin is the other major organic compound present in refuse. As cellulose and hemicellulose are slowly degradable material and lignin is difficult to degrade, their relative concentrations have been used to assess the degree of decomposition in a landfill at various stages (Bookter and Ham, 1982; Wang *et al.*, 1994; Baldwin *et al.*, 1998). In particular, the cellulose to lignin ratios (C/L) of approximately 0.8 have been recorded for 8-year-old landfill refuse, while ratios of 0.16-0.24 have been recorded at even older landfills. C/L ratios up to 4.04 have been recorded for fresh refuse (Bookter and Ham, 1982). Because an ADF test allows a screening of organic materials into readily degradable (and therefore of high short-term risk in landfill) and slowly degradable (such as cellulose) or inert organic material, in the Guidance on Landfill Completion (2003) it is suggested that waste samples with an ADF/LOI ratio of less than 0.1, or an ADF concentration of less than 1% may be regarded as sufficiently degraded to present little or no methane risk. Samples with an ADF/LOI ratio greater than 0.25, or an ADF concentration greater than 2.5% should be regarded as having the potential to produce methane.

Those samples falling within this range could be further evaluated by a biochemical methane potential (BMP) test.

By investigating the refuse samples excavated from a landfill Wang *et al.* (1994) found positive relationships between the methane yield and the sum of cellulose and hemicellulose contents (r^2=0.68), as well as the (C+H)/L ratio (r^2=0.67). Although these relationships were statistically significant (P<0.005), they were regarded as unreliable because among the data points there were large differences in the measured methane yields for those samples with similar cellulose and hemicellulose contents(Wang *et al.*,1994). Barlaz *et al.* (1997) and Eleazer *et al.* (1997) investigated nine MSW components (as showed in Table 3.4) A significant and positive relationship between the methane yield and the sum of cellulose and hemicellulose contents was found (r^2=0.49, P<0.025). However, similar to the result of Wang *et al.*(1994), in their results it is obvious that the methane yield of food waste was the highest while cellulose plus hemicellulose content was relatively lower, and they suggested that many confounding factors, which have not been clarified, prevented establishment of a quantitative relationship. There was no linear relationship found between the (C+H)/L ratio of each components and the biodegradability (r^2=0.02, Barlaz et al., 1997; Eleazer *et al.*, 1997). The possible reasons are (1) the cellulose and hemicellulose in some material was not bioavailable, which limited the methane production (Wang *et al.*, 1994); (2) the traditional procedures or techniques used for fibre determination are laborious and involve several filtrations, which may cause error for the results; (3) the readily degradation composition (such as sugars, fat) in some components influence the order of biodegradability when only lignocellulose is considered.

Komilis and Ham (2003) studied aerobic degradation of seven MSW components and mixtures. A decrease of C/L ratio was observed for most runs. Furthermore, the C/L ratio was recommended by the authors as a relatively accurate compost maturity indicator, because for most materials which had reached their degradation extent the

C/L ratio reduced to a value less than 0.5. Chandler *et al.* (1980) were the first to suggest using the Van Soest-developed procedures in estimating substrate methane fermentation biodegradability and have found several relationships relating substrate biodegradability to its composition. The substrate they studied is given in Table 3.5.

3.4 Biological tests

The biological methods used in this research include the BMP test and DRI test, which have been reported to be the most promising biological tests for the estimation of organic waste biodegradability (Adani *et al.*, 2004; Godley *et al.*, 2005). They are also the only two biological tests required by the EA monitoring guidance (Environment Agency, 2005) for evaluation of the performance of BMW diversion from landfill. This section gives a review of these two tests.

3.4.1 BMP Assay

When the reactivity or stability of a certain waste for landfilling is to be considered, the main interest is its gas generation potential under anaerobic conditions, which is the situation in the landfill. This gas generation potential can be determined by the laboratory fermentation test called BMP assay. Generally, the BMP assay involves the incubation of a small amount of representative test sample under controlled anaerobic conditions with gas production measured at the same time. The biochemical principles of this assay are explained in detail in Section 2.1.1. In a BMP assay, the production of the end products of microbial metabolism (notably methane) is used as an indication of microbial activity.

The BMP assay has been developed as a standardized method to determine the ultimate biodegradability and associated methane yield during the anaerobic methanogenic fermentation of organic substrate (Owen and Stuckey, 1979; Shelton

and Tiedje, 1984; Owens and Chynoweth, 1993; Eleazer *et al.*, 1997; Hansen *et al.*, 2004). It provides a more accurate prediction of the maximum yield of methane from the test material than could be predicted by a simple equation based on the mass of degradable matter in the test material, because the mass of actual bioavailable carbon is difficult to know.

Although the basic approach of BMP is similar, the technical approaches in terms of pre-treatment of the sample, inoculums, gas measurement technique and duration of the test vary significantly among the published methods, which depend on the purpose of study and the type of waste samples measured (Miller and Wolin, 1974; Owen and Stuckey, 1979; Jae Kyoung Cho *et al.*, 1995; Gunaseelan, 2004; Hansen *et al.*, 2004). Eleazer *et al.* (1997) measured methane yields of MSW components as a way to characterize the anaerobic biodegradability by long term incubation (more than 100 days to more than 600 days). The selection of the MSW components was based on the composition represented by the EPA waste characterization (EPA, 1994), which is shown in Table 3.4. The food waste (300.7 ml CH_4/dry g) measured showed the highest methane potential followed by office paper (217.3 ml CH_4/dry g) and the waste of leaves (30.6 ml CH_4/dry g) showed the lowest methane potential. They suggested that BMP data of each component may be used to evaluate the potential impact of the composition change in MSW on the methane yields from certain BMW.

For the purpose of monitoring performances of WDAs in BMW diversion, in Germany GB21 (or GS21) test, a 21 days of BMP test, is favoured because of its relatively short duration. This test is conducted with 50g DM of test sample, 100 ml microbial inoculum and 900 ml water in an air tight glass bottle incubated at the temperature of 35°C (Cossu *et al.*, 1999). Gas generation within 21 days is called to be GB21, which has been used as a characteristic indicator for the landfill gas production potential of pretreated waste (Binner and Zach, 1999; Binner, 2002; Soyez and Plickert, 2002). However, only the activity can be determined instead of

the biodegradation potential within 21 days because the total gas amount in 21 days was found to be only about 10% to 60% of the actual gas production potential (Binner *et al.*, 1999). Binner (2002) also summarized that the excessive lag phases caused by acid accumulation can lead to lower findings of GB21, although the parameter GB21 is very suitable for characterizing the reactivity of the waste. Therefore, in some studies, the test duration is extended to be 21 days plus the duration of the lag phase (Bockreis, *et al.*, 2007).

In the EA monitoring guidance (Environment Agency, 2005), the BMP test of BM100 is suggested, which is based on the 'milk bottle' anaerobic sludge degradability (SCA 1977) and GB21 test (Binner *et al.*, 1999). In BM100, the dried (at 70 °C) and ground BMW sample (equivalent to 20 g LOI) is mixed with 200 ml mineral aqueous medium and 50 ml seed sludge (of which DM is about 6%) in a small vessel and incubated anaerobicly at 35 °C. The biogas produced is recorded during the test and the test period may be up to 100 days or more depending upon cessation of gas production. Godley *et al.* (2007b) reported that the BM100 measured was in the range of 380-420 L/kg LOI for the untreated BMW and 105 L/kg LOI for the fully composted BMW. A common problem of BM100 is that if the waste is very biodegradable the test may become acid very quickly because of the imbalance between acidogens and methanogens, causing the inhibition of biogas production (Environment Agency, 2005). Additionally, from the practical and landfill management point of view, this time-consuming and costly test is not suitable (Laine-Ylijoki, 2004; Environment Agency, 2007).

Godley *et al.* (2005) studied the anaerobic biodegradability of eleven representative BMW components likely to be found in or diverted from MSW using the BMP test of BM100. The tested BMW components were newspaper, corrugated paper grass, twigs vegetables, meat, cotton, wool, nappies, compost (composted municipal yard waste) and pure cellulose, among which the meat had a much higher biogas production potential (633 L/kg LOI), followed by the corrugated paper (320 L/kg

LOI). The biogas production of the compost from municipal yard waste was 24 litre/kg LOI. Additionally, the order of biodegradability matches the studies of Harries *et al.* (2001) and Owens and Chynoweth (1993) for grass, corrugated paper, food wastes, newspaper, twigs (branches) and wool.

Based on the investigation of which types of anaerobic conditions could support or accelerate biodegradation of solid waste by means of the BMP assay, Chynoweth *et al.* (1993) summarized the ideal conditions for anaerobic decomposition in the case of the BMP test. These conditions include:

- inoculum Broad spectrum;
- Excess inoculum;
- Excess nutrients and substrate concentration below inhibitory levels;
- Excess buffering capacity;
- Moderate temperature; and
- Strict anaerobic conditions.

They also concluded that in order to accelerate waste feedstocks conversion to methane in the BMP assay, it was important to employ an inoculum-to-feed ratio of 2.0 or greater, and that particle size did not influence conversion rate in the range of 1-8 mm. Complete degradation of solid waste can be achieved in a period of a few weeks.

3.4.2 Respiration Test

Respirometric tests are a group of tests to measure the consumption of oxygen or production of carbon dioxide of organic matter, thereby providing a measure of biological stability. Among the numerous indices proposed for the evaluation of biological stability, the respiration index is now considered a suitable method and is also the standardized method (ASTM, 1996). The analytical procedures proposed for measuring the Respiration Index (RI) can be subdivided into dynamic and static methods according to whether the measurement is carried out with or without continuous aeration of the biomass (Scaglia *et al.* 2000; Adani *et al.*, 2004). Static

methods suffer from the disadvantage that they do not allow oxygen to be dispersed throughout the biomass, thus limiting diffusion and mass transfer (Paletski and Young, 1995).The results of the respiration test are usually reported as either the oxygen consumption rate (RI), or the cumulative oxygen consumption over a set period, e.g. four days.

One of the largest assessments of compost stability was completed by Scaglia *et al.* (2000). The authors evaluated forty-five organic materials of different types and origin by means of DRI and SRI (static respiration index). The SRI was found to underestimate the oxygen consumption by a factor of two compared with the DRI. In addition, based upon a review of the various substrates tested, the authors established a maximum DRI stability threshold of 1000 mg O_2 kg^{-1} LOI h^{-1}.

The ASTM (1996) provides a standard method and basis for measuring oxygen consumption to determine the stability of compost. This method requires a well-stabilized compost inoculum for the test samples, which is expected to accelerate the initiation of the composting process, and the aerobic composting takes place in an environment where temperature, aeration and humidity are monitored and controlled. The test is carried out in an incubator, the temperature of which is capable of being maintained at 58°C (±2°C). This test method produces results of a cumulative amount of oxygen consumption per gram LOI in the samples over a four-day period. The rate of oxygen consumption can also be monitored.

Based on ASTM (1996), a dynamic respirometric method was proposed by Adani *et al.* (2001) and the published tests were performed on MSW and derived products using a new dynamic respirometer designed by Adani *et al.* (2001). In their tests, no inoculum was added to the test samples, which is different from the ASTM (1996). Another major difference is that Adani et al. (2001) used thermal insulation apparatus to reduce heat loss during composting process, instead of performing the

test at a preset temperature which was considered to be limiting the biological activity and not enabling DRI determined in a realistic situation.

Adani *et al.* (2004) performed DRI measurements on 16 organic wastes of different origin, composition, and degree of biological stability to validate the test method and expression of DRI, and to propose biological stability limits. Their trials lasted four days each and throughout this time oxygen concentration (v/v %) and DRI (mg O_2 kg^{-1} LOI h^{-1}) measurements were recorded hourly. The instantaneous DRI was calculated as:

$$DRI(mgO_2kg^{-1}LOIh^{-1}) = Q \times \theta \times \Delta O_2 \times Vg^{-1} \times 31.98 \times LOI^{-1} \times \theta^{-1} \quad (3.8)$$

in which : Q(L h^{-1}) is the airflow, θ is the time of the acquisition (one hour); ΔO_2 (ml L^{-1}) is the difference between the oxygen concentrations in the inlet and outlet air streams of the reactor; Vg (L mol^{-1}) is the volume occupied by one mole of gas at inlet air temperature, which is 24.07 L mol^{-1} at 22°C; 31.98 (g mol^{-1}) is the molecular weight of O_2, and LOI (kg) is the total organic solids present at the time of measurement (starting LOI). The results revealed that the instantaneous DRI profile a marked peak was evident for lower stability waste; in contrast, the profile became practically flat for samples with higher biological stability. It was concluded that the DRI described the biological stability in relation to waste type and age well. The DRI values calculated as a mean of the highest microbial activity in 24 hours, of 1000 and 500 mg O_2 kg^{-1} LOI h^{-1} were proposed to indicate medium (e.g., fresh compost) and high (e.g., mature compost) biological stabilities, respectively.

The respiration parameter DRI4 is defined as the respiration activity within 4 days of testing, or more specifically, the cumulative amount of oxygen consumed per gram (or kilogram) DM or LOI of test samples in 4 days in an apparatus designed for the respiration test (Binner, 2002; Laine-Ylijoki *et al.*, 2004; Godley *et al.*, 2005). The DRI 4 observed by Adani *et al.* (2004) was 129-150 g O_2 kg^{-1} LOI $96h^{-1}$ for the untreated MSW samples and 43-84 g O_2 kg^{-1} LOI $96h^{-1}$ for the 20 days composted

MSW samples. Compared to the long term BMP test, the DRI4 is more easily determined, and the analysis time is short and appropriate for technical purposes, such as landfill management. Furthermore, a good correlation of DRI4 with the characteristic indicator for the landfill gas production (GB21) was experimentally proven in many cases (Soyez and Plickert, 2002; Laine-Ylijoki *et al.*, 2004). Therefore, DRI4 has been recommended as a preferred parameter for stability evaluation. However, because lag phases exceeding four days can occur during respiration activity determinations, The DRI4 value can be misinterpreted in these cases and be lower than the true level (Binner *et al.*, 1999; Binner, 2002). As far as the time scale of the test is concerned, it was suggested by ASTM (1996) to consider extending the present test time (t>4 days) if oxygen consumption in the last day was higher than that in the previous day.

Godley *et al.* (2005) investigated waste biodegradability by the DRI method which was based on ASTM (1996). The tests were carried out at 35°C over 7 days using the green waste compost as a microbial inoculum and an alkaline trap method for monitoring CO_2 production. The results were calculated as the DRI4 and DRI7 (seven days) oxygen consumption values in terms of the DM and LOI contents of the waste. The meat waste (219 g O_2 kg^{-1} LOI $96h^{-1}$) measured showed the highest DRI4 value followed by vegetables (169 g O_2 kg^{-1} LOI $96h^{-1}$) and the DRI4 of the compost was 26 g O_2 kg^{-1} LOI $96h^{-1}$. Their investigation found that biodegradation was incomplete within the seven days incubation time; however a good correlation between the completed BMP and DRI results was found for the selected wastes. At the same time, the DRI test might overestimate the biodegradability of mixed BMW wastes that contain a high fraction of readily biodegradable organic matter. The author also suggested that the oxygen consumption could vary considerably with storage conditions and be affected by status of microbes already present. In this case, oxygen consumption could not reflect the inherent biodegradability of the test material. The summary and comparison of the test methods BM100 and DRI4 are given in Table 3.6.

There are some reference values recommended on the stability of treated MSW in terms of respiration activity and gas generation. For example, Soyez and Plickert (2002) found that for untreated material, the typical values of DRI4 are in the range of 30 to 50 mg O_2/g DM. Additionally, a sufficient biological treatment results in an DRI4< 5 mg O_2/g DM and in a GB21< 20 l/kg DM, which are found as the actual state of the art in MBT technology. They also proved that these values represent a limit level of environmental impacts, which is acceptable from an ecological point of view, and the degree of stabilization is similar to that of humified organic matters in top soils. Another limit value of DRI4⩽ 7 mg O_2/g DM was established in the 'Guidelines for Mechanical-Biological Pretreatment of Solid Waste' in Austrian Landfill Ordiance (Binner, 2002).

Table 3.6. Comparison of the test methods BM100 and DRI4

Methods	Principle	Sample preparation	Sample amount	Test duration	Requirement
BM100	Under anaerobic conditions, test, biogas generation is measured, based on measurement of anaerobic sludge degradability (SCA 1977) and GB21 test	Dried and ground BMW	20g LOI	≧100 days	BMP vessels (350 ml), inoculum, mineral medium, biogas collection apparatus
DRI4	In aerobic conditions, respiration activity (O_2 consumption or CO_2 production) is measured, based on ASTM (1996)	Dried and ground BMW	400g	4 days	Composting vessels (2.5 litre), inoculum, gas monitoring apparatus

3.5 Summary

The intimate association between lignin and cellulose/hemicellulose can greatly hinder the microbial degradation of cellulose and hemicellulose. The lignin

concentration in BMW has been found to be related to the rate and extent of BMW degradation. However, the degree to which this relationship can be modelled is still under debate and research. Because lignocellulosic material is the major constituent of BMW, efficient treatment of BMW in composting or anaerobic digestion means that biodegradation of lignocellulose is also needed. The investigation of the reduction and/or transfer of these lignocellulosic materials is therefore of great significance in the target of diverting BMW from landfill.

Each test reviewed has its advantages and limitations for indicating the waste biodegradability under the circumstance of monitoring BMW diversion from landfills. In the non-biological test methods reviewed, the gravimetric method for moisture, DM, LOI and ash analysis are essential and form the basis for general waste characterization. The TC/TN ratio can not indicate the biological stability of compost without considering the initial TC/TN ratio of the material. The maturity limits in terms of TC/TN also needs to be determined. As a chemical method, fibre analysis has the advantage of being much quicker and easier to perform than some biological tests although its reliability and repeatability have been reported to be questionable. Furthermore, it can provide direct and useful information on the waste composition and with regular analysis can highlight changes during BMW treatment processes.

The BMP method has been widely applied to determine the ultimate methane production for a variety of MSW and consistently showed the most reliability in estimating the ultimate diversion of BMW. However, the BMP assay takes a long time to run, and is complex to execute in large scale MBT operations. The DRI method is relatively rapid and also showed great promise in estimating the stability of BMW at any stage of treatment, but it has some limitations when the timescales for the test is considered because in some cases DRI4 can underestimate the overall biodegradability. Excessive lag phases had been found to occur in either GB21/BM100 or DRI4 test, leading to lower values in both tests. Although several different methods and indicators have been developed and proposed regarding to the

BMP or respiration activity of MSW, no ideal methods or standard indicators have been issued so far.

Relevant studies on the relationship between lignocellulose contents and the biodegradability of MSW focused more on the specific MSW components and no reliable quantitive relationships were suggested, although some relationships are statistically significant (Wang *et al.*,1994; Barlaz et al., 1997; Eleazer *et al.*,1997). As far as the MBT or BMW diversion is concerned, the investigation of whole BMW biodegradation, especially the biologically treated BMW or the material at the end of MBT, is more significant than investigation of the untreated individual component. Therefore, for the purpose of evaluation of BMW removal by MBT, the relationship between relative lignocellulose content during biodegradation and the corresponding biodegradability is still promising based on the existing researches. As a proven technique, the FibreCap method provides a more reliable and rapid method for the analysis of fibre contents in feeding stuffs and this method of fibre analysis has potential for use in measuring BMW diversion through various pretreatment options. However, to date it has not been used to assess MSW or BMW diversion. Therefore, the correlation between fibre contents and the degree of MSW stability needs to be further investigated using the FibreCap technique through comparison with biological tests (such as, BMP and DRI).

CHAPTER 4 PRELIMINARY STUDY ON DEGRADATION OF MSW AFTER MBT

In order to verify the suitability of the fibre analysis using FibreCap technique (reviewed in Chapter 3) for characterising MSW and the reduction in biodegradation potential of the MSW treated by MBT, a preliminary experiment was conducted to study the anaerobic degradation of both the mechanically and aerobically pretreated MSW. In this preliminary experiment, the composition of the mechanically and aerobically pretreated MSW was analysed, particularly the contents of cellulose, hemicellulose and lignin. The anaerobic decomposition of both partly composted and fresh mechanically pretreated MSW samples obtained from the same source was monitored by BMP test in a series of small scale reactor vessels. The carbon yield (measured as carbon dioxide and methane production) from the waste reactor vessels and changes in fibre contents of both the partly composted and fresh mechanically pretreated waste samples were mainly monitored and linked, in order to establish some relationship between cumulative biogas production and the change of fibre contents. This chapter presents the results of this preliminary experimental study.

4.1 Experimental Design and Materials

20kg (wet weight) of a pretreated MSW composted for approximately 6 weeks through windrow composting, was obtained by using composite sampling ((Thompson, 2001) in March, 2005 from White's Landfill, Poole, Dorset in England,

which operates a large scale mechanical pre-treatment and composting facility. In addition, 20kg (wet weight) of a representative sample of mechanically pretreated fresh waste retained at the beginning of the composting process and subsequently stored in a freezer (-18°C) was also obtained. The wastes were sorted into the categories of paper/board, plastic, textiles, food/yard waste, glass, metal and other materials, which is based on the categories reviewed in Table 3.1. The components of the partly composted and fresh waste are compared in Table 4.1. All of the samples were dried at 70 °C to constant weight and shredded to pass through a 10 mm sieve prior to being placed in the reactor vessels. The compositional characterization of the two waste samples is presented in Table 4.2.

Table 4.1. Components of original composted waste and fresh mechanically pretreated waste samples

Components	Fresh Mechanically Pretreated Waste	Composted Waste
Paper/Board	33.1	20.5
Plastics	15.7	25.6
Textile	6.0	6.1
Yard/Food	2.4	2.5
Glass	12.0	16.7
Metal	5.1	5.1
Other [a]	25.7	23.5

Unit used is % weight

[a] components less than 10mm

Table 4.2. Characterization of original fresh and composted waste

Composition	Fresh Waste (%)	Composted Waste (%)
LOI	78.0	64.7
Cellulose	48.6	32.8
Hemicellulose	7.4	2.8
Lignin	8.4	18.6

4.2 Reactors

The BMP reactors were made from 1-litre Nalgene bottles attached to a gas collection system that allowed the volume of gas to be measured. 100g (dry weight) of remixed waste was used in each reactor bottle according to the composition of the fresh or composted waste samples collected. Seven reactors (C1-C7) were filled with the waste sample that had undergone approximately 6 weeks of composting and three reactors (F1-F3) with the mechanical pretreated fresh waste.

To accelerate the degradation of the waste by methanogenic anaerobic bacteria, 700 ml of a laboratory prepared medium containing mineral-nutrients and trace elements together with an anaerobic sewage sludge (from Millbrook Wastewater Treatment and Recycling Centre in Southampton, UK) was added to each reactor. The methanogenic mineral medium was adapted from Florencio *et al.*(1995) and contained the following (mg per litre): K_2HPO_4 (330), NH_4Cl (280), $MgSO_4 \cdot 7H_2O$(100), $CaCl_2 \cdot 2H_2O$(10), $FeCl_2 \cdot 4H_2O$(2), H_3BO_3(0.05), $ZnCl_2$(0.05), $MnCl_2 \cdot 4H_2O$(0.5), $CuCl_2 \cdot 2H_2O$(0.038), $(NH_4)_6MoO_{24} \cdot 4H_2O$(0.05), $AlCl_3 \cdot 6H_2O$(0.09), $NiCl_2 \cdot 6\ H_2O$ (142), $Na_2SeO_3 \cdot 5\ H_2O$ (0.164), $CoCl_2 \cdot 6H_2O$ (2), EDTA (ethylene diaminetetracaetic acid) (1), 36% HCl (0.001 ml L^{-1}). The mineral medium had previously been sparged with N_2 to remove any trace of oxygen. Anaerobic sewage sludge was added to the mineral medium in the ratio of 1 part sewage sludge to 9 parts mineral medium to a the methanogenic seed. The procedures of setting up the BMP reactors are detailed in Section 5.4.4. All the reactors were incubated in a water bath at 30°C to promote mesophilic methanogenic conditions. Two further control reactors containing 700 ml of methanogenic mineral media were used to determine the volume of gas produced from the seed alone. The methods used for collecting and measuring the gas produced from BMP reactors are also detailed in Section 5.4.4.

4.3 Analytical Methods

Individual reactors were sequentially opened for sampling during the period of the experiment (C1-C7, F1, F2 and F3), as shown in Table 4.3. Solids and associated leachate were collected separately from each of the reactors by filtration through GF/C filters (Whatman). The leachate samples were analysed for pH, volatile fatty acids (VFA), TC and inorganic carbon (IC). The solids samples were analysed for cellulose, hemicellulose and lignin after the removal of the easily identifiable non-degradable materials (metal, glass, stone and plastics etc). The contents of LOI, carbon, nitrogen, cellulose, hemicellulose and lignin in the solid material are expressed on the basis of the solids weight after removal of the easily identifiable non-degradable materials, whilst the results of biogas production are expressed on the basis of the total solids weight without removal of the easily identifiable non-degradable parts.

Biogas composition was measured using a Varian 3800 chromatograph (GC) which is detailed in Section 5.4.5. The dry weights of the solids in each sampled reactor were determined by drying the samples in an oven at 70°C to a constant weight (to avoid burning certain categories of waste). Non-biodegradable materials (plastic, metal, glass and stone) were then removed by visual inspection from each sample which was then re-weighed. The LOI content of the original samples was measured by ignition at 550°C in a muffle furnace for two hours. Waste samples for solids analyses were ground in a Foss Knifetec 1095 mill and then in a Foss Cyclotec mill to pass through a 1.0 mm sieve. The procedure of fibre analysis by FibreCap technique is detailed in Section 5.4.6.

The pH of the leachate in each dismantled reactor was measured using a Jenway Model 3010 pH meter (accuracy of±0.01). The leachate used for VFA measurement was centrifuged followed by the addition of 100% formic acid at a sample/acid ratio of 10:1 (v : v). Acidified samples were stored at 4°C.

Concentrations of VFA were determined by a Shimadzu GC-2010 incorporating a Flame Ionization Detector (FID) using a BP21 column. Injection temperature was 100°C, detection temperature was 250°C. A temperature ramp of 10°C /min from an initial column temperature of 60°C was applied to a final temperature of 180°C. Octonoic acid was used as an internal standard. TC and IC in the leachate were determined by a High-temperature Dohrmann-Rosemount DC-190 total organic carbon (TOC) analyser.

4.4 Results and Discussion

4.4.1 Gas Volume and Composition

The biogas yield attributable to the waste in each reactor was determined by subtracting the measured biogas yield of the control bottles from the total biogas produced by each reactor. The volume of biogas collected was converted to the volume at standard temperature and pressure (STP), which is detailed in Section 5.4.4.

The mean daily gas production rates and cumulative biogas yields of the reactors for the partly composted and fresh waste are presented in Figures 4.1 and 4.2. For the partly composted waste, the gas production rate was greatest on the 6-7th day of incubation, at about 1.4 ml $g^{-1}d^{-1}$. It then dropped to about 0.2 ml $g^{-1}d^{-1}$ after about 20 days. After that the reactors produced gas consistently but at less than 0.1 ml $g^{-1}d^{-1}$. After day 173, gas production had effectively ceased as observed in both the control and test reactors.

For the fresh mechanically pretreated waste, the gas production rate was highest (6.4 ml $g^{-1}d^{-1}$) on day 1 then dropped rapidly to zero after three (F1) or four (F2, F3) days. There was a recovery after 30 days and gas was produced at a rate of

approximately 4 ml $g^{-1}d^{-1}$ between days 57-62. Gas production was then constant until day 130. By day 173 the gas production rate had dropped to approximately 0.2 ml $g^{-1}d^{-1}$.

The total cumulative biogas production and composition for the fresh and partly composted waste following 173 days of anaerobic degradation is given in Table 4.4. The methane produced by the fresh waste is about 4 times that produced by the composted waste, and indicates that the amount of carbon degraded during composting pre-treatment accounts for a loss of 74% of the potential methane yield. This confirms that aerobic pre-treatment can be used to substantially reduce landfill gas emissions and therefore reduce biogas recovery costs. However, the loss of the potential methane could be seen as a major drawback of aerobic pre-treatment if methane is a desired by-product for energy recovery.

4.4.2 pH and VFA

The pH measured in the leachate of each reactor after dismantling is presented in Table 4.3 for both the pretreated fresh and partly composted waste samples.

Eight separate VFAs (acetic, proprionic, iso-butyric, n-butyric, iso-valeric, n-valeric, hexanoic and heptenoic acids) were measured in the leachate. For the fresh waste, the total VFA concentration reached 4783 mg/l on day 8 of which acetic acid comprised 1936 mg/l and propionic acid 1127 mg/l. The pH fell to 5.2 due to acid accumulation. Total VFA concentrations increased to 7630 mg/l by day 26 with acetic acid comprising 3117 mg/l and propionic acid 2040 mg/l. For the partly composted waste sample, acetic acid and/or propionic acid were only detectable up to day 14 with average values of 16 mg/L and 64 mg/L, respectively. The pH remained constant at about 7 and the VFA concentration dropped to zero between day 14 and 21. No acid accumulation was observed in the anaerobic degradation process of partly composted waste sample.

Table 4.3. Leachate pH, solids composition and cumulative biogas yields of sampled reactors

Day of termination (reactor)	pH	Mass remain (Dry wt., g)	NDF (%)	ADF (%)	Lignin (%)	(C+H)/L	Measured biogas (L/kg)
Composted							
0	7.3	100.0	54.2±1.0	51.3±1.4	18.6±0.9	1.9	0.00
4 (C1)	7.1	99.1	54.3±0.9	50.3±1.8	18.1±1.0	2.0	0.18
7 (C2)	6.9	97.8	54.2±0.5	49.6±0.4	18.4±0.5	2.0	0.65
14 (C3)	6.9	96.7	53.8±0.6	49.2±1.3	19.5±1.0	1.8	1.15
21 (C4)	7.0	95.7	44.8±0.6	40.5±0.5	17.5±0.9	1.6	1.53
70 (C5)	7.0	94.4	44.1±0.3	40.1±0.4	20.4±0.3	1.2	1.89
119 (C6)	7.3	94.3	48.4±1.1	44.1±0.1	22.6±1.0	1.1	2.03
173 (C7)	7.4	94.0	42.8±0.9	39.0±1.1	20.5±0.3	1.1	2.06
Fresh mechanically pretreated							
0	7.3	100.0	64.3±1.3	57.0±0.9	8.4±0.9	6.7	0.00
8 (F1)	5.2	93.0	56.4±0.6	48.6±0.6	7.9±0.5	6.2	0.32
26 (F2)	5.1	92.5	---	---	---	---	---
173 (F3)	7.3	72.8	44.9±0.7	40.6±0.9	17.6±0.4	1.6	93.7

Table 4.4. Comparison for methane production for composted and fresh waste

Waste	Days	Cumulative gas ($CO_2 + CH_4$, ml/g)	CO_2 (%, by volume)	CH_4 (%, by volume)	CH_4 (m^3/t)
Composted	173	20.6	27.0	73.0	15.0
Fresh	173	93.7	37.0	63.0	59.0

Compositional analysis (Table 4.2) of the fresh mechanically pretreated and partly composted waste samples indicates that approximately 14% of the initial LOI were removed by the initial composting process. These parts of LOI can be designated as the most readily degradable compounds. ten Brummeler and Koster (1990) suggest that at least 20% of the LOI needs to be converted during the aerobic composting

stage before VFA consumption is in balance with methane formation. Since no accumulation of VFAs was observed during the anaerobic degradation process of partly composted waste samples, the use of the methanogenic seed may be viewed as being highly effective in stimulating rapid methanogenesis, and that the LOI reduction during an initial aerobic degradation stage does not need to be as great as the 20% suggested by ten Brummeler and Koster (1990). These observations also confirm that the anaerobic acidogenic (VFA formation) stage in a landfill could be reduced and the start-up of anaerobic degradation accelerated or indeed kick-started by an aerobic pre-treatment stage (reviewed in Section 2.3.2). Therefore, aerobically treated waste can potentially be considered for in-vessel anaerobic degradation processes as a possible option to enhance methanogenic conditions and further reduce the leachable organic emissions from the residue when finally disposed to landfill.

4.4.3 Fibre

Table 4.5 presents the fibre contents and LOI of the sorted organic components in the fresh mechanically pretreated waste and partly composted waste. In the components of the fresh mechanically pretreated MSW, the lignin content of newsprint was higher than that of the other paper. The (C+H)/L ratio of the other paper was the highest among all the components. After composting, the lignin content increased resulting in reduced (C+H)/L ratios due to decomposition of cellulose and hemicellulose during the aerobic treatment process.

The mean results from the fibre analyses for the composted waste are presented in Table 4.3 and Figure 4.3, as NDF%, ADF % and the cellulose, hemicellulose and lignin contents of the solids after removal of the easily identifiable non-degradable material. The range of results indicated in Figure 4.3 shows that the method employed for the fibre analysis provided very consistent results. As presented in Table 4.3, the percentage content of cellulose, hemicellulose and the (C+H)/L ratio

of the partly composted waste (day 0) were much lower than those of the fresh mechanically pretreated waste (day 0) and the difference represents the overall change in fibre content due to the composting process. The hemicellulose content decreased from 7.4 % to 2.8 % and the cellulose content from 48.6 % to 32.8 % as a result of the composting process. The lignin content was 8.4 % in the original pretreated fresh waste, increasing to 18.6 % in the composted waste due to the decrease in relative easily degradable fraction, such as hemicellulose and cellulose.

Table 4.5. Characterization of components of original fresh and composted waste

Components	LOI (% Dry wt.)	Cellulose (% Dry wt.)	Hemicellulose (% Dry wt.)	Lignin (% Dry wt.)	(C+H)/L
Fresh					
Newsprint	82.3	46.0	7.1	9.7	5.5
Other paper	81.0	52.2	8.8	4.6	13.4
Yard/Food	97.6	31.5	8.6	14.3	2.8
Textile	84.3	46.1	2.0	18.1	2.7
Composted					
Paper/Board	62.5	30.5	2.9	13.8	2.4
Textile	82.8	38.8	0.1	31.2	1.3
Yard	91.5	36.5	9.1	27.0	1.7

There are some discrepancies in the results from the anaerobic degradation of the partly composted waste, e.g., the ADF and NDF values on day 119 (s.d. = 0.1% and 1.1%, respectively) were higher than that on day 21 (s.d. = 0.5% and 0.6%, respectively) and day 70 (s.d. = 0.4% and 0.3%, respectively). This is likely to be a result of the heterogeneous nature in the waste and the sampling. However, for the most part clear trends of diminishing ADF, NDF and cellulose are discernible through the anaerobic digestion period. As shown in Figure 4.3, the percentage ADF and NDF values show a decreasing trend, whilst the percentage ADL (lignin) increased, which is as expected as lignin is a recalcitrant material (Stinson and Ham, 1995). The hemicellulose content of the pretreated composted waste changed very little during anaerobic degradation in the reactor vessels indicating that nearly all the

biodegradable hemicellulose was removed during initial composting. After 173 days of anaerobic degradation, the (C+H)/L ratio of the partly composted waste dropped from 1.9 to 1.1, while the (C+H)/L ratio of the fresh waste dropped from a very high 6.7 to 1.6.

The relationship between (C+H)/L ratio of the partly composted samples and their cumulative biogas yields during anaerobic degradation is presented in Figure 4.4. It showed that when the ratio of (C+H)/L decreased from 1.9 to 1.1 during anaerobic degradation, a cumulative biogas volume of 2 L/ 100 g MSW was produced correspondingly. There was a significant linear correlation (r^2=0.88, P<0.01) between the (C+H)/L ratio and cumulative biogas yield measured as the partly composted waste samples continued to produce gas under anaerobic conditions. It indicates that the change of the (C+H)/L ratio of MSW was related to the degree of MSW biodegradation. The (C+H)/L ratio at day 0 was slightly lower than that on day 4, which, as stated earlier, may be caused by small inherent differences in sample composition between individual reactors. It can be concluded upon examination of the data of ADF, NDF values and the contents of cellulose, hemicellulose and lignin that (C+H)/L ratio is a more precise indicator of the extent of decomposition than the total fibre content or total solids.

Figure 4.5 shows the biogas potential as a function of (C+H)/L ratio for both the pretreated fresh waste and the partly composted waste samples. It was assumed that the cumulative biogas produced after 173 days of incubation (BY_{173}) was the biogas potential for each type of waste, i.e. biogas potential of 9367 ml for 100 g of mechanically treated fresh waste sample and 2063 ml for 100 g of composted waste. The biogas potentials for the other waste sampled during anaerobic degradation were calculated using Equation 4.1.

$$BP_n = BY_{173} - BY_n \quad (4.1)$$

BP_n: biogas potential of the waste sampled at day n; BY_{173}: the cumulative biogas production at the end of tests, different for two types of samples; BY_n: cumulative biogas production measured at day n.

It can be seen that there is a significant correlation (r^2=0.99, P<0.01) between the change of (C+H)/L ratio and the biogas production potential during both processes of biological pretreatment and following anaerobic decomposition. Figure 4.5 also highlights the overall reduction in (C+H)/L ratio of the mechanically pretreated waste that had been composted for approximately 6 weeks as part of a commercial MBT process. It is clear that not only has the initial mechanical pre-treatment process substantially elevated the fibre composition of the waste but that the main degradable components of the waste (cellulose and hemicellulose) diminish rapidly during subsequent composting.

4.5 Summary

The preliminary experimental results demonstrate that fibre analysis using the FibreCap technique may provide reliable measurements of fibre contents in BMW. A strong correlation between biogas production potential and (C+H)/L ratio was established during decomposition and the diversion of degradable fraction by composting treatment was illustrated by the change of the (C+H)/L ratio. All these data show that fibre analysis has a real potential as a direct measurement to demonstrate BMW diversion. In the further work a larger variety of waste samples should be investigated and more evidence should be presented.

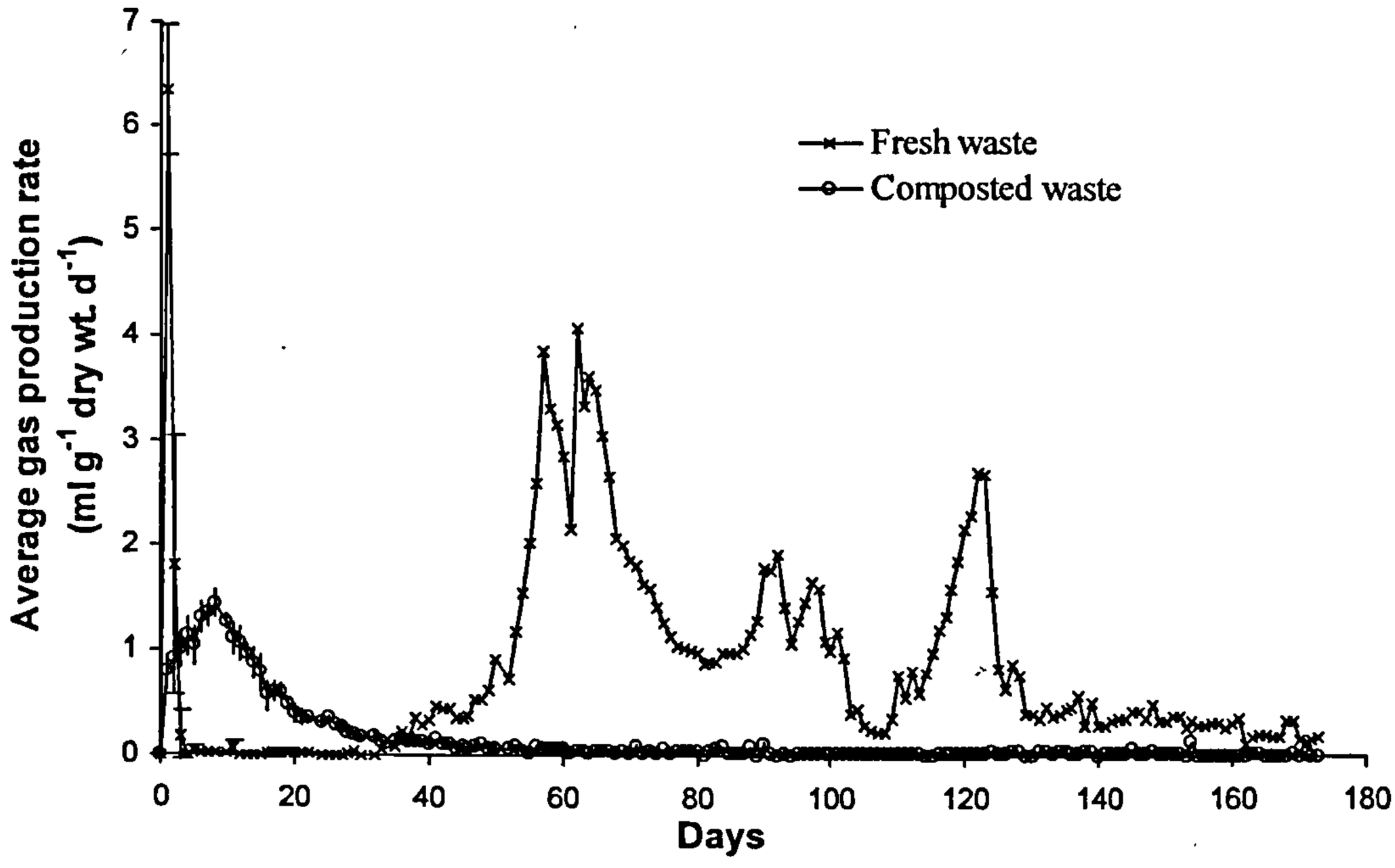
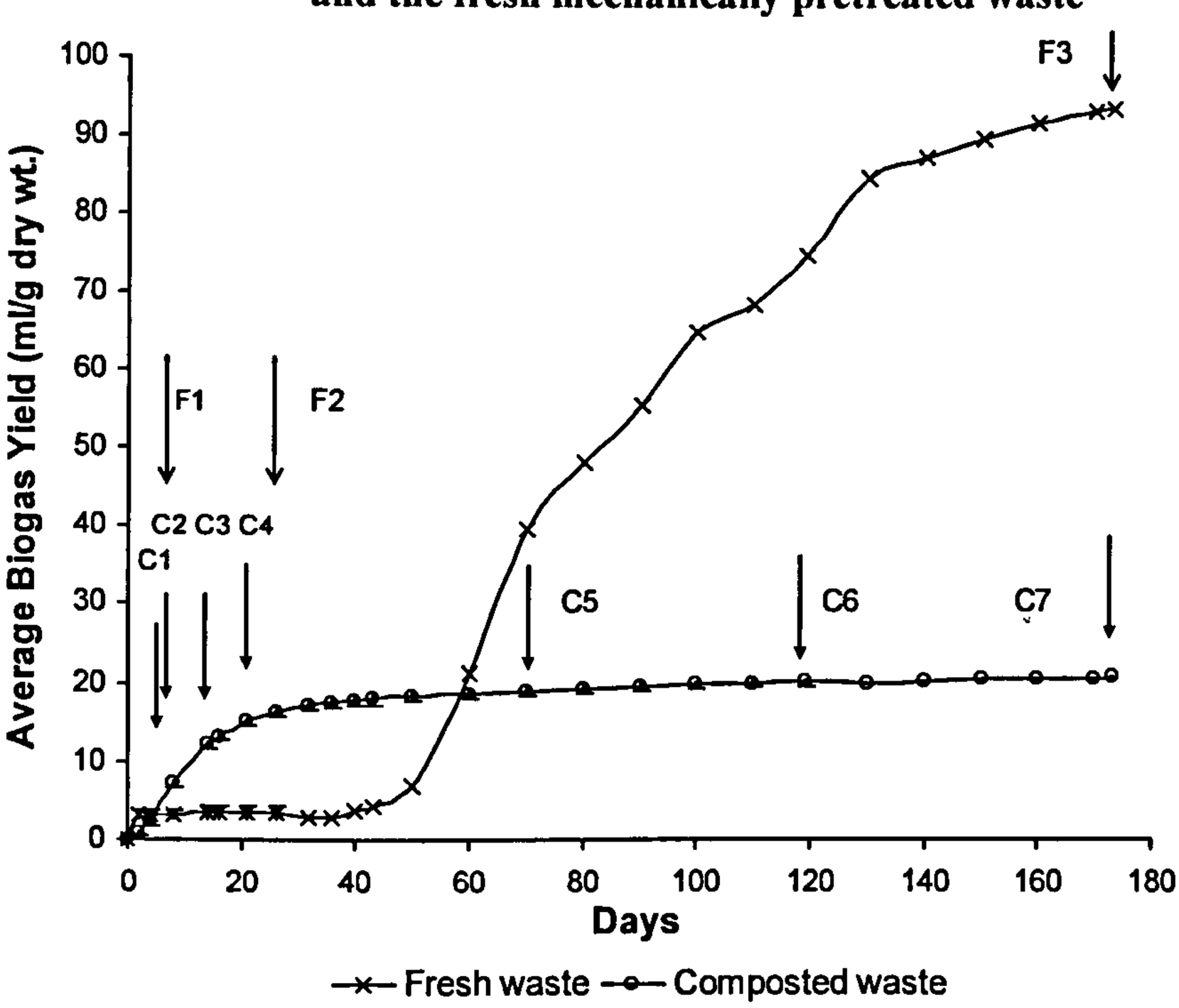


Figure 4.1. Mean daily gas production rate of the reactors for the composted waste and the fresh mechanically pretreated waste



Reactor C1-C7 (composted waste) and Reactor F1-F3 (fresh mechanically pretreated waste) are as numbered in Table 3.

Figure 4.2. Mean Cumulative gas yields of the remaining reactors for fresh and composted waste

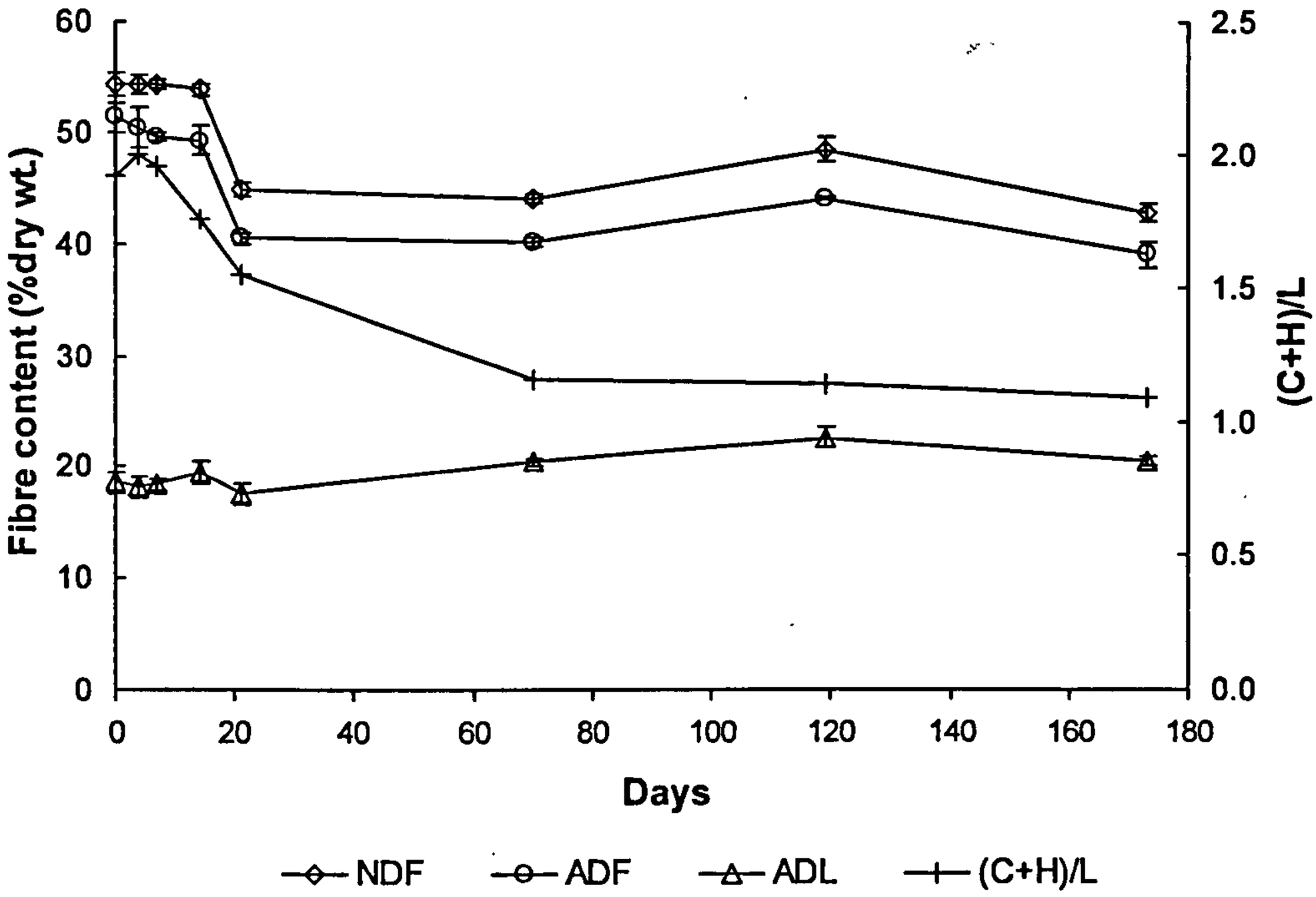


Figure 4.3. Fibre content and (C+H)/L ratio in the sampled reactors of composted waste

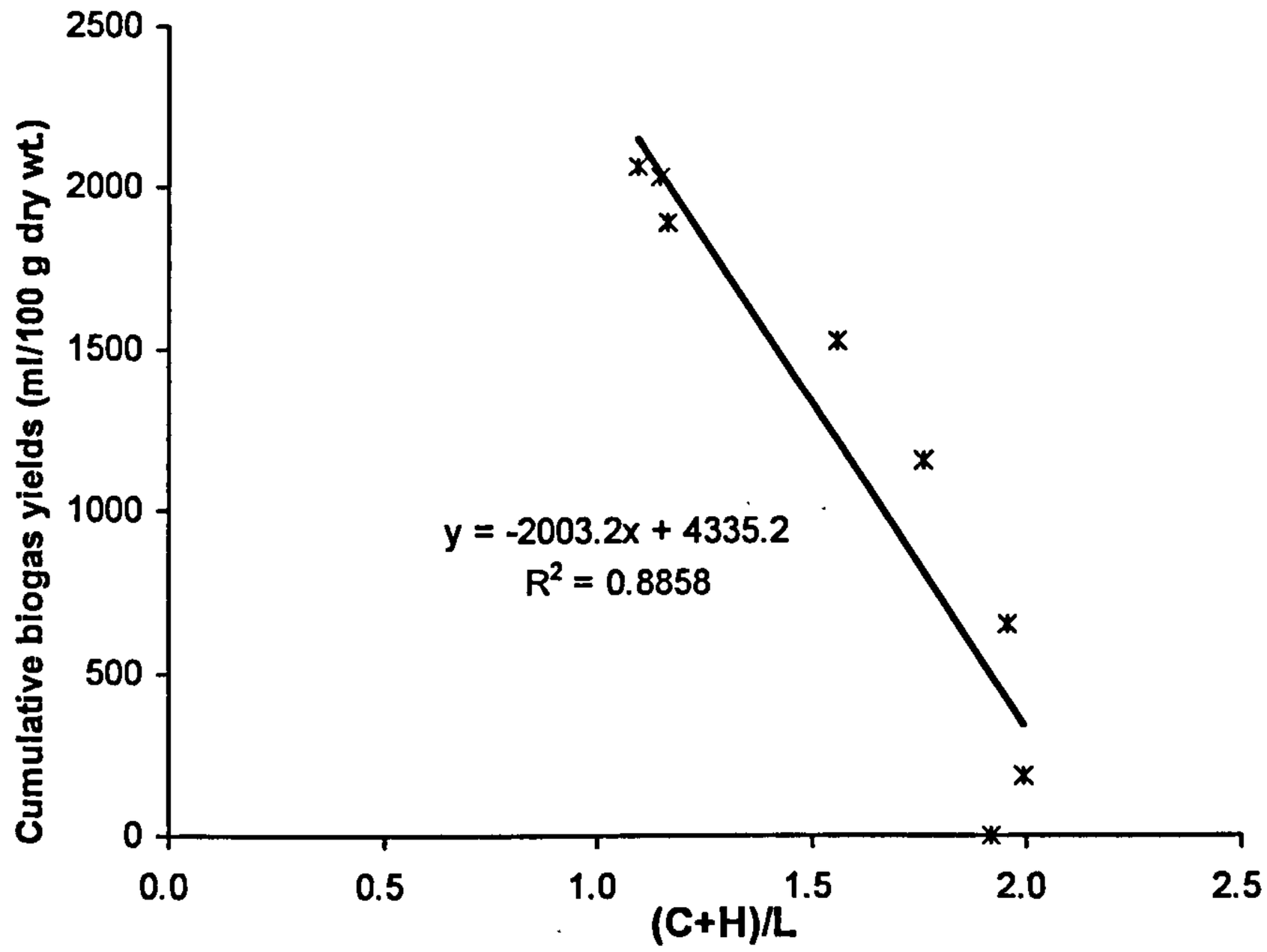


Figure 4.4. Relationship between (C+H)/L ratio and cumulative gas yield of composted waste

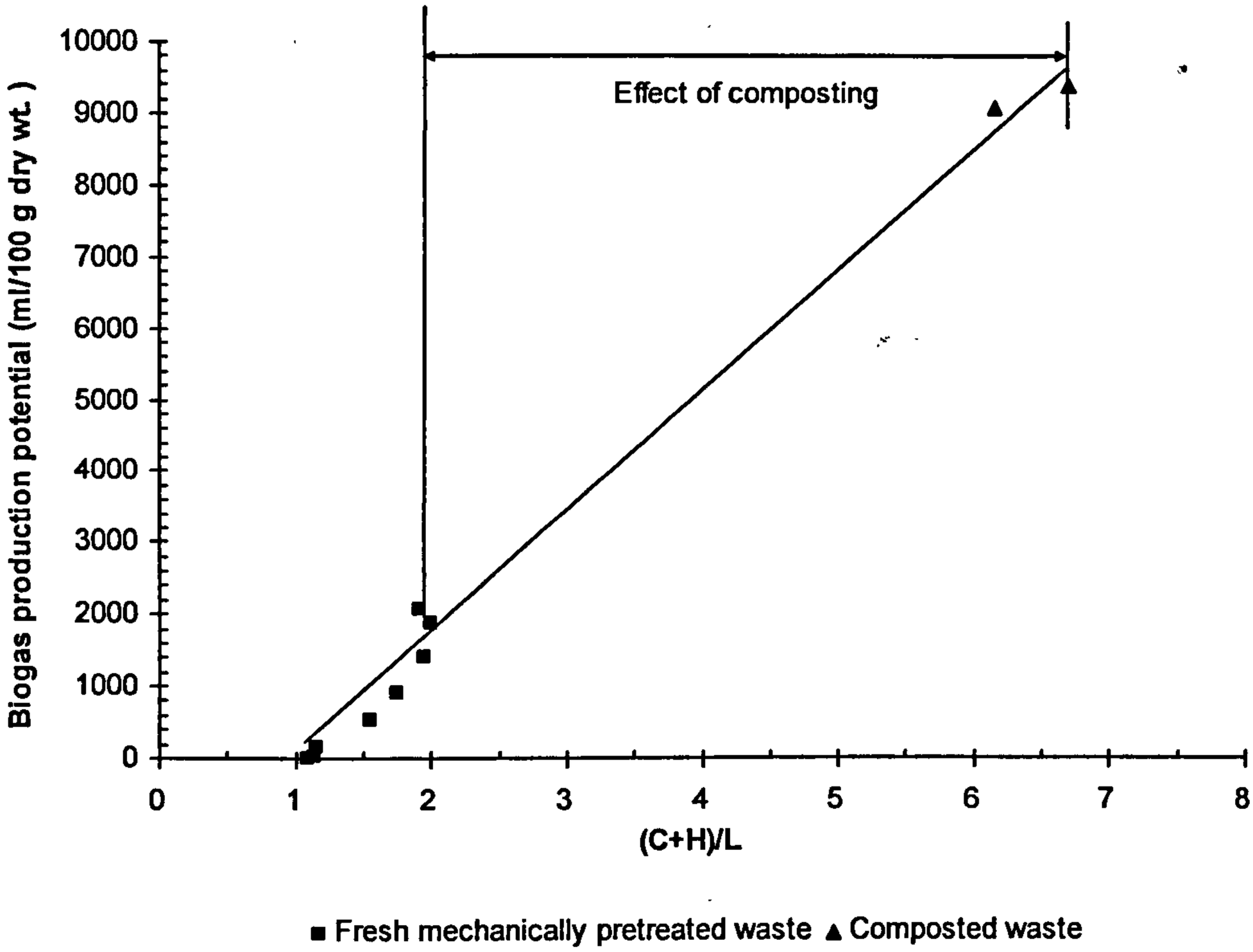


Figure 4.5. Change in (C+H)/L with biogas production for mechanically pretreated MSW during anaerobic degradation

CHAPTER 5 MATERIALS AND METHODS

5.1 Introduction of Research Methodology

In order to investigate whether fibre analysis can be used in the assessment of the biodegradable content of the MSW that has been removed by a treatment process prior to being landfilled, a correlation between fibre contents and the degree of biodegradation needs to be identified. The key methodology is to conduct fibre analysis in parallel with established biological methods, which are the BMP assay and DRI measurements using the same fresh or treated BMW samples, and comparing the results between these tests.

Therefore, fibre analysis by the FibreCap method, BMP assay and DRI measurement were conducted in parallel using a variety of fresh BMW and aerobically treated BMW samples. The aerobically treated samples were obtained from two composting reactors designed to perform simultaneous DRI measurement. The design of the composting experiments was based on the review on the composting process in Chapter 2. Periodical sampling from the reactors allowed the fibre compositional changes to be tracked and correlated with corresponding BMP and DRI measurements. Additionally, the solid samples were analysed for DM, LOI, TC and TN contents to characterize the change in waste composition. These tests were carried out on the basis of the reviews in Section 3.3 and 3.4. Figure 5.1 illustrates the specific analysis and sequential procedures for each experimental trial. This chapter describes:

- The setup and operation of laboratory-scale composting reactors;
- The sampling procedures;
- Methods of analysis and tests;
- The material used and description of composting runs conducted. The technical parameters and calibration methods applied to the sensors used in monitoring the composting processes were detailed in Appendix A.

5.2 Laboratory Composting Reactor System

To simulate and replicate the composting processes used to stabilise mechanically pretreated MSW, two laboratory scale composting reactors were built for batch composting. The reactors were designed to accept approximately 6-7 kg BMW samples at a moisture content of about 60-70%. During the composting process, samples were collected periodically from the reactors and analysed to track the change of the organic fraction and the degree of stability. In addition, the reactor system was also designed to allow oxygen consumption, CO_2 production and temperature to be measured throughout the composting process. The design of the reactors and their operation were based on the review of laboratory reactors by Petiot and Guardia (2004) and the computer controlled laboratory composters of Stringfellow (1998). The laboratory composting system used in this study comprised two identical compost reactors, a computer control and data recording system.

5.2.1 The Composting Reactors

Figure 5.2 presents the schematic diagram of the reactor set-up and Figure 5.3 shows the picture of the whole system. Each reactor was fabricated from a stainless

steel container with a diameter of 340 mm and depth of 350 mm. The stainless steel container was chosen because it is a good heat conductor and resists corrosion. A plastic pot with a capacity of 18 litres was set inside the stainless steel container to contain the composting waste so that there was a 5 mm gap between the pot and the container. The purpose of this was to even out the temperature in the environment surrounding the inner pot. A plastic circle stand 60 mm deep was put inside on the bottom of the steel container to elevate the pot and to leave space for aeration and condensed water vapour to accumulate. The bottom of the pot had been removed and replaced with a perforated stainless steel plate with an open area of 79% to distribute the air supplied from the bottom of the steel container and to support the weight of the composting substrate. A tap was fitted to the bottom of the container so that condensed water that could accumulate within the container could be removed and the volume measured. The container was sealed with a rubber gasket inside the rim of the lid.

A Holroyd Insulated Drum Heater Mat (Model HISD-A) was fixed around the container to reduce heat loss. The heating mat is designed to be used on standard plastic and steel drums for the warming of the drum contents.

A pump was used to supply air into the reactor to meet the respiration needs of the microorganisms present during composting through a gas tube connected to the bottom of the reactor. Key Instruments MR3000 flowmeters with the ranges of 0.1-1.2 LPM and 0.4-5.0 LPM of air and the accuracy of ±4% full range were used to measure the air flow rate. Additionally, a valve was connected which incorporated the flowmeter to regulate the airflow. Three ports were located in the lid of the reactor for an exhaust gas tube (diameter of 8mm), humidity transmitter and temperature probes. An EE0-FTB4A3 humidity transmitter (Sensors and Transmitters UK, Ltd) was installed on the lid to monitor the humidity within the reactor. The exhaust gas on the top went first through a water vapour trap made from a conical flask. The CO_2 and O_2 concentrations of the exhaust gas were monitored by a BCP CO_2 sensor (BlueSens Company, Germany) and an M-09 O_2-

Medical sensor (Sensors and Transmitters UK, Ltd) connected to a manifold system (Figure 5.4) and linked to the exhaust gas tube.

Three thermistor temperature probes were placed individually at the centre (Tc), the quarter point of each reactor (Tq) and in the gap between the inner vessel and the stainless steel container (Tg). They were always located at the mid height of the substrate. All the sensors and probes used were calibrated before each composting run. The related technical parameters and calibration methods used for these sensors are presented in Appendix A.

5.2.2 The Computer Control and Data Recording System

The computer control and datalogging system comprised a Datataker model DT 505 and was used to monitor the CO_2 and O_2 concentrations in the exhaust gas, the temperatures (Tc, Tq and Tg) and the humidity inside the reactor, and on a feedback loop to control the temperature of the external heating mat. The schematic diagram of the control system is shown in Figure 5.5.

The DT 505 has 10 analog and 7 digital channels which are multipurpose. A channel expansion module was attached to the Datataker to provide increased channel capacity through an expansion connector. All the sensors and probes were connected to the datalogger using individual channels. To automatically control the temperature of the heating mats, two power relays (10A 5Vdc) were connected and controlled by the datalogger.

The datalogger was connected to a personal computer via a RS232 COMMS connector and programmed by the Delogger 4 software, which is a data acquisition package that operates in a Windows environment. The program had the following main functions:

- Converting analogue outputs to digital information and displaying the measurements according to the instrument calibration factors.

- Setting the data acquisition intervals and data storage intervals. For example, the datalogger was set to acquire data every minute, and to record and display the average values in 30 minute intervals depending on the speed of the biological composting process and the amount of data stored. All the measurements were stored in a memory card and uploaded to the computer.
- Setting the temperature regime for controlling the heating mats. As mentioned above, the function of the heating mats was to reduce the heat loss during composting and also to enable the normal self-heating of the composting substrate. The heating temperature of the heating mat was set at 55°C. Therefore, in the computer programme the heating mats were switched on when (1) the temperature of the substrate core (Tc) was at least 0.5°C greater than that in the gap (Tg) between the reactor wall and inner vessel, that is, in the case of heat loss (2) the temperature of the substrate core (Tc) was not greater than 55°C. In other words, the heating mats were switched off when the temperature loss of self-heating was less than 0.5°C or the temperature of the substrate core (Tc) was greater than 55°C.

5.3 Material and Sampling Procedures

5.3.1 Preparation of Composting Material

Four batches of fresh MSW samples used for composting in the laboratory were obtained at various time from Otterbourne waste transfer station operated by Veolia Hampshire Ltd (Otterbourne, Winchester, Hampshire, UK), and serving residential kerbside collections from the Southampton, Eastleigh and Winchester areas of the county. The MSW was identified to be from the Winchester area where a separate source segregated kerbside collection of dry recyclable materials also operates. Therefore, in theory the material collected by the refuse collection vehicle (RCV)

should have a reduced content of plastic bottles, newsprint, metal cans and glass which are the targeted materials for the separate collection. Approximately 400 kg of roughly mechanical treated MSW were taken each time as a representative sample of the material discharged from RCV and transported to the lab. This was done by using a mechanical shovel and placing the material in an open area where a primary sorting and mixing took place which was to remove obvious bulky non-biodegradable wastes, such as nappies, textiles, electrical appliances and construction material residues.

In each collection, approximately 30 kg MSW was taken from the initial 400 kg MSW by using composite sampling (Thompson, 2001; Environment Agency, 2005). A flow chart demonstrating the sampling process is presented in Figure 5.6. The 30 kg samples were hand sorted into biodegradable organics including paper/board, yard waste and kitchen waste, and macroscopically visible plastic, metal, glass, textiles, and leather materials were removed. The compositions of the four batch MSW samples and the sorted BMW samples are given in Section 5.5, as well as the time of MSW collection from the transfer station. The sorted BMW were then shredded to less than 10 cm and stored in a freezer (-18 °C) until required.

Before filling the composting reactors, the components (paper, yard waste and kitchen waste) of BMW were spread out on a plastic cover on the floor and well mixed using a shovel. At the same time, water was added to reach a moisture content of about 60-70%, and inoculum was added, which was expected to accelerate the initiation of composting process. The details are given in Section 5.5. Samples for composting were obtained using the cone-and-quarter method for homogenization of the original BMW samples. The BMW was coned and quartered until sufficient BMW samples were obtained. In each run, 6-7 kg wet seeded BMW sample, which was the maximum amount each composting reactor could contain, was obtained to fill the reactor by dividing the BMW into quarters.

5.3.2 Sampling Technique and Procedure

Samples for analyses and testing were taken when preparing the composting material at the start and then from the reactors during composting. Considering the size of the composting reactors and the scale of tests and analyses undertaken, the homogenization of the waste was important to ensure representative sampling and reproducibility of the samples. Therefore, at the start, the components of BMW were well mixed before sampling. When samples were taken from the reactors, the composted mixtures were well mixed in the composting reactors. After size reduction of these samples, sub-samples were collected using the cone-and-quarter method for moisture, dry matter (DM), LOI, BMP, fibre and elemental determination.

The sampling sequence is also presented as Figure 5.6. At the start of composting, about 800 g well mixed BMW sample was collected. During composting and at the end, about 500 g samples were collected from each reactor at each interval. These samples were then shredded to a size less than 1 cm, and then re-mixed to further facilitate obtaining representative sub-samples for analysis and tests. About 160 g sub-samples were taken for moisture, DM and LOI determination. Approximately 100-200 gram sub-samples were retained for TC, TN and fibre content determination. These sub-samples were first dried at 45 ℃ to a constant weight, and then milled by a Foss Knifetec 1095 mill and then by a Foss Cycolotec mill to pass through a 1.0 mm sieve. About half of the milled sub-samples were then dried at 105 ℃ for fibre analysis. Sub-samples for BMP assay were stored in a freezer (-18 ℃) until required.

5.4 Tests and Analysis

5.4.1 DRI

This test is designed to be performed in two laboratory-scale composting reactors, the detail of which is described in Section 5.2. The procedure of filling the composting reactors is described in Section 5.3.1. The composting experiments are not only for determining DRI, but also for studying the degradation of BMW and the changes of solid composition during aerobic composting processes. Furthermore, a large ratio of inoculum to composting waste (such as 1:1, dry matter basis) would influence the composition analysis of the studied material. Therefore, in these tests a small ratio of inoculum to composting waste was used (the amounts are given in Section 5.6), which is one major difference from the other DRI tests described in the literatures (e.g. Gray *et al.* 1971; ASTM 1996; Komilis and Ham, 2000; Godley *et al.*, 2005).

Oxygen and carbon dioxide concentrations were monitored throughout the test and the instantaneous DRI calculated using Equation 3.8. The results are expressed on a LOI basis. The oxygen concentration in the exhaust gas was kept higher than 140 ml L^{-1}, which has been found to not limit the composting process (Adani *et al.*, 2001). This was achieved by manually adjusting the gas flow rate through the flow meter and the valve.

5.4.2 Moisture, DM, LOI and Ash Content

Moisture and DM of the samples were determined according to the standard method of ISO 11465. An accurately weighed sample was dried at 105°C to constant weight in a convection drying oven. DM is the amount of solids remaining after drying and its content is calculated by:

$$DM\% = (Weight_{dry\ sample} / Weight_{test\ sample}) \times 100 \quad (5.1)$$

Consequently, the moisture content can be calculated as:

$$Moisture\% = 100\% - DM\% \quad (5.2)$$

The LOI content was obtained by ignition of the weighed, dried samples at 550°C for two hours in a muffle furnace according to the test method described in TMECC (Thompson, *et al.*, 2001). The percentage ash content is calculated from the weight of the remaining ash while the percentage LOI content is calculated as the percentage of weight loss. The percentage LOI content is usually expressed on the basis of DM (Equation 5.3). These gravimetric tests were conducted in triplicate for each sample where 40-50 g sample was used for each replicate.

$$LOI\% = (1 - Weight_{ash} / Weight_{dry\ sample}) \times 100 \quad (5.3)$$

5.4.3 TC and TN

The fresh and composted samples were dried at 45°C to constant weight, milled and then analysed for total carbon and total nitrogen contents (described in Section 5.3.2). The total carbon and nitrogen contents were determined using a CE Instruments 1112 Flash Elemental Analyser (Thermo Finnigan) by dry combustion at 900°C in an oxygen atmosphere with 140ml/min helium carrier gas and TCD detection of the gases produced. This analysis was conducted in triplicate where 4-7 mg sub-samples were used in each replicate.

5.4.4 BMP

The BMP assays used for the purpose of biogas potential determination were designed according to the conclusions of Chynoweth *et al.* (1993) where the main conditions such as the inoculums, particle sizes and incubation temperature are

considered. Since there is a possibility of the inhibition of biogas production caused by acid accumulation in the BM100 and GS21, reviewed in Section 3.4.1, a larger ratio of inoculum-to-feed was use in the BMP test presented. According to Chynoweth *et al.* (1993) an inoculum-to-feed ratio of 2:1 was shown to give maximum conversion rates. Therefore, the inoculum-to-feed ratio was chosen at about 1.0 because the larger sample sizes could be used than the ratio of 2.0 and this ratio did not affect the rate of anaerobic degradation. The experimental setup for BMP assay in this study is shown in Figure 5.7 and Figure 5.8. The test outline is explained as follows.

Vessel/Reactor: 1-litre plastic bottles (HDPE, Nalgene Ltd.) were used with rubber bungs on the lids which were connected with three way valves and flexible plastic tubing to pass gas to the collection cylinder.

Inoculum: 700 ml sewage sludge was used for each BMP reactor as inoculum. Two control reactors containing only 700 ml sewage sludge each were used to determine the volume of gas produced from the inoculum alone.

Incubator: Water bath was set at 35 °C for BMP reactors.

Gas collection cylinders: The cylinders were calibrated in volume with three way valve at the top for connecting to a vacuum pump to remove gas produced by the BMP reactors.

Gas displacement liquid: Water was acidified to pH 2 with HCl. Acid-water was used to prevent CO_2 dissolution. The acid-water was displaced as the gas accumulated within the cylinders, which were calibrated independently.

Test BMW samples: By assuming the LOI content of sewage sludge was 0.02g/ml, 14 g LOI of BMW samples was used in each BMP reactor when using the inoculum-to-feed ratio of 1.0. Therefore, the actual amount of wet sample used for filling each BMP rector can be calculated using the percentage LOI content and

moisture content of the corresponding sample. The particle size of the waste sample was not greater than 10 mm.

Procedures: After mixing well the sewage sludge with the test sample in the reactor, the reactor was flushed with N_2 for about 5 minutes to remove oxygen and then sealed under an N_2 atmosphere before being placed in the water bath. No mechanical mixing of the waste was carried out during the test. The level of gas collected in the gas cylinder was recorded daily and the gas was removed from the gas cylinder by suction using the vacuum pump from the top of the cylinder. For each different sample tested, a BMP test was carried out in duplicate.

The volume of gas collected (V) was calculated from the decrease in height of the acid liquid level (H) in the collection cylinder and the area (A) of the cylinder, as expressed in Equation 5.1a. The volume of gas produced by the test sample (Vs) was then adjusted for the average sewage sludge gas production by Equation 5.1b. All gas production was converted to the volume at standard temperature and pressure (STP), which is at temperature of 0°C (273K) and pressure of 101.3 kPa (760 mm Hg). The reference temperature measured in the laboratory during the test was 22 °C. Therefore, the gas production at STP is converted by Equation 5.1c.

$$V = H \times A \quad (5.1a)$$

$$V_s = V - V_{sludge} \quad (5.1b)$$

$$V_s\,(STP) = (273/293) \times (p_a/101.3) \times V_s \quad (5.1c)$$

Where, p_a is the measured atmospheric pressure in the laboratory in kPa.

5.4.5 Gas Composition

The composition of biogas produced by the BMP reactors was measured daily in the first week after set up and every three to five days thereafter. The gas samples were

collected using hypodermic syringes inserted through the three way valves on the top of BMP reactors. The samples were immediately analysed using a Varian 3800 gas chromatograph (GC), operated isothermally at 50°C and incorporating a Molecular Sieve 13X, 60/80 mesh column and a HayeSep Q 80/100 mesh column operating in a back-flush mode, in conjunction with a thermal conductivity detector (TCD) at 150°C, using argon as the carrier gas.

5.4.6 Fibre Analysis

This test was performed using the FibreCap technique described in Section 3.3.3, which follows the principle of Van Soest developed procedures. The apparatus used was Foss Analytical FibreCap 2021/2023 system (Kitcherside *et al.*, 2000), which is shown in Figure 5.9. The test includes three separate procedures of ADF, NDF and ADL analysis to partition cellulose, hemicellulose and lignin in the test samples. As showed in Figure 5.10, each analysis includes the procedures of digestion in heated solutions, drying the residue and determining the ash content in the residue.

First, the 105 °C dried and milled sub-sample (0.3-1.0 g) after the preparation procedure described in Section 5.3.2 was weighed into a FibreCap cylindrical capsule, which is 58 mm in length, 23 mm in diameter. NDF analysis was performed using a neutral (pH 7) solution made from sodium lauryl sulphate, EDTA, sodium tetraBorate dicahydrate and disodium hydrogen phosphate that removes starch, protein, organic acids, etc. α-Amylase was also used in the test to improve the hydrolysis of starch. The ADF analysis requires digestion of the sample using 0.5M sulphuric acid containing 2% CTAB, for one hour after the solution reaches boiling point. ADL analysis requires that the sample first subjected to the ADF test. The wet residue from ADF test is further treated with 72% (w./w.) sulphuric acid at room temperature to hydrolyse cellulose, leaving lignin as the residue. After digestion in various solutions, in each analysis the residues remaining in the capsules were dried at 105°C for at least 5 hours in a convection drying oven. Then

these weighed, dried residues and capsules were ashed at 600±10°C for four hours in a muffle furnace. The dry samples and ash after ignition were gravimetrically determined. The processes yield cellulose, hemicellulose and lignin contents calculated using Equation 3.5 to 3.7. For each different sample analysed, each fibre analysis was carried out in triplicate.

5.5 Statistics analysis

Standard statistical techniques were employed to analyse the results of the experiments. All the test results are expressed as the mean±s.d. (standard deviation) of the duplicate or triplicate samples analysed. Regression analyses were performed to assess correlations between tests. The goodness of the correlations was checked through correlation coefficient (R^2). Statistical significance of R was checked by an F-test. Confidence intervals for regression coefficients were determined at the confidence level of 95% (Kreyszig, 1998).

5.6 Material and Operation of Composting Experiments

Four batches of fresh mechanically pretreated MSW were used in the composting experiments. For each batch, there were duplicate composting runs, which are named R1 and R2. The first batch (B1) of fresh MSW sample was obtained in July, 2006. The components of the MSW were then sorted and shredded. The composition of the wastes is presented in Table 5.1. Sewage sludge (from Millbrook Wastewater Treatment and Recycling Centre in Southampton, UK) was added as inoculum at the start of composting to initiate the composting process as inoculum. The BMW samples used for filling each reactor were well mixed with about 440 ml sewage sludge at a ratio of 155:1 (g. DM : g. DM). Additional demineralised water was added to achieve a moisture content of about 60% at the start. The actual

moisture after analysing the collected samples was 59.71%. The amount and composition of the initial BMW samples is detailed in Appendix C and Chapter 6.

The composting duration of the B1 was 104 days. The actual amounts of the partly composted BMW samples collected from the reactors are given in Appendix C. After each sampling event, water was added to the remaining waste which was then re-mixed in the reactors in order to maintain the moisture content of the waste. The sampling intervals were as following:

At the start of composting: Day 0 (D 0)

After 5 days: Day 5 (D 5)

After 10 days: Day 10 (D 10)

After 20 days: Day 20 (D 20)

After 35 days: Day 35 (D35)

After 68 days: Day 68 (D 68)

After 104 days: Day104 (D 104)

Table 5.1. Composition of original fresh MSW and sorted BSW

	B1		B2		B3		B4	
Components	**MSW Percentage (%)**	**BMW Relative Percentage (%)[a]**	**MSW Percentage (%)**	**BMW Relative Percentage (%)[a]**	**MSW Percentage (%)**	**BMW Relative Percentage (%)[a]**	**MSW Percentage (%)**	**BMW Relative Percentage (%)[a]**
Paper	21	38	23	33	21	31	21	34
Yard waste	12	22	17	24	22	33	22	37
Kitchen waste	22	40	31	43	24	36	17	29
Other[b]	45	--	29	--	34	--	40	--
Total (BMW)	55	100	71	100	66	100	60	100

a- The relative percentage of the BMW components for composting doesn't include water and compost inoculum added.

b- Metal, glass, plastics, rubber, etc., which are not recognised in the category of paper, yard waste or kitchen waste.

The fresh mechanically pretreated MSW samples for the 2nd, 3rd and 4th batches (B2, B3 and B4) were obtained individually in Nov. 2006, Jan. 2007 and Feb. 2007. The components of MSW and the sorted and shredded BMW samples are given in Table 5.1. About 400 to 500 gram small size (<10mm) of the stabilized compost from the former composting runs were used as inoculum. In B2, B3 and B4, the composting was limited to 28 days. The actual amounts and composition of the initial BMW samples for composting and the samples collected for testing and analyses are given in Appendix C and Chapter 6. The sampling intervals for both the B2 runs (R1 and R2) were as follows:

At the start of composting: Day 0 (D 0)

After 6 days: Day 6 (D 6)

After 12 days: Day 12 (D 12)

After 20 days: Day 20 (D 20)

After 28 days: Day 28 (D28)

The sampling intervals for both the B3 runs (R1 and R2) were as following:

At the start of composting: Day 0 (D 0)

After 4 days: Day 4 (D 4)

After 8 days: Day 8 (D 8)

After 17 days: Day 17 (D 17)

After 28 days: Day 28 (D28)

The sampling intervals for both the B4 runs (R1 and R2) were as following:

At the start of composting: Day 0 (D 0)

After 4 days: Day 4 (D 4)

After 8 days: Day 8 (D 8)

After 16 days: Day 16 (D 16)

After 28 days: Day 28 (D28)

The labelling of samples indicated which batch, which run and when the sample was collected, following the sequence of batch (B1, B2, B3 or B4), runs (R1 or R2) and then the days of composting before samples collected, which was counted since the start of each batch of composting runs. For example, the samples in B1 were labelled as in Figure 5.11, and B1R1D5 meant 5 days composted sample collected on the start of day 6 in R1 of B1. In the following Chapters 6-9, the term of 'waste' all means BMW with the exception of additional explanation; the 'untreated waste' means fresh mechanically pretreated BMW and the 'treated waste' means mechanically and biologically treated BMW.

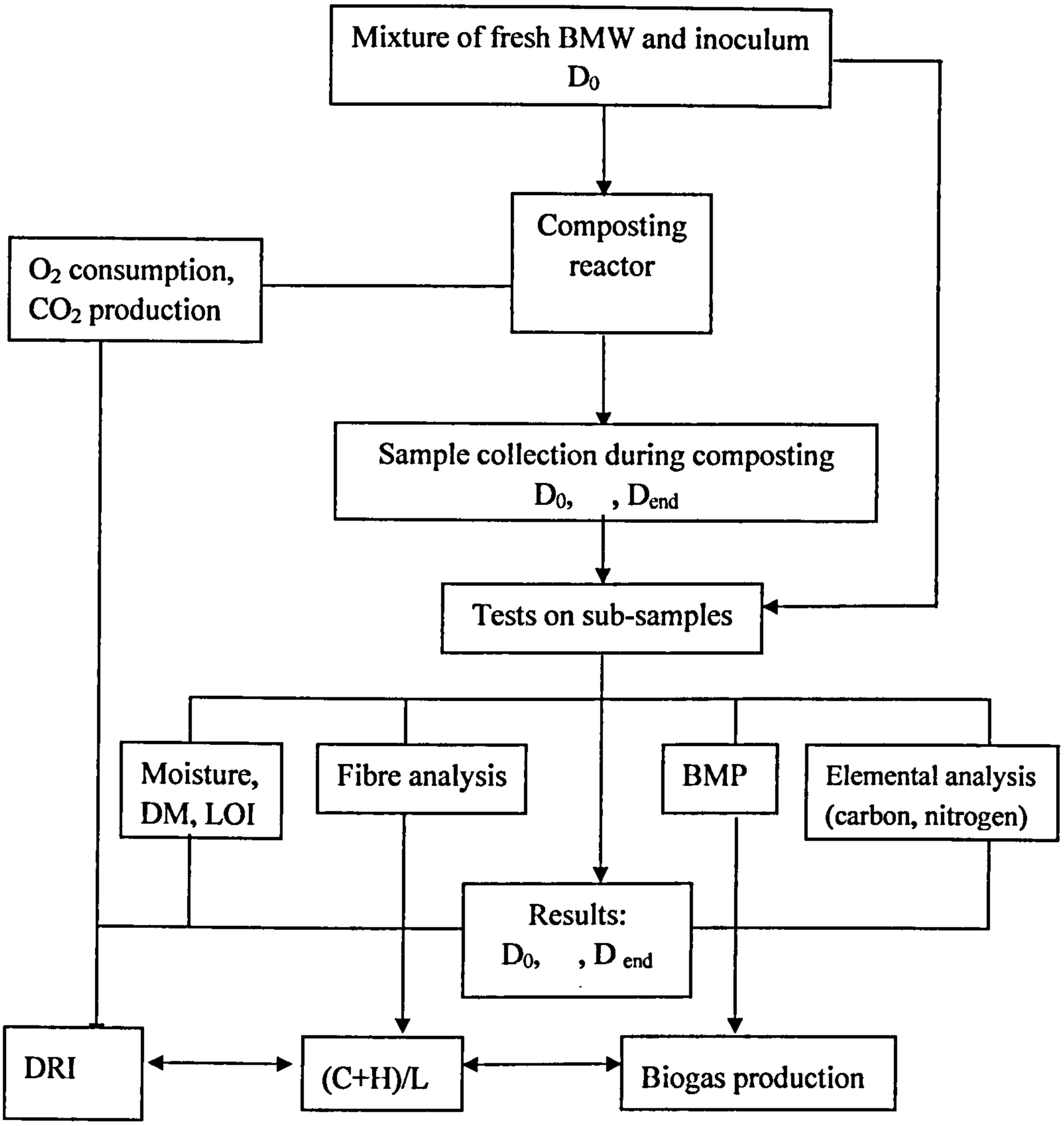


Figure 5.1. Sequential sampling and tests procedures for samples collected at the start of and during the composting process

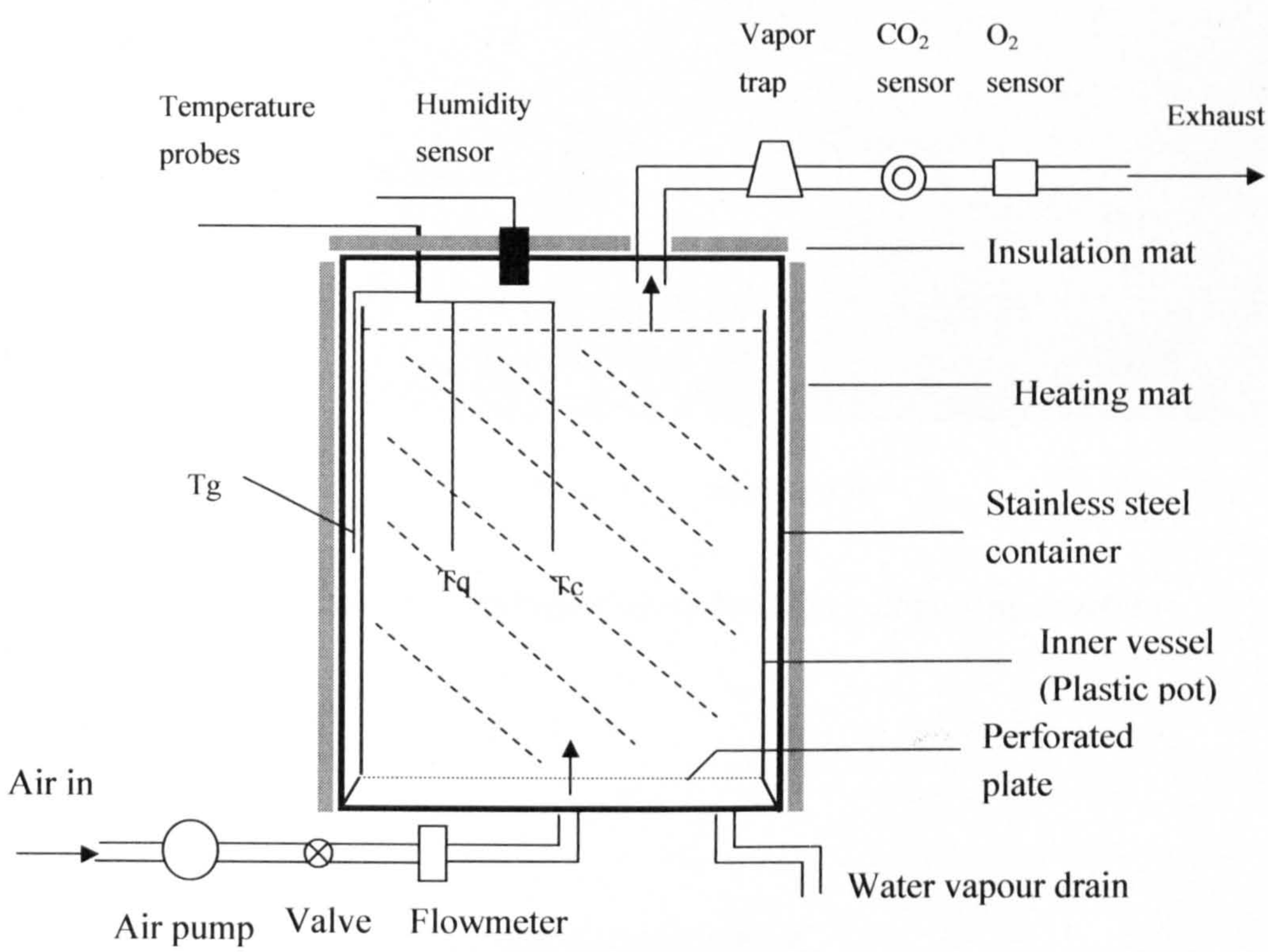


Figure 5.2. Schematic diagram of the composting reactor

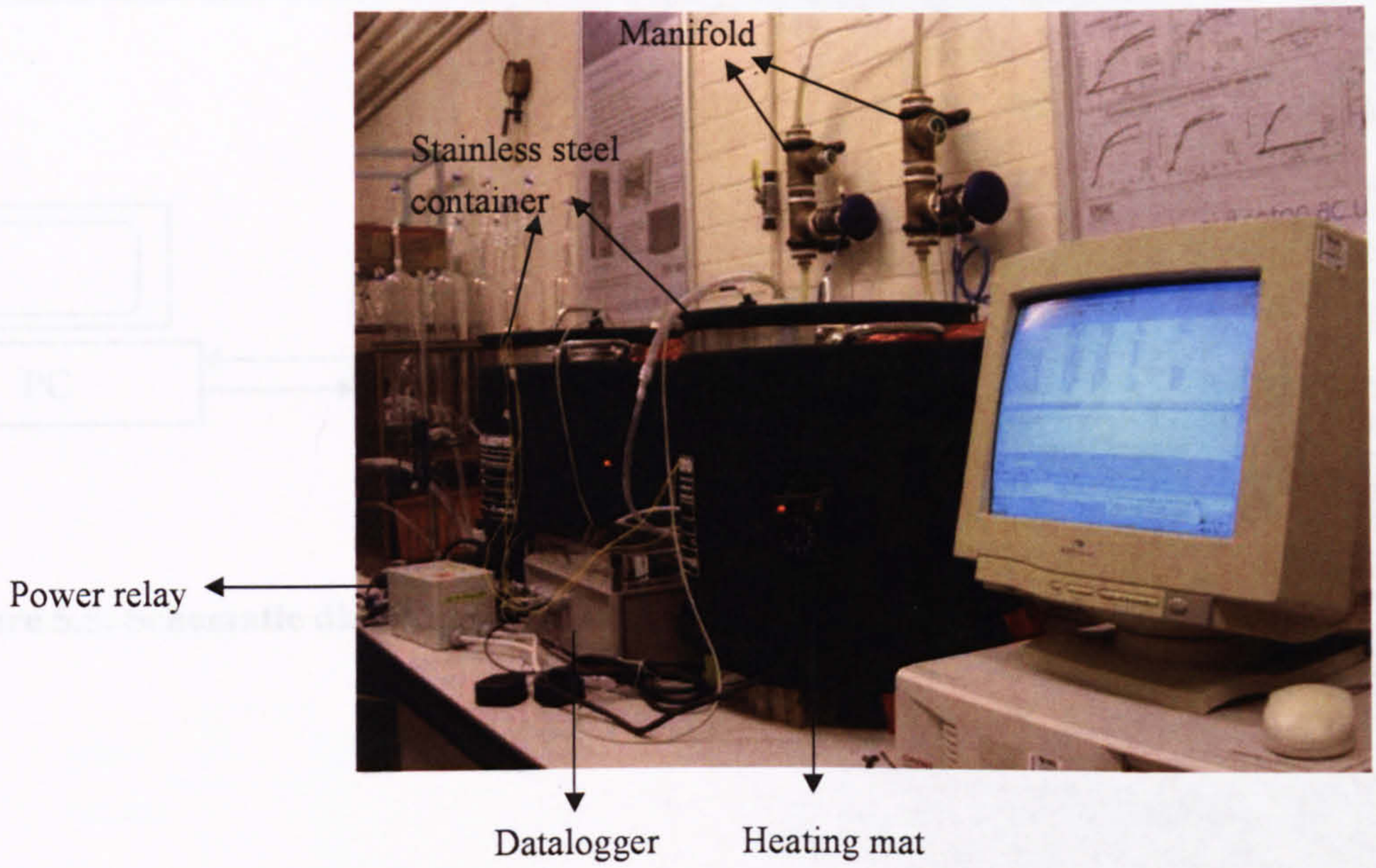


Figure 5.3. Computer, datalogger, relay box and composting reactors

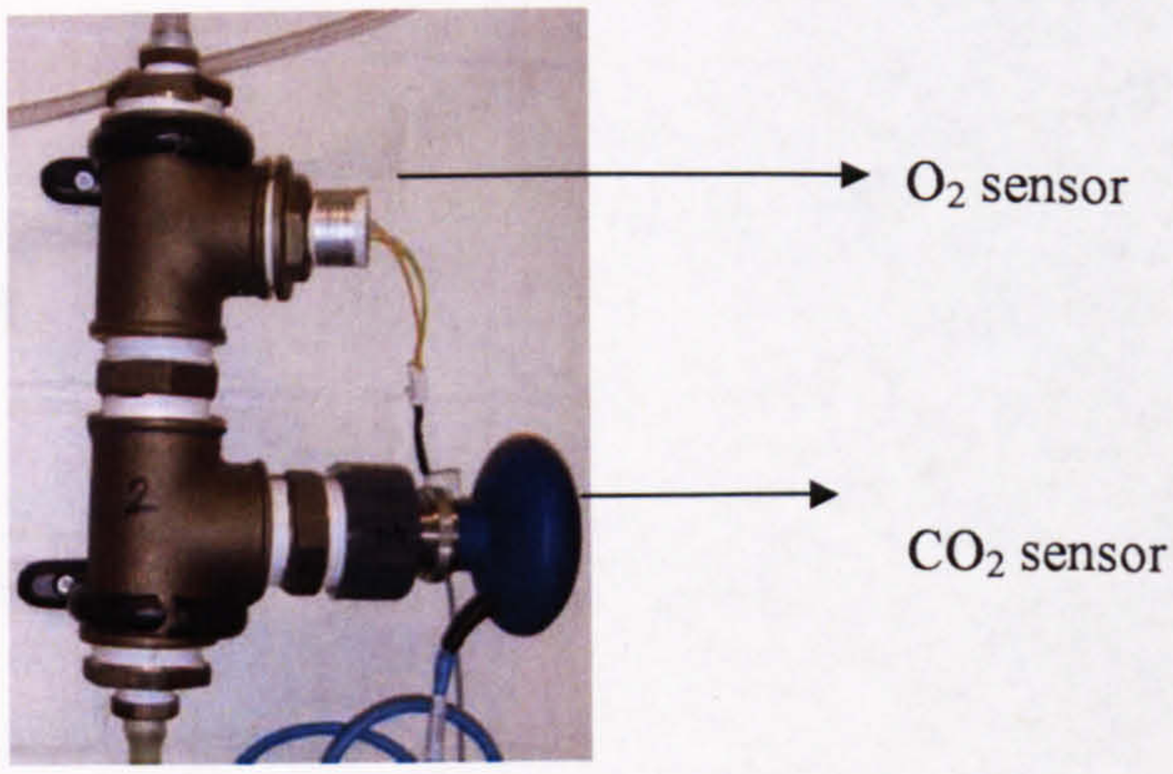


Figure 5.4. Manifold connecting O_2 and CO_2 sensors

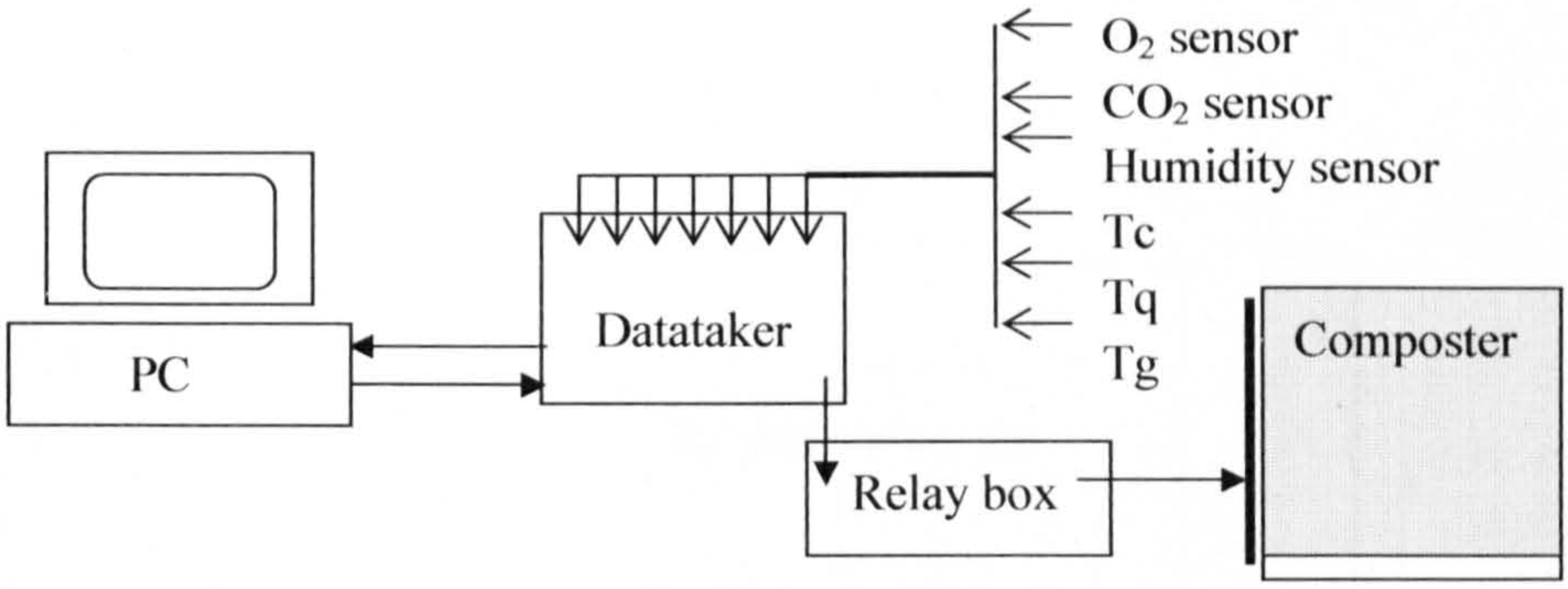


Figure 5.5. Schematic diagram of the computer controlled composting system

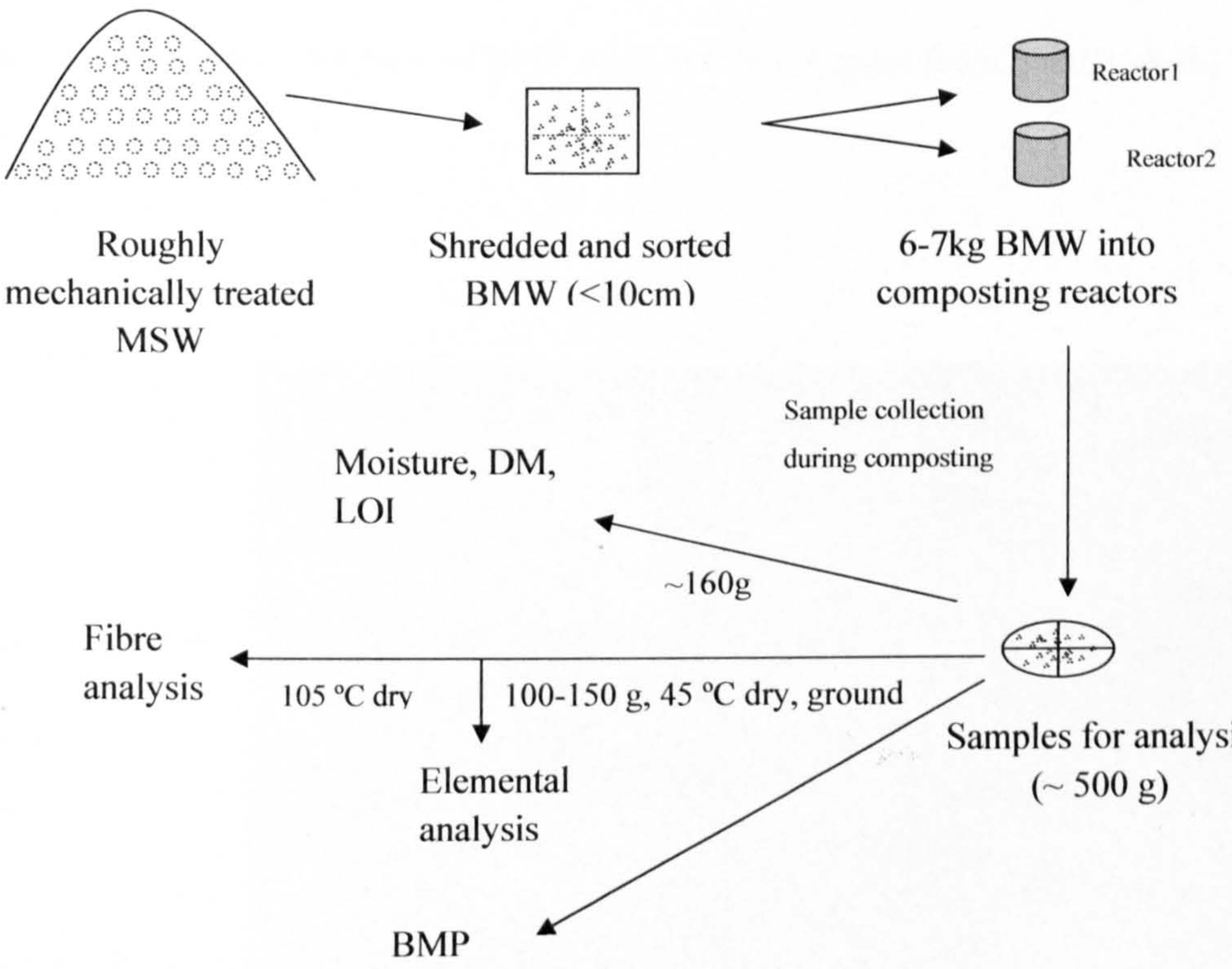


Figure 5.6. Sampling procedures and sub-sample preparation for analysis

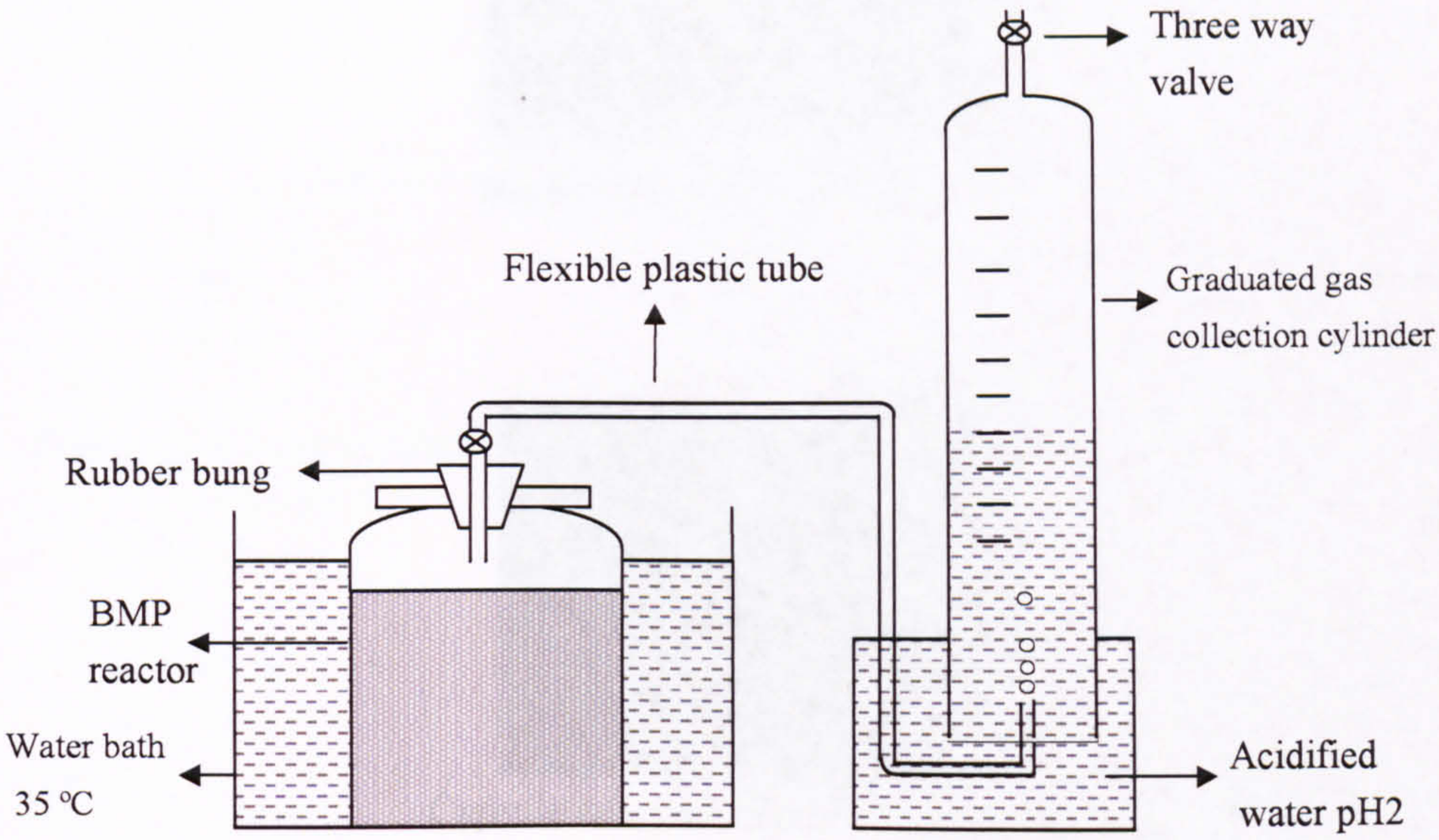


Figure 5.7. Schematic diagram of BMP assay set up (adapted from Godely et al., 2005)

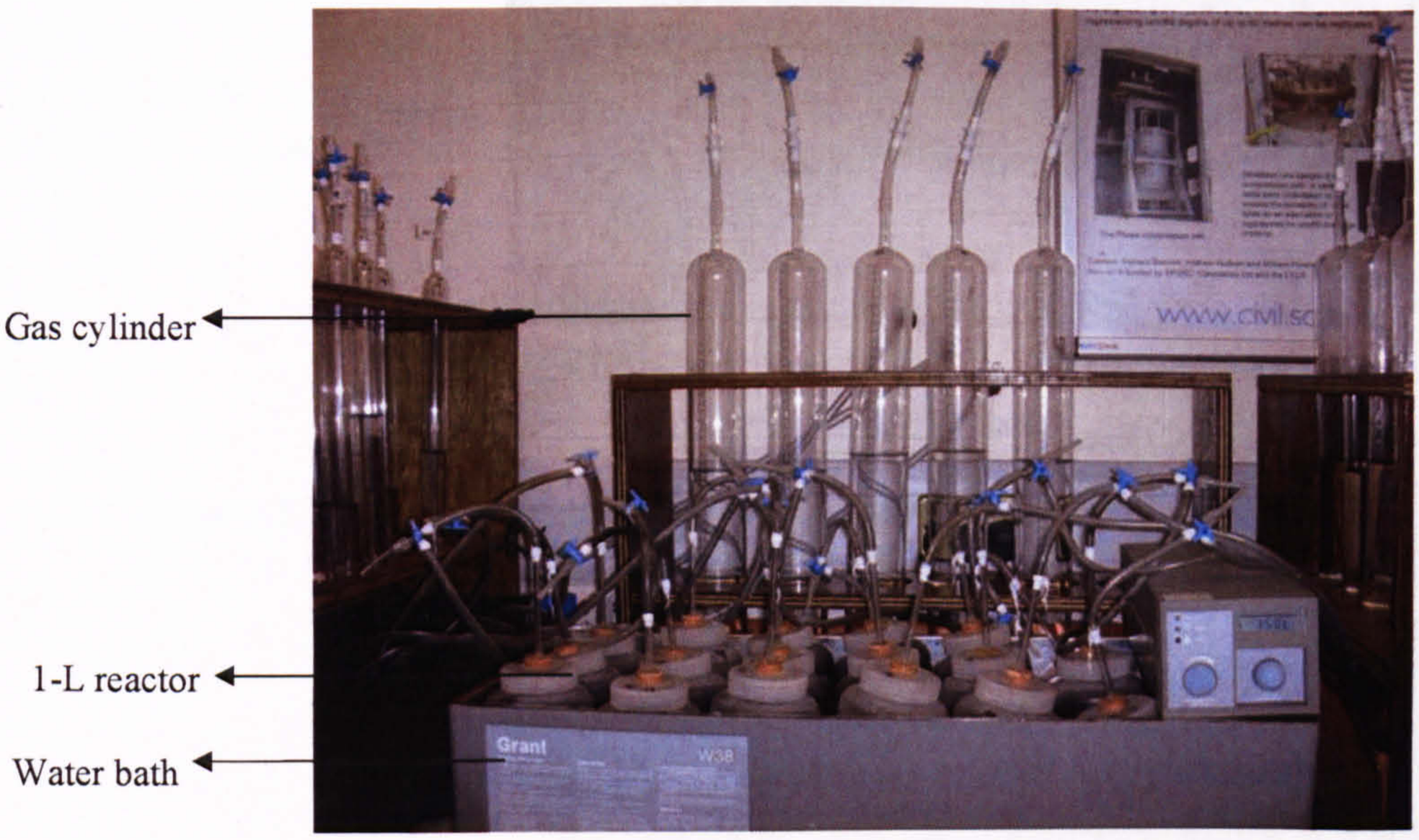
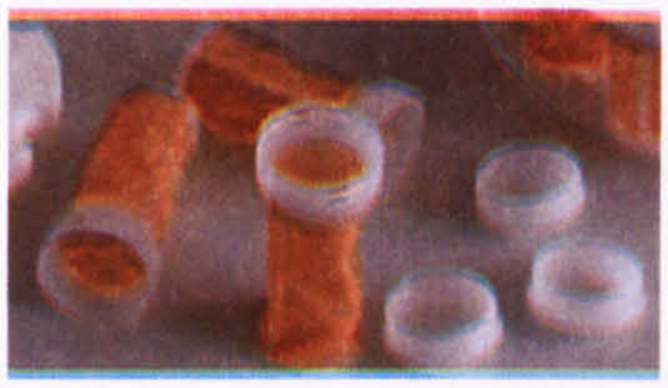
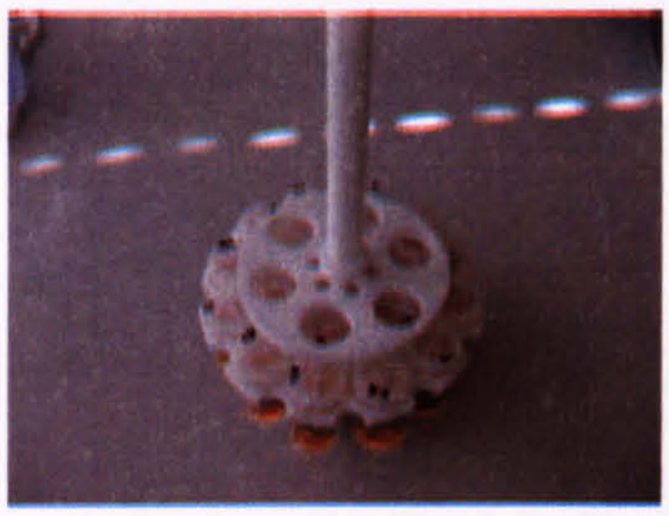


Figure 5.8. Apparatus for BMP assay



FibreCap cylindrical capsule

↓

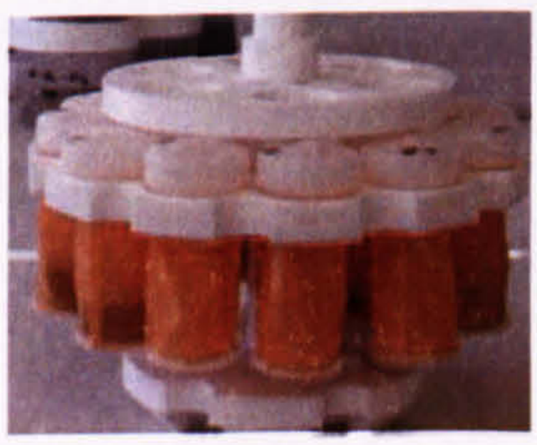


Capsule tray and stand with test samples

↓

Digest in heated solutions

↓



Dried residue

Figure 5.9. Apparatus for fibre analysis by Fibre-Cap system

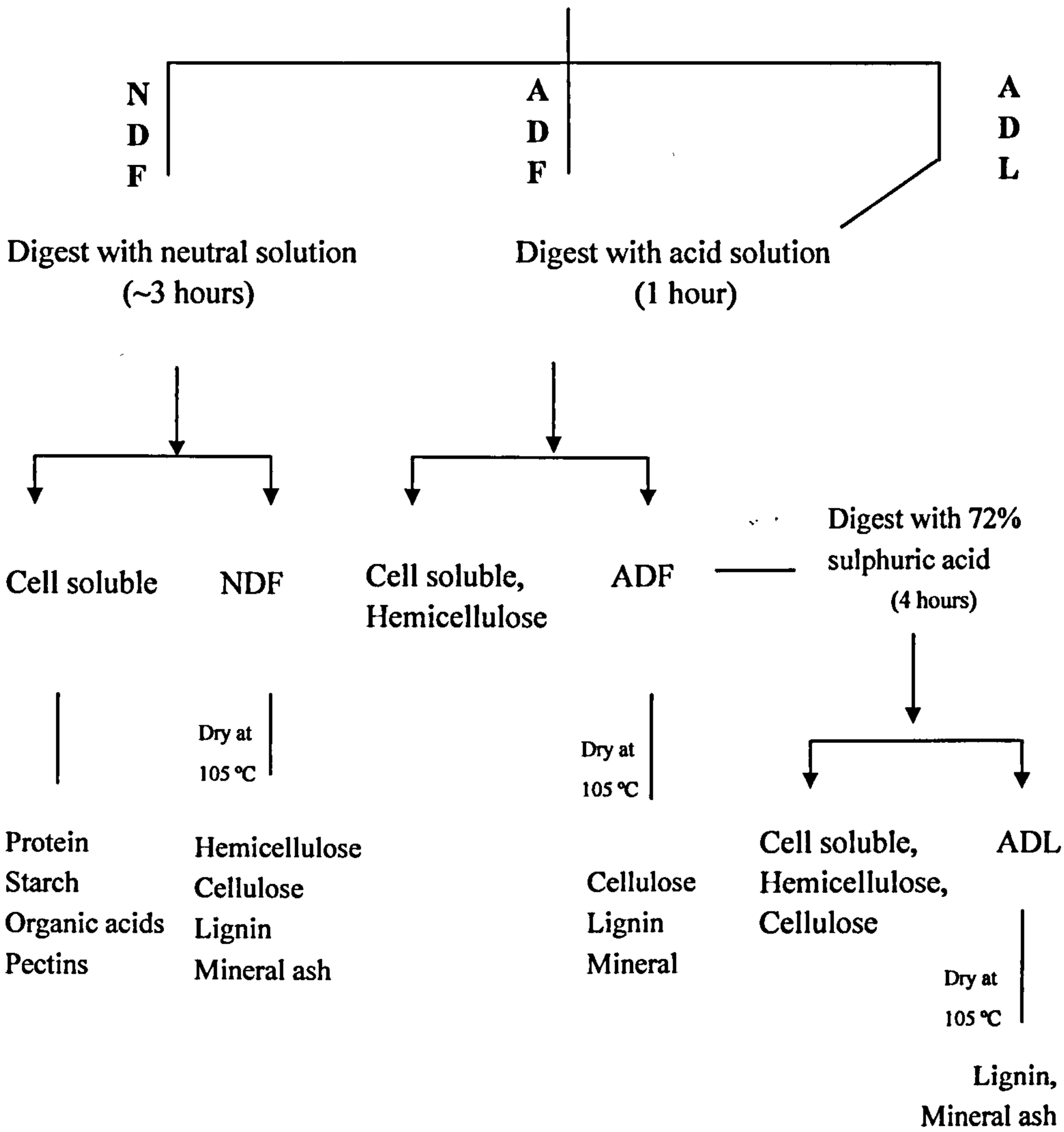


Figure 5.10. Procedures of fibre analysis by FibreCap system

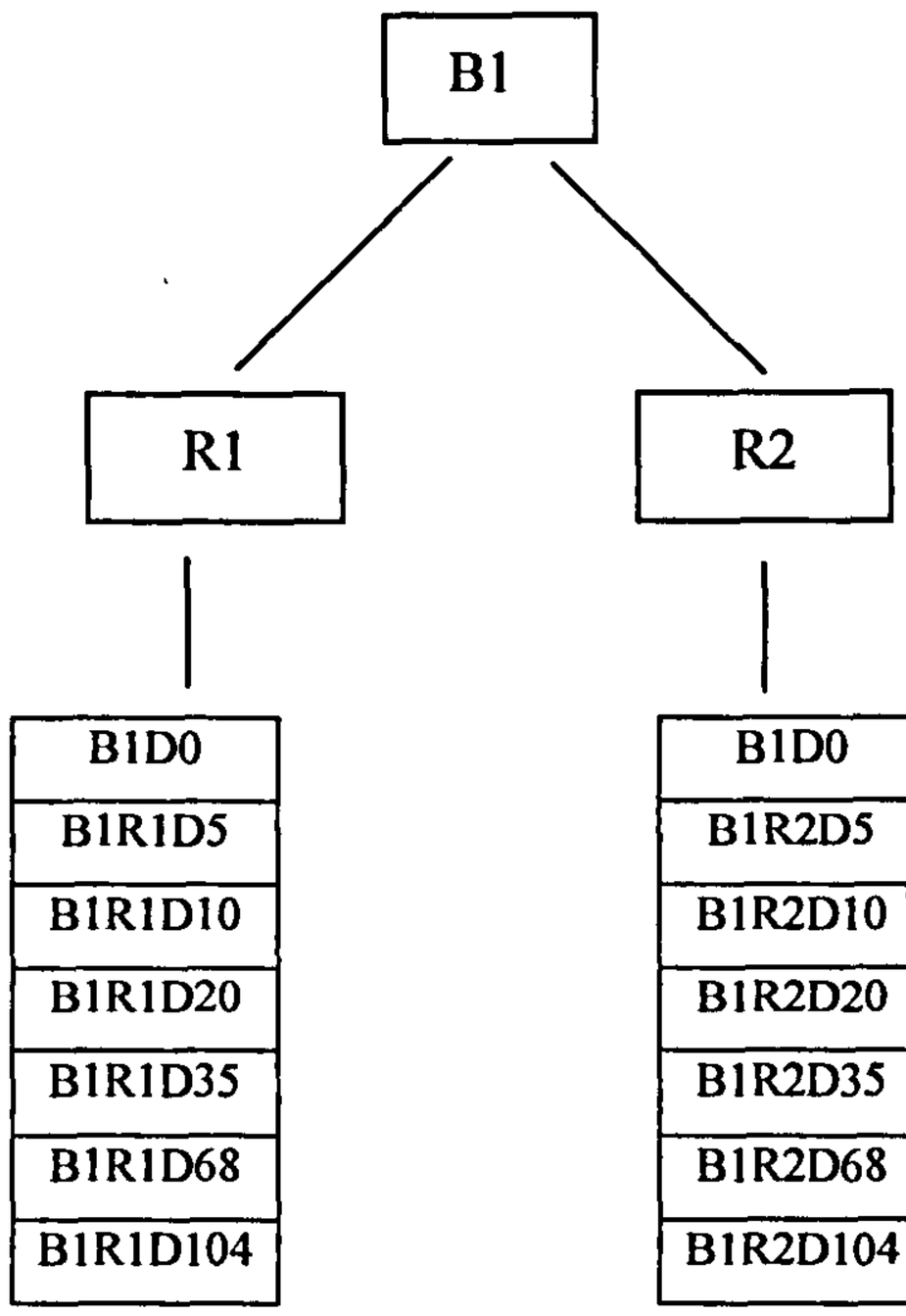


Figure 5.11. Example of sample labelling in the Batch 1

CHAPTER 6 OPERATION AND RESULTS OF COMPOSTING RUNS

Four batches of composting experiments were conducted. One of the aims of B1 was to allow the determination of the optimal composting duration for the subsequent experiments (Batch 2, 3 and 4). The composting duration of B1 (104 days) was long enough to observe all the virtually biological activity. As all the measured composting parameters in B1 indicated that most of the degradation took place in the first 20 days of composting as far as the practice of diverting biodegradable waste quickly from landfill in MBT is concerned, in the following repeated composting experiments (B2, B3 and B4), the run time was shortened and limited to 28 days.

In this Chapter, the operation (such as sampling, moistening) and results of these four batches composting experiment are described and discussed. The composting process is described in terms of the temperature and respiratory activity changes (Section 6.1 and 6.2). The degradation of the BMW samples is then characterized by examining the changes in solid composition of the waste (Section 6.3 and 6.5). The reduction in biogas production potential after aerobic treatment is presented in Section 6.4. The composting processes and waste degradation in all the four batches (B1-B4) are also compared and summarized in the corresponding sections.

6.1 Temperature

The composting reactors (R1 and R2) were designed to let the compost inside the reactors self-heat under their own metabolic activity. Whenever heat loss occurred, the heating mats were automatically switched on to a temperature of 55 ℃, which was controlled by the relay box (Section 5.2.2). In B1, the temperature of the

compost, the time of samplings (S), moistening (W) are illustrated in Figure 6.1a. As no significant activity took place in the period after day 35 (the total composting duration was 104 days), only data from day 0 to day 35 are shown. The temperatures of both composts (R1 and R2) rose rapidly over the first eight hours and reached > 50 °C by the second day. Both R1 and R2 reached peak temperatures of 64 °C and 63 °C on the sixth day of composting, immediately after a moistening and mixing operation. Temperatures then stabilized at approximately 55 °C. In the first 29 days, the temperatures of the two composts reached peak temperatures of above 50 °C and then declined slowly, showing similar trends after each moistening and mixing. The compost in R1 was still able to reheat to the peak temperature of 48°C, whilst that in R2 continued to cool and did not reheat after moistening and mixing on the twenty-ninth day, indicating the end of the active phase for R2. After 35 days of composting, the temperature at the centre of the compost (Tc) was less than 30 °C and lower than the temperature in the gap (Tg). Therefore the power relay of the heating mat was removed and the heating mats were kept at 30 °C to allow the continuous maturation of the compost.

The temperature changes of both composts (R1 and R2) in B2, B3 and B4, and the corresponding times of mixing, moistening and/or sampling during composting are illustrated respectively in Figures 6.1b to 6.1d. In B2, the temperatures of both runs (R1 and R2) rose rapidly after 18 hours and reached 55 °C after 28 hours. Temperatures in R1 stayed above 50 °C over the following 11 days except during mixing, moistening and sampling events. In both runs after about 13 days of composting, temperatures started to decline steadily to around 30 °C.

In B3, the temperatures of both runs (R1 and R2) rose rapidly to above 55 °C over the first 28 hours. The compost in R1 reached a peak temperature of 58 °C and the temperature fluctuated between 55 °C and 58 °C over the following 5 days except during the mixing or sampling events. The temperature in R2 was generally constant around 55°C, but then declined rapidly to less than 40 °C after 3 days of composting. It then rose rapidly back to 55°C after moistening and mixing on day 5.

Temperatures in both runs declined slowly after 6 days of composting. This cooling process continued for about 3 days until the temperature in both runs dropped to below 30 °C after which it rose again. This reheating process took about 3 days before a peak temperature of about 43 °C were reached, which was then maintained for one day in both reactors. Two days later, temperatures were observed to increase once again to around 42 °C. After 20 days of composting, the composts cooled again. It seems that after six days of composting, the biological process was limited more by the biodegradable material available, instead of by the moisture of the compost, as re-moistening and mixing did not accelerate the biological activity in this case.

In B4, temperatures in both runs rose rapidly at the beginning and reached a maximum of 55 °C in twenty-seven hours. The temperatures in R1 and R2 were then stable at around 55 °C for the following 2 days (R1) and 3 days (R2). After the first sampling and mixing events on day 5, temperatures in both runs climbed back to around 55 °C rapidly from the heat loss. From day 9 (the time of the second sampling event) to day 17 (the time of the third sampling event), there was a lag phase (cooling phase) before the temperature in R1 was observed to increase again, while the compost in R2 reheated rapidly and the temperature was relatively constant at 47 °C to 50 °C. The third sampling and moistening event on day 17 attributed to the temperatures drop in both composts by more than 10 °C. Temperatures then only climbed slightly (3 °C in R1 and 5 °C for R2) before declining steadily, indicating the end of the active phase of composting. After the fourth moistening procedure on day 24, temperatures remained stable (<30 °C) until the termination of the experiments.

In summary, the waste material generated heat rapidly and reached a temperature (> 50 °C) in less than one day. The operation of sampling, moistening and mixing disturbed the composting process, but in cases when moisture content was not optimal or air was not evenly distributed, it accelerated the biological activity. Additionally, before the end of the active phase of composting the waste material continued to generate heat and the temperature climbed back to > 50 °C after every

time of sampling, moistening and/or mixing. After about 20 days of composting, temperatures in the three batches of composts were below 30°C, indicating the end of the active phase.

6.2 Respiration Activity and Index

6.2.1 O_2 Consumption Rate and CO_2 Production Rate

The instantaneous dynamic respiration index (DRI_i) was calculated using Equation (3.8) (Adani *et al.*, 2004). Because the actual amount of LOI (kg) remaining in both reactors decreased due to degradation and periodical sampling, the LOI value used for the DRI_i calculation was taken as the LOI value after sampling instead of the total LOI at the start of the composting， except for the period from the start to the first sampling.

The instantaneous DRI obtained in the first 35 days of composting in B1 is illustrated in Figure 6.2a. The corresponding profile of CO_2 production rate is presented in Figure 6.3a. Changes in O_2 consumption rate and CO_2 production rate in the two reactors followed a similar trend to that observed for temperature. The oxygen consumption rate rose rapidly in the first 8 hours to a primary maximum biological activity (about 3700 mg O_2 kg^{-1} LOI h^{-1}), which lasted for about 2 hours after which it was observed to decrease rapidly. After 23 hours the oxygen consumption rate started rising slowly again. On the sixth day, after the first moistening and mixing operation, the respirometric activity accelerated with a rapid rise in the DRI_i from 2628±32 mg O_2 kg^{-1} LOI h^{-1} to a peak of 4877±194 mg O_2 kg^{-1} LOI h^{-1} (mean values of the two runs) in four hours being observed. The maximum DRI_i during the whole composting (the third peak) occurred on the eleventh day. In the first 20 days, after every moistening and mixing event O_2 consumption rate and CO_2 production rate rose rapidly to a peak and then decreased gradually until the next moistening and mixing. After 20 days of composting, the respirometric activity

slowed as the DRI_i decreased to a value of below 1000 mg O_2 kg^{-1} LOI h^{-1}. Furthermore, moistening and mixing affected the respirometric activity to a much less degree compared to that observed in the first 20 days, indicating that the easily degradable material was depleted and that the composts at this stage were of a higher stability.

The changes of DRI_i and the profiles of CO_2 production rate in the B2, B3 and B4 composting experiments are illustrated in Figures 6.2b to 6.2d and Figures 6.3b to 6.3d, respectively. In B2, the oxygen consumption rate rose rapidly at the start, and after 17 hours the first maximum biological activity was reached, with the average DRI_i of the two composts being 3076 mg O_2 kg^{-1} LOI h^{-1} (s.d.=100 mg O_2 kg^{-1} LOI h^{-1}). This was maintained for one hour after which the oxygen consumption decreased rapidly. On day 7, after the first sampling and re-moistening event, the respirometric activity accelerated with a rapid rise of the DRI_i to a peak of 5528 mg O_2 kg^{-1} LOI h^{-1} in R1 and 4007 mg O_2 kg^{-1} LOI h^{-1} in R2. These were the maximum DRI_i observed during the whole composting processes in both R1 and R2. The oxygen consumption rate then decreased. After 15 days of composting, the DRI_i decreased to around 1000 mg O_2 kg^{-1} LOI h^{-1}. It continued to decline even after the sampling and mixing events (on day 21), and stayed below 300 mg O_2 kg^{-1} LOI h^{-1} from day 20 until the termination of the runs. Thus, respirometric activity was virtually complete after 20 days of composting, when the easily degradable material was depleted and the composts attained a higher stability (European Commission 2001).

In B3, the oxygen consumption rate rose rapidly to more than 2000 mg O_2 kg^{-1} LOI h^{-1} after 12 hours. The maximum DRI_i over the first 4 days was 3788 mg O_2 kg^{-1} LOI h^{-1} in R1 and 4634 mg O_2 kg^{-1} LOI h^{-1} in R2. This intense respirometric activity was maintained for about 5 hours before it slowed. After four days of composting, when the reactors were opened for the first sampling procedure, the oxygen consumption rate again rose rapidly to a peak value of 3253 O_2 kg^{-1} LOI h^{-1} in R1and 4668 O_2 kg^{-1} LOI h^{-1} in R2 after about 14 hours. Again, the oxygen

consumption rates subsequently declined slowly. After about 10 days of composting, the oxygen consumption rate rose and then again declined with the DRI_i peaks around 1500 mg O_2 kg^{-1} LOI h^{-1} and 2000 mg O_2 kg^{-1} LOI h^{-1} respectively, which were lower than the peaks in the first 6 days of composting. Upon the termination of the composting runs, the DRI_i had dropped to less than 500 mg O_2 kg^{-1} LOI h^{-1}.

In B4, the oxygen consumption rate rose rapidly to more than 2000 mg O_2 kg^{-1} LOI h^{-1} in 12 hours. The DRI_i then fluctuated at values around 3000 mg O_2 kg^{-1} LOI h^{-1} before gradually declining after 3 days of composting. After 4 days when the reactors were opened for the first sampling event, the oxygen consumption rate rose rapidly to peak values of 4262 O_2 kg^{-1} LOI h^{-1} in R1and 4364 O_2 kg^{-1} LOI h^{-1} in R2 after 10 hours. These were the maximum DRI_i values attained for each run (R1 and R2) in the whole composting process. The oxygen consumption rates then declined slowly. After 11 days of composting, when the DRI_i in R1 had declined to around 500 O_2 kg^{-1} LOI h^{-1}, it then increased slowly and remained at about 1000 O_2 kg^{-1} LOI h^{-1} for a period of 4 days. After that, it showed a declining trend. The DRI_i in R2 declined steadily after 11 days of composting. Re-moistening after 16 days and 23 days did not lead to a resumption of the respirometric activity, indicating the depletion of readily degradable material and the end of active phase. Before the termination of the runs, the DRI_i values of both composts were less than 400 O_2 kg^{-1} LOI h^{-1}. According to Figure 6.3d, the degradation of the waste material in the first 11 days was much faster than that after 11 days, when the easily degradable material was nearly depleted. In these two runs, re-moistening and mixing at any time after 8 days of composting did not accelerate the oxygen consumption rates and CO_2 production rates. The reason may be that the easily degradable material was nearly depleted.

Figure 6.4 shows the cumulative CO_2 production curves of the four batches composting experiments over 28 days. The marked points are the mean values of the two runs in each batch. According to this figure, CO_2 production rates in B2, B3 and B4 were similar until day 7. Generally, after about 16 days of composting, the CO_2

production became steadily much slower compared with that in the first 16 days. During 28 days of composting the CO_2 produced was 1120 g/kg LOI (B2 and B3), 1014 g/kg LOI (B1) and 886 g / kg LOI (B4). In B1, the CO_2 produced in the last 76 days of maturation phase was 34 g/kg LOI.

In summary, the profiles of DRI_i and the CO_2 production rate followed a similar trend to the changes of temperature in each batch, showing the consistency of the waste degradation phase. At the start of composting, the respiration rate rose rapidly to a peak value, but then declined rapidly before reaching another peak value. This indicated that the biological activity was limited by some factors. Although there were small peaks after the highest peak, generally the whole profiles of DRI_i and the CO_2 production rate consist of a lag phase at the start, an increasing phase and associated peak, followed by a decreasing phase, showing a similar shape as the curve of temperature variation with time in Figure 2.2.

In general, after each moistening and mixing event, oxygen consumption rates and CO_2 production rates rose rapidly to a peak and then dropped until the next moistening and mixing event where easily biodegradable material was available. In some of these cases, the moisture contents were in the optimal range (such as sampling and moistening on day 7 in B2). Therefore, the operation of mixing enhanced the respirometric activity, although the forced aeration system used during the whole composting process and oxygen content in the reactors was controlled to an optimal level. Additionally, when moistening and mixing did not improve the respirometric activity of the composting, the compost was deemed to have already attained a state of high stability.

As described in section 5.2.2, in this study the incubation temperature for the DRI test or composting was controlled by the switch of the heating mats which was used to reduce heat loss. When the heating mat was switched on, its temperature was set at 55°C. The fluctuation of temperature over a small range in the temperature changes of the two runs in each batch was found consistent with the on or off of the

heating mats. Therefore, this method of preventing heat loss may unavoidably influence the biological activity to some extent.

6.2.2 Expressions of Biological Stability by DRI

In this study, the biologically treated waste samples were obtained through periodically sampling from the composting runs in each batch, as detailed in Section 5.5. The duration of the DRI test of the untreated waste (fresh mechanically pretreated BMW, day 0) and the corresponding treated waste (partly composted BMW) waste samples in each batch was interpreted in two ways (A and B), which is presented in Table 6.1. On the one hand (A), the whole composting run in a batch was regarded as several concurrent DRI tests which lasted from start (day 1, for the untreated waste) or from the previous sampling event (for the treated waste) to the very end of the composting run. On the other hand (B), the test duration was 4 days, which was the test length for determining DRI4 (Section 3.4.2) and counted from start (day 1, for the untreated waste) or directly after each sampling event (for the treated waste). For example (in the way B), in B1 the DRI test duration for the untreated waste sample was from day 1 to day 4; as the first sampling took place after 5 days of composting, after sampling, water addition and mixing, the 5 day composted waste remaining (D5) in the reactors for further composting can be regarded as a different test sample (biological treated BMW sample) for another DRI test, the duration of which was counted from day 6 to day 9; similarly, the DRI test for the D10 sample spanned day 11 to day 14, etc..

Table 6.1. Description of the tested samples and test duration

Batch	Sample	Description	Test duration (day)	
			A	B
B1	D0	Untreated BMW at D0	1-104	1-4
	D5	5 days composted BMW	6-104	6-19
	D10	10 days composted BMW	11-104	11-14
	D20	20 days composted BMW	21-104	21-24
	D35	35 days composted BMW	36-104	36-39
	D68	68 days composted BMW	69-104	69-72
	D104	104 days composted BMW		
B2	D0	Untreated BMW	1-28	1-4
	D6	6 days composted BMW	7-28	7-10
	D12	12 days composted BMW	13-28	13-16
	D20	20 days composted BMW	21-28	21-24
	D28	28 days composted BMW	-	-
B3	D0	Untreated BMW	1-28	1-4
	D4	4 days composted BMW	5-28	5-8
	D8	8 days composted BMW	9-28	9-12
	D17	17 days composted BMW	18-28	18-21
	D28	28 days composted BMW	-	-
B4	D0	Untreated BMW	1-28	1-4
	D4	4 days composted BMW	5-28	5-8
	D8	8 days composted BMW	9-28	9-12
	D16	16 days composted BMW	17-28	17-20
	D28	28 days composted BMW	-	-

The results of the respiration index tests used to express the degree of biological stability are calculated using the first three methods in Table 6.2 recommended by Adani *et al.* (2004) including DRI_{ave}, DRI_{max}, and DRI4. The test durations used for the calculation in these three methods is described as B in Table 6.1. For example, the DRI4 was calculated as the cumulative DRI_i in 4 days from day 1 for the untreated waste samples or from the previous sampling event for the treated BMW. Another index DRI_{tot} used in this study is calculated as the cumulative DRI of the sample in the whole experiment in order to evaluate the overall biodegradability in

terms of oxygen consumption. The duration of $DRI_{tot.}$ is presented in Table 6.1 as A. The results of different DRI expressions are given in Appendix B. The corresponding changes of stability expressed by these four different DRI methods during the aerobic treatment process for the 4 batches composting runs are shown in Figures 6.5 to 6.8, respectively, where the points are the average values of the two runs in each batch.

Table 6.2. Different DRI expressions for evaluating stability used in this study

Name	Calculation Mode	Unit	Reference
$DRI_{ave.}$	$\sum_{t=0}^{24}(DRI_t)/24$	mg O_2 kg^{-1} LOI h^{-1}	Scaglia *et al.* (2000)
$DRI_{max.}$	Maximum value	mg O_2 kg^{-1} LOI h^{-1}	Iannotti *et al.* (1993)
DRI4	$\sum_{t=0}^{96} DRI_t$	mg O_2 kg^{-1} LOI 96 h^{-1}	Müller *et al.* (1998)
$DRI_{tot.}$	$\sum_{n}^{end} DRI_t$	mg O_2 kg^{-1} LOI	This work

$DRI_{ave.}$- Average value of DRI_t taken during the 24 hours of the most intense biological activity in 4 days from the start (for D0 sample) or from the previous sampling event;

$DRI_{max.}$-Maximum DRI_t value in 4 days from the start (for D0 sample) or from the previous sampling event;

DRI4- Cumulative DRI_t value in 4 days from the start (for D0 sample) or from the previous sampling event;

$DRI_{tot.}$-Total respiration index, calculated as the sum of DRI_t from the start (for D0 samples) or from the previous sampling event to the very end of the composting experiment (e.g., in B1, n: day 1, 6, 11, 21, 36, 69).

The three methods (DRI_{ave}, DRI_{max} and DRI4) of calculating DRI yielded similar trends in the change of biological stability in each batch of the composting runs, which is consistent with the results of Adani *et al.* (2004). According to the

degradation status of the samples tested, in each batch the untreated waste sample (D0) should yield the highest respiration index and be less stable than any other composted sample tested in this batch because some amounts of biodegradable material were degraded in the subsequent composting processes for the composted waste. However, according to the change of $DRI_{ave.}$ (Figure 6.5), $DRI_{max.}$ (Figure 6.6) and DRI4 (Figure 6.7) in B1, B2 and B4, these values did not give the indication of the stability in the sequence of composting time. For example, in B1 the $DRI_{ave.}$, $DRI_{max.}$ and DRI4 of the 5 days composted waste sample (4102 mg O_2 kg^{-1} LOI h^{-1}, 4877 mg O_2 kg^{-1} LOI h^{-1} and 303 g O_2 kg^{-1} LOI 96 h^{-1}) and 10 days composted waste sample (3758 mg O_2 kg^{-1} LOI h^{-1}, 5838 mg O_2 kg^{-1} LOI h^{-1} and 251 g O_2 kg^{-1} LOI 96 h^{-1}) were higher than those of the untreated waste sample (B1D0) (2488 mg O_2 kg^{-1} LOI h^{-1}, 3742 mg O_2 kg^{-1} LOI h^{-1} and 172 g O_2 kg^{-1} LOI 96 h^{-1}). Similarly, in B2 the $DRI_{ave.}$, $DRI_{max.}$ and DRI4 of the 6 days composted waste sample (4027 mg O_2 kg^{-1} LOI h^{-1}, 4767 mg O_2 kg^{-1} LOI h^{-1} and 265 g O_2 kg^{-1} LOI 96 h^{-1}) was the highest, then the values of untreated waste (B2D0) (3077 mg O_2 kg^{-1} LOI h^{-1}, 3734 mg O_2 kg^{-1} LOI h^{-1} and 225 g O_2 kg^{-1} LOI 96 h^{-1}); The DRI values of the 20 days of composted waste (292 mg O_2 kg^{-1} LOI h^{-1}, 367 mg O_2 kg^{-1} LOI h^{-1} and 18 g O_2 kg^{-1} LOI 96 h^{-1}) were the lowest.

According to the profile of the DRI_i in B1, B2 and B4, the respirometric activity was far from complete in 4 days for B1D0, B1D5, B1D10, B2D0 and B4D0 waste samples. For these waste samples, the degradation of available biodegradable material was still proceeding after 4 days of composting. This suggests that under the incubation condition of these batches of composting runs, the 4 days DRI test duration was not enough for the untreated BMW to be degraded to the maximum degree.

In B3, the $DRI_{ave.}$, $DRI_{max.}$ and DRI4 of the untreated waste (B3D0), 4 days composted waste (B3D4) and 8 days composted waste (B3D8) showed a steady decreasing trend respectively, which was as expected. Therefore, the respirometric activity in the first 4 days was the most intense in the whole process, suggesting that

the untreated waste was degraded to the maximum degree over the test duration of 4 days in this batch. However, according to the profile of the DRI_i, the respirometric activity was far from complete in 4 days for the B3D0 waste. Additionally, the DRI_{ave}, DRI_{max} and DRI4 values of the 17 days composted waste sample (1597 mg O_2 kg^{-1} LOI h^{-1}, 1891 mg O_2 kg^{-1} LOI h^{-1} and 88 g O_2 kg^{-1} LOI 96 h^{-1}) were observed to be higher than those of the 8 days composted waste sample (847 mg O_2 kg^{-1} LOI h^{-1}, 1430 mg O_2 kg^{-1} LOI h^{-1} and 58 g O_2 kg^{-1} LOI 96 h^{-1}), indicating that the respirometric activity from at least day 9 to day 12 (4 days of test duration) was limited by some factor.

The oxygen consumption rate of the tested samples could be affected by several factors influencing the composting process, for example, moisture, microbial population and temperature, as reviewed in Section 2.2.2. If the conditions were not optimum, aerobic degradation would be limited, resulting in a delayed or reduced decomposition rate and consequently a longer treatment time. In such a scenario, the maximum degradation could not be reached in the short time of 4 days. Therefore, for example, when the first three calculation methods (DRI_{ave}, DRI_{max} and DRI4) were used, the 5 days composted waste sample (B1D5) may show a higher DRI than the corresponding untreated waste sample (B1D0). As a consequence, the methods of DRI_{ave}, DRI_{max} and DRI4 could not give a correct sequence of stability for the test materials. Their use depends on the composting conditions.

Nevertheless, in B1 the DRI_{ave}, DRI_{max} and DRI4 of the less than 20 days composted waste were one magnitude larger than those of the 20 days and longer composted waste, indicating a clear difference. According to the biological stability limit value of $DRI_{ave.} \leqslant 500$ mg O_2 kg^{-1} LOI h^{-1} proposed by Adani *et al.* (2004), the 20 days and longer composted waste were of high biological stability as their $DRI_{ave.}$ values were all less than 500 mg O_2 kg^{-1} LOI h^{-1}. Therefore, the waste samples composted for less than 20 days were not considered biologically stable. Similar to the data of B1, an obvious difference existed for the DRI_{ave}, DRI_{max} and DRI4 of B2 between the less than 20 days composted wastes and the 20 days composted waste.

The DRI_{ave} of 20 days composted waste was less than 500 mg O_2 kg^{-1} LOI h^{-1}, indicating it was of high biological stability according to the criteria proposed by Adani *et al.* (2004). In B3, the DRI_{ave} of the 17 days composted waste was not low enough to be regarded as biologically stable. In B4, the DRI_{ave} of the 16 days composted waste was less than 1000 mg O_2 kg^{-1} LOI h^{-1}, indicating it was of medium biological stability according to the biological stability criteria proposed by Adani *et al.* (2004). Therefore, though the three methods ($DRI_{ave.}$, $DRI_{max.}$ and DRI4) may not give the true or exact DRI values of the waste sample tested, they are still suitable for determining the stability of BMW, thereby providing a criterion for the division of the unstable and the stable.

The waste samples in each batch showed a similar trend in the change of stability in terms of $DRI_{tot.}$ (Figure 6.8), which followed the sequence of composting time. The expression of DRI_{tot} was closer to the overall oxygen consumption of the test material compared to the other three expressions of DRI, because it was calculated as the cumulative DRI_i to the end of composting experiments for the untreated and treated waste samples in each batch. Therefore, it can be expected to be the most reliable expression for the evaluation of biological stability among the four DRI expressions. However, low correlation coefficients were found between the $DRI_{tot.}$ and the other three DRI methods. The reason for this may be the time scale of the test selected for the calculation. This further confirmed that the methods of the DRI_{ave}, DRI_{max} and DRI4 may not measure the overall biodegradability.

In summary, the first three methods (DRI_{ave}, DRI_{max} and DRI4) of calculating DRI yielded similar trends for the change of biological stability of the untreated and treated waste, which demonstrated good correlations. But they did not give the sequence of stability in the order of composting time in each batch. In addition, no correlation was found between $DRI_{tot.}$ and the other three methods. A common phenomenon in all the four batches of composting runs was that the biodegradation of BMW could not always reach the maximum degree in a relatively short test time, such as 4 days.

Godley *et al.* (2005) also reported that even under conditions of 35 °C and 1:1 of seed : substrate (dry weight basis), the readily degradable organic carbon was not totally depleted in 4 days or 7 days and that biodegradation still proceeded after 7 days incubation. He concluded that the cumulative DRI_i value in 4 days of test duration (DRI4) represented a biodegradation rate rather than a measure of overall biodegradability of the test material because the availability of readily degradable organic carbon is linked to the production rate of enzymic hydrolysis of polymeric materials in the wastes. In the study of DRI test by Adani *et al.* (2001), they also discussed that the incubation conditions, such as temperature and moisture content, affected aerobic degradation and consequently the oxygen consumption of the test material.

Therefore, on one hand, the first three DRI methods ($DRI_{ave.}$, $DRI_{max.}$ and DRI4) can not always give correct results to demonstrate biodegradability. Their use depends on the prevailing test conditions, which need further investigation. Additionally, the DRI4 could be misleading for indicating the biodegradability because the test duration of 4 days would be less than the time for complete degradation of the degradable material in some cases. On the other hand, some rough evaluation can still be made between materials, for example, raw BMW and compost. Thus, though the first three methods may not give the true or exact DRI values of the sample tested due to the test conditions, they are still suitable for determining the stability of BMW if there is a clear criterion for the division of the unstable and the stable.

6.3 LOI, TC and TN

The LOI, TC and TN contents of DM for the BMW sample at the start, during and at the end of composting are presented in Appendix C. In B1, after 104 days of composting, the LOI content decreased from 85.3% DM to 58.1 ±0.4% DM, with a reduction of 61.5% ±1.1% DM compared with the total LOI present at the start, which was calculated by Equation 6.1. In B2, B3 and B4, the LOI content in terms

of DM was reduced from averages of 83.9 %, 84.2 % and 83.9 % to 62.0 %, 68.3% and 75.9 %, respectively, after 28 days of composting. LOI reduction was 56.9 ± 2.7% of the total LOI at the start in B2, 46.9 ± 2.2% in B3, and 40.7 ± 1.4% in B4.

$$LOI \quad reduction \quad \% = \frac{LOI_0 - LOI_r - \sum LOI_S}{LOI_0} \times 100 \qquad (6.1)$$

In which, LOI_0: the mass of LOI (g) at the start; LOI_s: the mass of LOI (g) removed by sampling; and LOI_r: the mass of LOI (g) remaining at the end of composting.

During composting, the methane content of the exhaust gas from the composting reactors was measured periodically and was found to be close to that concentration found in ambient air, indicating that no carbon was removed by means of methanogenesis or that any methane formed was rapidly oxidized by methane oxidizing bacteria. Therefore, the carbon in the form of CO_2 (Cco_2), which was monitored by the sensors during composting, represents the carbon losses by means of respiration activity. The amounts of carbon losses by respiration activity (Cco_2) between sampling intervals in the two reactors of each batch are shown in Table 6.3a and 6.3b. In B1, the Cco_2 in the first 20 days (500.3 g in R1 and 459.6 g in R2) was much higher than those in the following 84 days (15.3 g in R1 and 8.1 g in R2); in B2, the degradation of carbon in the first 12 days was much higher than in the following 16 days and more than a half of the total carbon removed was degraded in the first 6 days; in B3 and B4, the degradation of carbon in the first 8 days was much higher than the following 20 days. These changes in terms of carbon degradation were all consistent with the profile of DRI_i. (Figures 6.2a to 6.2d).

At the same time, the carbon content of the solid waste samples collected in each batch was tracked by elemental analysis (Appendix C). The changes of total carbon content with time during composting in the four batches (B1-B4) are shown in Figure 6.9. The carbon losses in each run were also be calculated using the

analytical approach (Carbon $losses_{anal.}$) (Equation 6.2) and compared with Cco_2 in terms of Carbon $losses_{anal.}$ / Cco_2 ratio (Table 6.4). Theoretically, the carbon losses calculated by the repirometric data (Cco_2) should be in agreement with Carbon $losses_{anal.}$, which is the case for the two runs in B3 and B4. The possible reasons for ratios higher than 1 in B1 and B2) were that the mixing and moistening procedures could cause some carbon missing when the lids were opened and/or that the general variation in waste composition and the small sample size for elemental analysis might cause some error in the carbon content. The maximal percentage error in the amounts of carbon losses by respiration activity (Cco_2) was estimated to be 33.3% and 20.0% for the composting in B1 and B2, respectively.

$$\text{Carbon losses}_{anal.} = C_0 - C_s - C_r \qquad (6.2)$$

In Equation 6.2, C_0 was the total mass of carbon at the start of composting and was the same for two runs (g):

C_s was the total mass of carbon removed by sampling; and

C_r was the mass of carbon remaining in the final compost at the end of composting (g). And the mass of carbon was calculated as following:

$$\text{Mass of carbon} = C\ (\%) \times \text{Dry wt. (g)}$$

In B1 after 104 days of composting, the total carbon losses by respiration activity (Cco_2) was 41.42% ± 2.87% (the average of the two runs and the standard deviation) of the total carbon at the start (Equation 6.3). After 28 days of composting, the total Cco_2 was 41.7 ± 1.6 % (the average of the two runs) of the total carbon at the start in B2, 42.8% in B3, and 38.1 ± 0.7 % in B4.

$$C_{CO_2}\ \% = \frac{\sum C_{CO_2}}{C_o} \times 100 \qquad (6.3)$$

Table 6.3a. Carbon losses in the form of CO_2 in B1 (Cco_2)

Run	D1-D5	D6-D10	D11-D20	D21-D35	D36-D68	D69-D104	Total
R1 (g)	152.4	187.7	160.2	10.4	2.9	2.0	515.7
R2 (g)	157.4	200.0	102.2	4.1	2.5	1.5	467.7

Table 6.3b.Carbon losses in the form of CO_2 in B2, B3 and B4 (Cco_2)

	Run	D1- D6	D7-D12	D13-D20	D21-D28	Total
B2	R1 (g)	241.9	132.9	42.4	2.8	420.1
	R2 (g)	263.3	90.0	37.9	6.9	398.1
	Time	**D1-D4**	**D5-D8**	**D9-D17**	**D18-D28**	**Total**
B3	R1 (g)	130.6	74.3	45.0	36.9	286.8
	R2(g)	151.9	101.7	46.9	39.3	339.8
	Time	**D1-D4**	**D5-D8**	**D9-D16**	**D17-D28**	**Total**
B4	R1 (g)	113.2	108.4	36.2	26.3	284.1
	R2 (g)	133.3	110.6	30.2	2.7	276.8

Table 6.4. Comparison of Cco_2 with Carbon $losses_{anal}$

Run		C_0 (g)	C_s (g)	C_r (g)	Carbon $losses_{anal.}$ (g)	Carbon $losses_{anal.}$/ Cco_2
B1	R1	1187.0	263.0	173.5	750.5	1.46
	R2	1187.0	248.4	214.8	723.9	1.55
B2	R1	980.3	227.6	227.1	525.6	1.25
	R2	980.3	225.5	258.4	496.4	1.25
B3	R1	750.4	188.5	274.2	287.7	1.00
	R2	750.4	190.4	227.9	332.1	0.98
B4	R1	735.7	202.7	248.1	284.9	1.00
	R2	735.7	212.3	222.9	300.5	1.09

The biodegradable carbon content was expressed as the total carbon loss by composting in the form of CO_2 for each unit of LOI tested as LOI includes both biodegradable carbon and unbiodegradable carbon. It was calculated by the measured amount of CO_2 production from the start of composting to the termination of composting experiments (Equation 6.4). A calculation example is given in Appendix C. The biodegradable carbon contents detected for the untreated waste samples were 288.3 g carbon/kg LOI, 306.0 g carbon/kg LOI, 298.0 g carbon/kg LOI and 241.9 g carbon/kg LOI in B1 to B4, respectively. There would be errors of 33.3% and 20.0% in the biodegradable carbon contents in B1 and B2, respectively, which were introduced by the errors in the amounts of carbon losses by respiration activity (Cco_2).

$$Biodegradable\ \ carbon = \sum_{1}^{end} \frac{C_{CO_2}}{LOI} \qquad (6.4)$$

The changes of total carbon to total nitrogen ratio (TC/TN) during composting in the four batches are presented in Figures 6.10a to 6.10d, respectively. As a whole, the carbon content showed a decreasing trend and the TC/TN ratio decreased during the composting process in each batch. In B1, the TC/TN ratio decreased rapidly during the first 20 days of composting, from 37.4 at the start to 16.6±0.2 (the average of the two runs). Then it decreased relatively slowly to 12.9±0.4 after 84 days more composting. The TC/TN ratio decreased from 24.1±1.9 at the start to 14.3±0.8 at the end of composting in B2, from 23.6±0.4 at the start to 15.6±0.8 in B3, and from 27.7±1.4 to 16.0±0.9 in B4. Generally, the changes of TC/TN ratio in the two runs of each batch were consistent. According to Chayansak and Kubota (1981), a ratio of less than 20 in the solids is indicative of maturity, the composts obtained at the end of each composting experiment could be considered mature.

6.4 Change of BMP after Aerobic Treatment

14 g LOI of the test samples from both runs (R1 and R2) in each batch (B1 to B4) were used for each BMP test. The actual weights of wet samples used are presented in Appendix D. The BMP assays were terminated after 37 days of incubation when the cumulative biogas production curves of the untreated waste and the treated waste samples became relatively flat, as shown in Figures 6.11a to 6.11d, respectively. The results are expressed as cumulative volumes of biogas (CO_2 and CH_4) per unit of LOI. The marked points in the Figures 6.11a to 6.11d are the mean values of duplicate BMP assays for each sample tested. Generally, the longer the BMW was composted, the less the biogas yields.

The final cumulative biogas productions for the tested samples are given in Figure 6.12 to demonstrate the change of biogas potential after aerobic treatment in B1 to B4. The marked points are the mean biogas production of two samples collected on the same day from the two reactors in each batch, except those of the untreated waste (D0) samples. The standard deviations of the mean biogas production are also presented in Figure 6.12, which were low for tested samples in all four batches. In B1, the lowest value was 0.4L/kg LOI for the mean biogas production of 34.1 L/kg LOI of the D104 samples, and the highest value was 37 L/kg LOI for the mean biogas yield of 370.6 L/kg LOI of the D5 samples. In B2, the standard deviations ranged between 4.7 L/kg LOI for the mean biogas production of 147.6 L/kg LOI (D20) and 23.6 L/kg LOI for the mean biogas production of 561.1 L/kg LOI (D0). In B3, the lowest standard deviation was 1.5 L/kg LOI for the mean biogas production (D28), and the highest standard deviation was 37.4 L/kg LOI for the mean biogas production of 183.1 L/kg LOI (D17). In B4, the lowest standard deviation was 2.8 L/kg LOI for the mean biogas production (D28), and the highest value of 30.1 L/kg LOI for the mean biogas production of the untreated waste. This suggests that the samples in the two reactors were closely matched. Furthermore, the

samples collected either at the start or during composting were representative of the waste in the reactors.

Biogas production potential of the untreated waste in B2 was the highest of the untreated wastes used in all the batches. It decreased by almost a half after 6 days of composting. In general, after about 8 days of composting, further changes of the biogas potential caused by composting treatment were reduced. In B1, the reduction of biodegradability in terms of biogas production due to 104 days of aerobic treatment was 347.2 L/kg LOI, which was 91.0% of the anaerobic biogas potential of the untreated waste (D0, 381.3 L/kg LOI). In B2, B3 and B4 after 28 days of composting, the reduction of biodegradability in terms of biogas production was 409.2 L/kg LOI, 293.2 L/kg LOI and 261.8 L/kg LOI, which was 72.9%, 65.1% and 72.0% of the anaerobic biogas potential of the untreated waste, respectively. The average biogas production remaining after 28 days of composting treatment measured in the B2, B3 and B4 was 136.5 L/kg LOI. Therefore, gas emissions from the BMW were significantly reduced by composting.

6.5 Contents of Cellulose, Hemicellulose and Lignin

The results of fibre analysis for the 13 samples collected at the start and from both reactors (from day 0 to day 104) are presented in Figure 6.13a in terms of the percentage contents of NDF, ADF and ADL (lignin) in DM. In the same way, those results of fibre analysis for the 9 samples collected in B2, B3 and B4 from day 0 to day 28 are presented in Figures 6.13b to 6.13d. In each batch the two composting runs showed similar trends with respect to changes in NDF, ADF and ADL contents during composting. In B1, the contents of NDF and ADF were observed to increase from day 0 (42.9±0.7% DM and 34.5±1.2% DM) to day 10 (44.8±1.1% DM and 39.9±0.8% DM). They then decreased steadily to 33.8±0.6% DM and 30.4±0.6%

DM at the termination of composting experiments (day 104). In B2, the contents of NDF and ADF were observed to increase from day 0 to day 6, but then decreasecd. By the end of the experiment they (36.1±0.4% and 32.3±0.7%) were higher than those at the beginning (day 0), which were 34.2±0.7% and 26.3±0.1%, respectively. In B3, the contents of NDF and ADF fluctuated during composting with 44.9±0.6% and 35.8±0.3% at day 0 and 45.7±0.9% and 39.4±0.8% at the end of 28 days composting. The NDF and ADF contents of the D0 waste sample in B4 (57.1±1.1% and 47.4±2.4%) were the highest among the four batches. Also, in B4 the contents of NDF and ADF fluctuated during composting. At the end of composting they (52.5±0.9% and 45.4±0.4%) were slightly lower than the initial contents.

The lignin contents and the calculated results of the cellulose, hemicellulose contents in terms of both the percentage of DM and LOI are given in Table 6.5. Their changes during composting in the four batches are also presented respectively in Figures 6.14, 6.15 and 6.16. Generally, the percentage of cellulose and hemicellulose decreased during composting although noticeable discrete increases were observed to occur over certain periods of the composting process. For example, in B1 the cellulose content of R1D5 (33.4% DM) was higher than that at the start (D0, 28.6% DM); in B2 the cellulose content after 6 days (26.0% DM) was higher than that at the start (day 0) which was 20.4% DM. The reason for these few inconsistencies may be the general variation in the waste composition and the small sample sizes used for the tests (Section 5.1). The lignin contents of the untreated waste samples (D0) were around 6% DM in the four batches, and were observed to increase steadily during composting, which was as expected. The lignin content at the end of composting in B1-B4 was between 17.3±0.4% (B1) to 19.1±0. 8% (B4).

At the end of composting experiments in each batch the sum of cellulose, hemicellulose and lignin contents in LOI (the NDF contents in terms of LOI) were 58.2%, 58.3%, 67.1% and 69.2% for B1toB4 respectively. They were higher than the contents at the start, which were 50.3%, 40.9%, 53.4% and 66.3% for B1toB4

respectively (Figure 6.17). Because the cellulose and hemicellulose contents in terms of LOI were observed to decrease over the composting period compared with those contents at the beginning (day 0) in each batch, the lignin content increased more than the decrease of the cellulose and hemicellulose contents.

Table 6.5. Fibre contents during composting

Time	Lignin (%)[a]	Lignin (%LOI)	Cellulose (%)[a]	Cellulose (%LOI)	Hemice-llulose (%)[a]	Hemice-llulose (%LOI)	(C+H)/L[b]
B1D0	5.9	7.0	28.6±1.2	33.5	8.4±1.4	9.9	6.2
B1R1							
D5	6.5	7.8	33.4±0.5	39.8	5.8±0.8	6.9	6.0
D10	10.8	13.4	28.5±1.0	35.5	5.4±0.7	6.7	3.2
D20	11.8	17.0	21.7±0.8	31.3	5.4±1.1	7.8	2.3
D35	15.2	22.6	16.4±0.4	24.4	7.1±0.3	10.5	1.6
D68	16.7	26.5	14.3±0.6	22.8	6.7±0.5	10.7	1.3
D104	17.4	30.0	12.7±0.4	22.1	3.2±0.7	5.5	0.9
B1R2							
D5	6.5	7.9	24.5±0.8	29.7	5.4±0.9	6.6	4.6
D10	10.6	13.3	29.8±0.6	37.5	4.3±1.1	5.5	3.2
D20	11.9	15.9	22.9±1.3	30.6	4.7±1.1	6.3	2.3
D35	15.7	24.2	17.7±0.3	27.4	7.3±0.4	11.3	1.6
D68	15.9	25.4	14.1±0.5	22.5	3.9±0.5	6.2	1.1
B2D0	5.9	7.1	20.4±0.5	24.3	7.9±0.7	9.5	4.8
B2R1							
D6	13.1	17.9	25.2±0.8	34.5	5.9±0.8	8.0	2.4
D12	13.2	19.0	22.7±0.3	32.6	3.6±0.3	5.1	2.0
D20	16.1	25.1	15.1±0.9	23.6	3.5±0.9	5.5	1.2
D28	17.9	29.9	15.1±0.4	25.1	3.7±0.4	6.1	1.0
B2R2							
D6	13.1	18.2	26.7±0.5	37.0	4.9±0.5	6.7	2.4
D12	13.6	19.1	21.8±0.5	30.7	6.1±0.5	8.6	2.1
D20	16.9	25.3	18.2±0.3	27.2	4.0±0.4	6.0	1.3
D28	17.7	27.7	13.7±0.8	21.4	4.1±0.6	6.4	1.0

Time	Lignin (%)[a]	Lignin (%LOI)	Cellulose (%)[a]	Cellulose (%LOI)	Hemicellulose (%)[a]	Hemicellulose (%LOI)	(C+H)/L[b]
B3D0	6.6	7.9	29.2±0.4	34.6	9.2±0.7	10.9	5.8
B3R1							
D4	11.6	14.6	34.5±0.9	43.6	5.9±0.5	7.4	3.5
D8	14.5	19.1	34.5±0.9	45.6	3.0±0.2	4.0	2.6
D17	14.7	19.9	24.2±1.0	32.6	6.0±0.7	8.1	2.0
D28	17.9	25.3	18.7±0.7	26.6	6.5±0.8	9.3	1.4
B3R2							
D4	9.6	12.6	26.2±0.7	34.2	1.7±0.7	2.3	2.9
D8	12.1	16.0	28.6±0.3	37.7	7.0±0.6	9.3	2.9
D17	14.7	21.0	27.0±1.6	38.6	1.7±1.2	2.4	2.0
D28	18.4	27.8	23.7±0.6	35.8	6.1±0.8	9.3	1.6
B4D0	6.2	7.3	41.2±2.4	47.8	9.7±2.6	11.2	8.1
B4R1							
D4	12.4	14.7	35.8±1.1	42.7	10.2±0.7	12.1	3.7
D8	14.7	18.2	32.2±0.8	39.9	10.6±1.3	13.1	2.9
D16	19.6	25.4	33.1±1.7	42.8	7.9±0.9	10.2	2.1
D28	19.0	25.3	25.9±0.9	34.4	7.1±0.5	9.4	1.7
B4R2							
D4	13.4	15.7	35.5±0.9	41.6	10.0±1.1	11.8	3.4
D8	18.3	22.5	35.3±0.4	43.6	7.8±0.5	9.6	2.4
D16	18.4	22.8	28.9±1.0	35.8	9.4±0.7	11.6	2.1
D28	19.2	25.1	26.6±0.3	34.7	7.2±0.7	9.4	1.8

[a] Expressed as % of DM; [b]Based on %DM.

The (C+H)/L ratio decreased steadily during composting (Figure 6.18). In B1, the (C+H)/L ratio decreased rapidly in the first 20 days of composting from 6.2 to 2.3; then it decreased slowly to an average of 1.0 in the following 84 days at the termination of the composting runs. The (C+H)/L ratio decreased from 4.8 to an average of 1.0 in B2, from 5.8 to 1.5 in B3, and from 8.1 to 1.7 in B4 over 28 days of composting. The cellulose to lignin (C/L) ratio at the termination of composting in B1 was 0.7, which is close to the C/L ratio of 0.8 recorded for 8 years old landfill refuse reported by Bookter and Ham (1982).

The actual masses of cellulose, hemicellulose and lignin at the start and during the sampling events were calculated from their contents in DM (Table 6.5) and the mass

of DM. The results for B1 and B2-B4 are given in Table 6.6a and 6.6b. The proportions of the degradation of cellulose, hemicellulose and lignin during the course of sampling intervals were interpreted as the reduction of the masses respectively (Table 6.6a and 6.6b). The reduction of cellulose (or hemicellulose, lignin) during the interval between two successive samplings was calculated as the mass of cellulose (or hemicellulose, lignin) degraded in that interval divided by the mass of cellulose (or hemicellulose, lignin) at the start of that period. For example, in B1 the reduction (rdc) of cellulose during day 11 to day 20 was calculated as following:

$$\text{Cellulose rdc}_{(D11\text{-}D20)}\,\% = \frac{Cellulose_{D10} - Cellulose_{D20}}{Cellulose_{D10}} \times 100 \qquad (6.5a)$$

Where, Cellulose_{D10} : the mass of cellulose (g) remaining in the reactor *after* sampling at day 11 (10 days composted); Cellulose_{D20} : the mass of cellulose (g) remaining in the reactor *before* sampling at day 21 (20 days composted).

Since samples were collected throughout the composting process, the total reduction in cellulose from the start (day 0) to the end was calculated by Equation 6.5b which accounts for the cellulose loss by sampling. The total reduction of hemicellulose and lignin from the start to the end was calculated in the same way. Some detailed examples are given in Appendix E.

$$\text{Cellulose rdc}\,\% = \frac{Cellulose_{D0} - Cellulose_{end} - \sum Sampled\ \ Cellulose}{Cellulose_{D0}} \times 100 \qquad (6.5b)$$

Generally, the mass of cellulose decreased during composting. In B1, the total cellulose degradation of 512.9 g (mean value of the two runs) was measured over 104 days of composting, which was 68.6% (mean value of the two runs) of the cellulose presented at day 0. The maximal cellulose reduction occurred during the sampling interval of day 10 to day 35. In B2, 248.5± 4 g cellulose was measured to be degraded over 28 days, which was 52.8% of the cellulose presented at day 0 (470.2 g). The cellulose reduction was maximal at 35.0 % over day 12 to 20 in R1 and 33.3% over day 6 to 12 in R2. In B3, 254.6±10.4 g cellulose was measured to be degraded over 28 days, which was 47.1% of the cellulose at day 0. The cellulose reduction was greatest at 37.4 % from day 8 to 17 in R1 and 23.5% from day 17 to 28 in R2. In B4, 369.5±9.4 g cellulose was measured to be degraded over 28 days, which was 53.3% of the cellulose at day 0. The cellulose reduction was greatest at 33.5 % over day 16 to 28 in R1 and at 36.7% over day 8 to 16 in R2.

The mass of hemicellulose was also seen to decrease in all four batches. In B1 after 104 days of composting, the degradation of hemicellulose was measured to be 72.9±0.6 % (the mean value of the two runs) of the mass at the start (220.5 g). In B2, 129.4±6.4 g was detected to be degraded B2 over 28 days of composting, which was 70.7 % of hemicellulose at day 0 (182.9 g). In B3 and B4, 106.6±8.2 g (B3) and 71.2±1.6 g (B4) hemicellulose was measured to be degraded over 28 days, which was 62.7% and 43.8% of the amount present at day 0. In all four batches, hemicellulose was the most degradable compound among the three compounds.

During some periods of the composting processes, the mass of cellulose and hemicellulose were found to have increased. For example, the mass of cellulose was measured to increase by 10.1% from 747.7 g (day 0) to 823.5 g (day 5) in B1R1; the mass of hemicellulose increased from 63.0 g (B1R1) and 61.7 g (B1R2) at day 21 to 71.0 g (B1R1) and 79.8 g (B1R2) at day 35 with the increases of 12.6% (B1R1) and 29.4% (B1R2). These discrepancies were caused either by the general variation in the waste composition and/or the small sample sizes used for the tests, or by the

production of some newly synthesised stabilized microbial components as suggested by Morel *et al.* (1985).

The mass of lignin was detected to decrease during some periods of composting in all four batches, for example, in B2 16.8% (31.5±3.1g) of the lignin at day 6 was removed during the composting period from day 6 to 12. However, compared to the amount of lignin at the start, the mass of lignin was found to increase by the end of composting in all four batches. It increased by 21.7% (33.7±6.8 g, mean value of the two runs) of that at the start, 53.4% (73.0±9.3 g) in B2, 51.0% (62.5±4.9 g) in B3, and 91.6 % (96.4±5.1 g) in B4.

Table 6.6a. Reduction of fibre mass during composting (B1)

Time		DM(g)	Cellulose (g)	Cellulose rdc(%)	Hemicel-lulose(g)	Hemicel-lulose rdc(%)	Lignin(g)	Lignin rdc(%)
	D0	2618.9	747.7		220.5		155.3	
B1R1								
Before	D5	2467.7	823.5	-10.1	142.4	35.4	161.1	-3.8
After	D5	2297.9	766.8		132.6		150.1	
Before	D10	1750.4	499.6	34.9	94.2	29.0	188.7	-25.8
After	D10	1623.8	463.4		87.4		175.0	
Before	D20	1292.0	280.8	39.4	69.5	20.4	152.6	12.8
After	D20	1171.9	254.7		63.0		138.4	
Before	D35	1006.0	164.7	35.3	71.0	-12.6	152.5	-10.2
After	D35	888.8	145.5		62.7		134.7	
Before	D68	807.1	115.7	20.5	54.3	13.4	134.6	0.1
After	D68	647.8	92.9		43.6		108.1	
	D104	581.4	74.1	20.3	18.5	57.5	100.9	6.7
Total			160.9[a]	68.6[b]	42.1[a]	72.5[b]	83.3[a]	-18.6[b]
B1R2								
Before	D5	2485.2	607.9	18.7	134.2	39.1	161.0	-3.7
After	D5	2314.8	566.2		125.0		150.0	
Before	D10	1767.1	526.9	6.9	76.7	38.6	187.7	-25.1
After	D10	1640.2	489.1		71.2		174.2	
Before	D20	1417.6	324.5	33.7	66.8	6.2	168.5	3.2
After	D20	1309.5	299.8		61.7		155.7	
Before	D35	1096.4	193.9	35.3	79.8	-29.4	171.6	-10.2
After	D35	979.7	173.3		71.3		153.3	
Before	D68	893.4	125.8	27.4	34.9	51.0	142.1	7.4
After	D68	752.1	105.9		29.4		119.6	
	D104	672.7	90.5	14.5	24.9	15.4	115.7	3.3
Total			144.8[a]	68.5[b]	33.8[a]	73.4[b]	78.1[a]	-24.8[b]

Before-before sampling; After-after sampling; rdc-Reduction;

[a]Total sampling ; [b]Total reduction from the start (day 1) to the end (day104)

Table 6.6b. Reduction of fibre mass during composting (B2, B3 and B4))

Time		DM(g)	Cellulose (g)	Cellulose rdc(%)	Hemicel-lulose(g)	Hemicel-lulose rdc(%)	Lignin (g)	Lignin rdc(%)
B2								
	D0	2306.3	470.2		182.9		136.5	
B2R1								
Before	D6	1582.8	398.2	15.3	92.8	49.3	206.6	-51.3
After	D6	1418.2	356.8		83.2		185.1	
Before	D12	1143.6	259.4	27.3	40.8	51.0	151.4	18.2
After	D12	961.2	218.1		34.3		127.3	
Before	D20	936.2	141.6	35.0	33.2	3.3	150.5	-18.3
After	D20	697.6	105.5		24.7		112.1	
	D28	662.9	100.0	5.2	24.4	1.2	118.9	-6.0
Total			118.9[a]	53.4[b]	24.6[a]	73.2[b]	84.0[a]	-48.6[b]
B2R2								
Before	D6	1616.6	431.5	8.2	78.6	57.0	212.3	-55.5
After	D6	1448.0	386.5		70.4		190.2	
Before	D12	1182.8	258.0	33.3	72.0	-2.3	160.9	15.4
After	D12	1019.9	222.4		62.1		138.7	
Before	D20	990.2	180.1	19.0	39.8	36.0	167.1	-20.5
After	D20	768.7	139.8		30.9		129.7	
	D28	759.1	103.8	25.8	31.1	-0.6	134.4	-3.6
Total			120.8[a]	52.2[b]	27.0[a]	68.3[b]	81.7[a]	-58.3[b]
B3								
	D0	1852.6	540.1		169.9		122.6	
B3R1								
Before	D4	1611.8	556.7	-3.1	94.4	44.5	186.4	-52.1
After	D4	1454.9	502.5		85.2		168.3	
Before	D8	1213.6	419.0	16.6	36.4	57.3	175.4	-4.2
After	D8	1059.1	365.7		31.7		153.1	
Before	D17	947.9	229.0	37.4	56.7	-78.9	139.5	8.9
After	D17	775.7	187.4		46.4		114.2	
	D28	687.9	128.9	31.2	45.0	3.1	122.8	-7.6
Total			149.1[a]	48.5[b]	24.1[a]	59.3[b]	65.8[a]	-53.9[b]
B3R2								
Before	D4	1601.8	418.9	22.4	27.7	83.7	154.2	-25.8
After	D4	1445.3	378.0		25.0		139.2	
Before	D8	1167.7	334.1	11.6	82.2	-228.4	141.6	-1.8
After	D8	997.1	285.3		70.2		120.9	
Before	D17	924.0	249.9	12.4	15.8	77.5	136.0	-12.4
After	D17	737.9	199.6		12.6		108.6	
	D28	644.9	152.7	23.5	39.6	-214.5	118.5	-9.1
Total			140.1[a]	45.8[b]	17.9[a]	66.1[b]	63.1[a]	-48.2[b]

Time		DM(g)	Cellulose (g)	Cellulose rdc(%)	Hemicel-lulose(g)	Hemicel-lulose rdc(%)	Lignin (g)	Lignin rdc(%)
B4								
	D0	1684.0	693.1		162.6		105.2	
B4R1								
Before	D4	1554.4	556.7	19.7	158.2	2.7	192.0	-82.5
After	D4	1408.5	504.4		143.4		174.0	
Before	D8	1253.8	403.7	20.0	132.7	7.5	184.7	-6.2
After	D8	1065.9	343.2		112.8		157.0	
Before	D16	906.2	299.9	12.6	71.4	36.7	177.9	-13.3
After	D16	742.8	245.8		58.5		145.8	
	D28	631.6	163.4	33.5	44.9	23.3	120.2	17.6
Total			166.9[a]	52.4[b]	47.6[a]	43.1[b]	77.8[a]	-88.1[b]
B4R2								
Before	D4	1561.7	554.3	20.0	155.9	4.1	207.7	-97.4
After	D4	1411.4	501.0		141.3		188.2	
Before	D8	1197.2	423.2	15.5	97.8	30.8	229.2	-21.8
After	D8	1015.8	359.1		83.2		194.9	
Before	D16	786.1	227.2	36.7	85.1	-2.4	166.8	14.4
After	D16	611.8	176.8		69.8		136.7	
	D28	560.6	149.0	15.7	45.6	34.6	121.3	11.3
Total			167.9[a]	54.3[b]	44.6[a]	44.5[b]	83.9[a]	-95.0[b]

Before-before sampling; After-after sampling; rdc-Reduction

[a]Total sampling ; [b]Total reduction from the start (day 1) to the end (day 28)

In summary, lignin contents of the untreated waste samples (D0) used in the four batches were close to each other ranging between 5.9% DM to 6.6 % DM with an average content of 6.2% DM. Cellulose contents exhibited a wider range of 20.4 - 41.2 % DM with an average of 29.8% DM. Hemicellulose contents were around 8-9% DM with an average of 8.8% DM. The lignin and hemicellulose contents of the untreated BMW samples in the four batches were close to those of the BMW used in the preliminary experiments, presented in Chapter 4. In all four batches, in terms of waste stabilization the change of the (C+H)/L ratio showed good consistency with those of oxygen consumption, anaerobic biogas production and TC/TN ratio.

After aerobic treatment, the contents of cellulose and hemicellulose decreased in terms of both DM and LOI, while the lignin content increased. The percentage of NDF in LOI in all the runs also increased after composting. In all four batches, cellulose and hemicellulose were degraded over most periods of composting, with a

few exceptions of a detected mass increase. Lignin degradation was also seen during some periods of the composting processes. The production of measured 'lignin' was also detected, which may be humic matter, a metabolic product during composting, and is co-measured in the 'lignin' fraction (Komilis and Ham, 2003). The production of measured 'lignin' was much higher than lignin degradation, resulting in the amount of lignin in the end of composting runs being higher than those at the beginning (D0). The relatively high lignin content of the compost at the end of the composting runs is not only due to the more extensive degradation of the other relatively easily degradable constituents, but also due to the production of measured 'lignin' during composting.

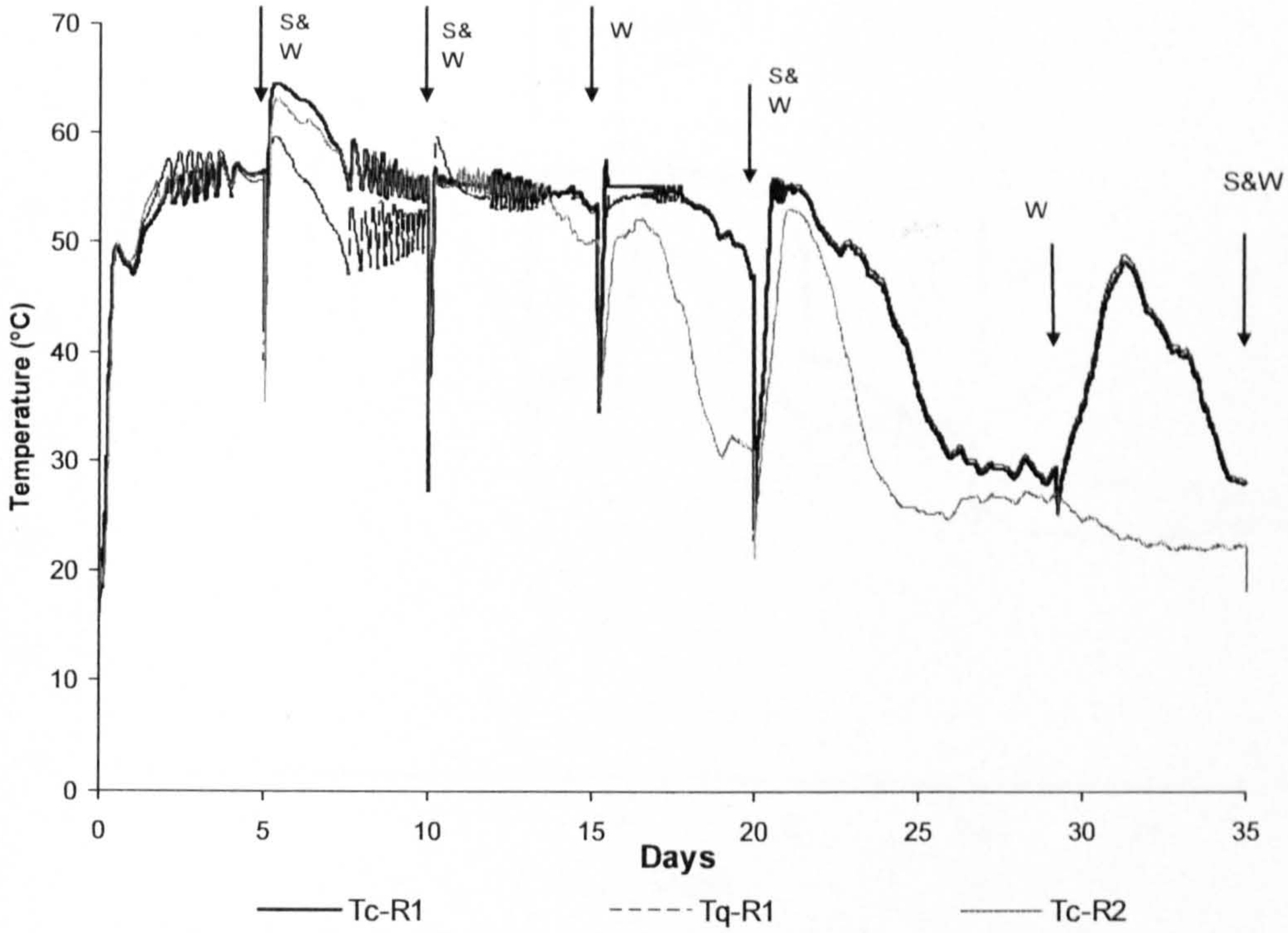


Figure 6.1a. Temperature changes during the first 35 days of composting in B1

(S-sampling, W- adding water; Tc-the temperature at the centre of the compost, T_q -the temperature at the quarter of the compost)

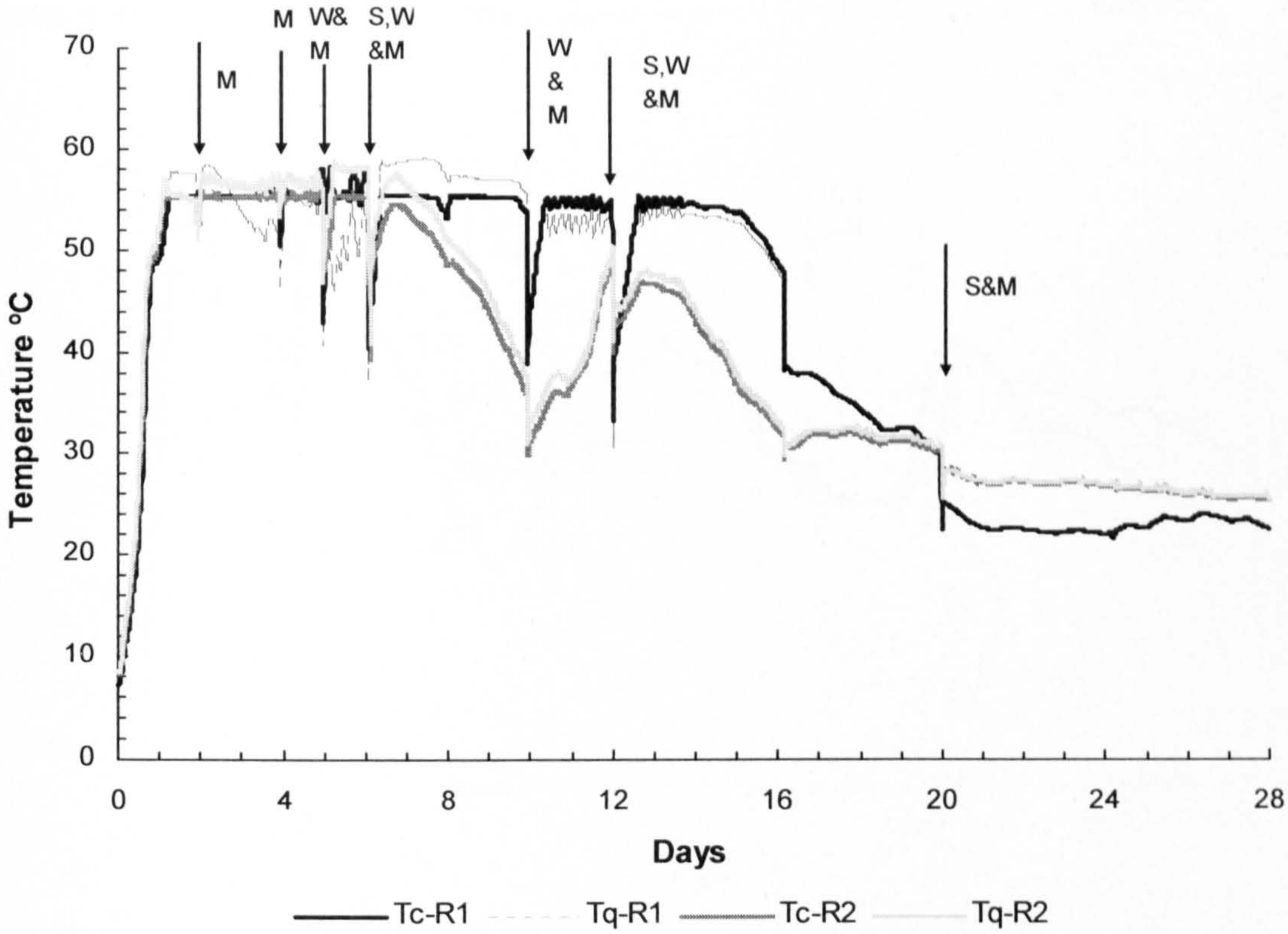


Figure 6.1b.Temperature changes of the B2 two runs

(M-mixing, W- adding water, S-sampling)

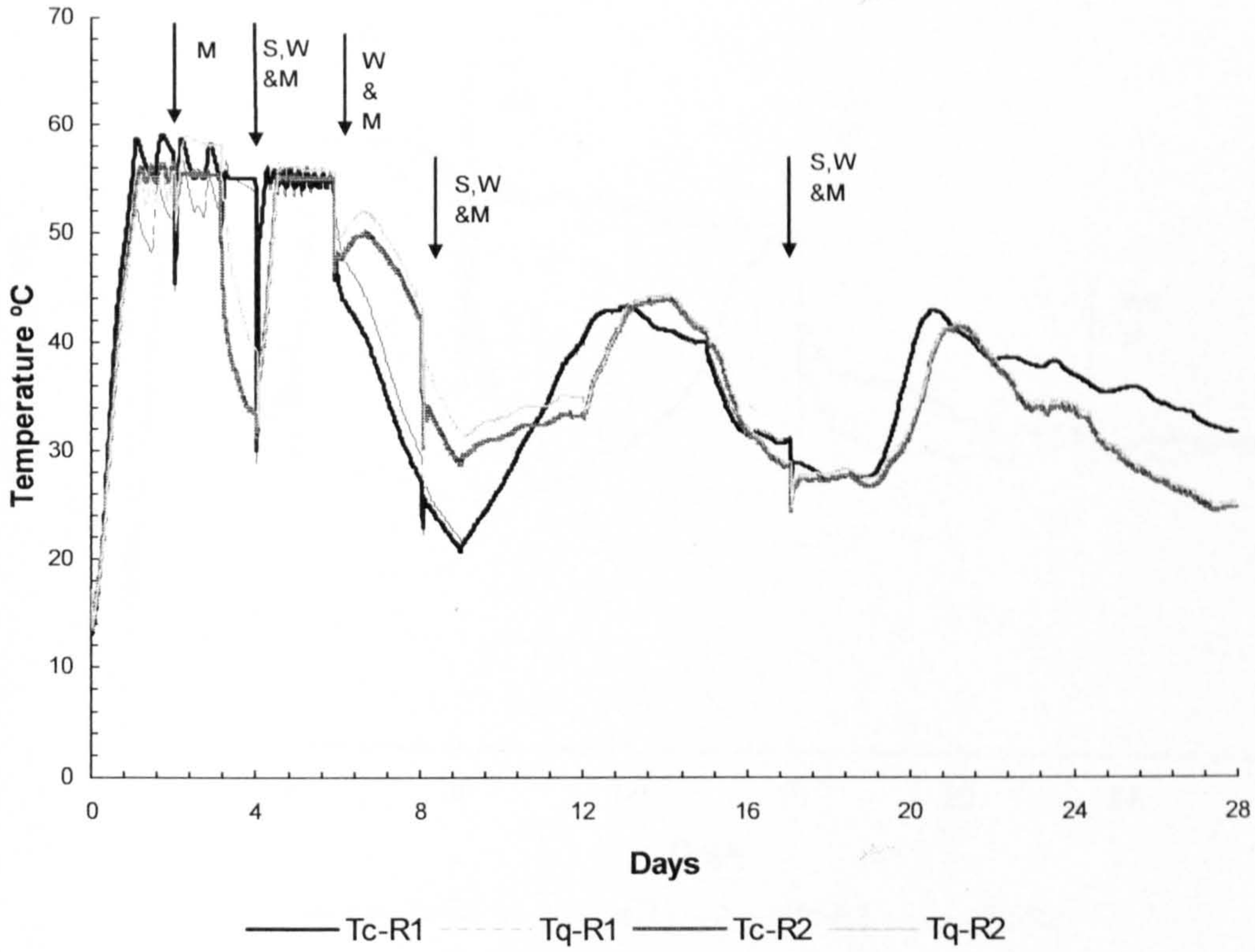


Figure 6.1c. Temperature changes of the B3 two runs

(M-mixing, W- adding water, S-sampling)

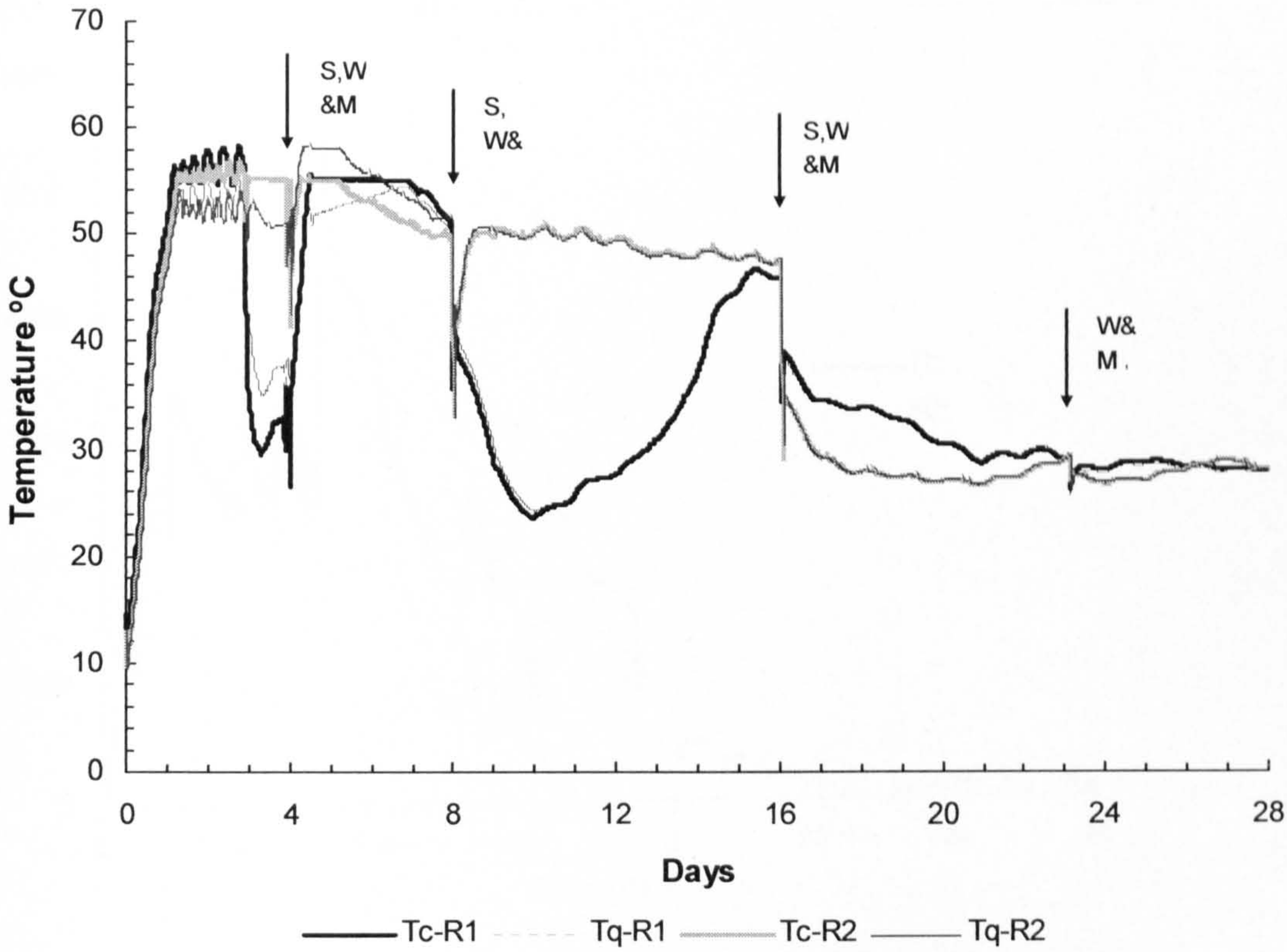


Figure 6.1d. Temperature changes of the B4 two runs

(M-mixing, W- adding water, S-sampling)

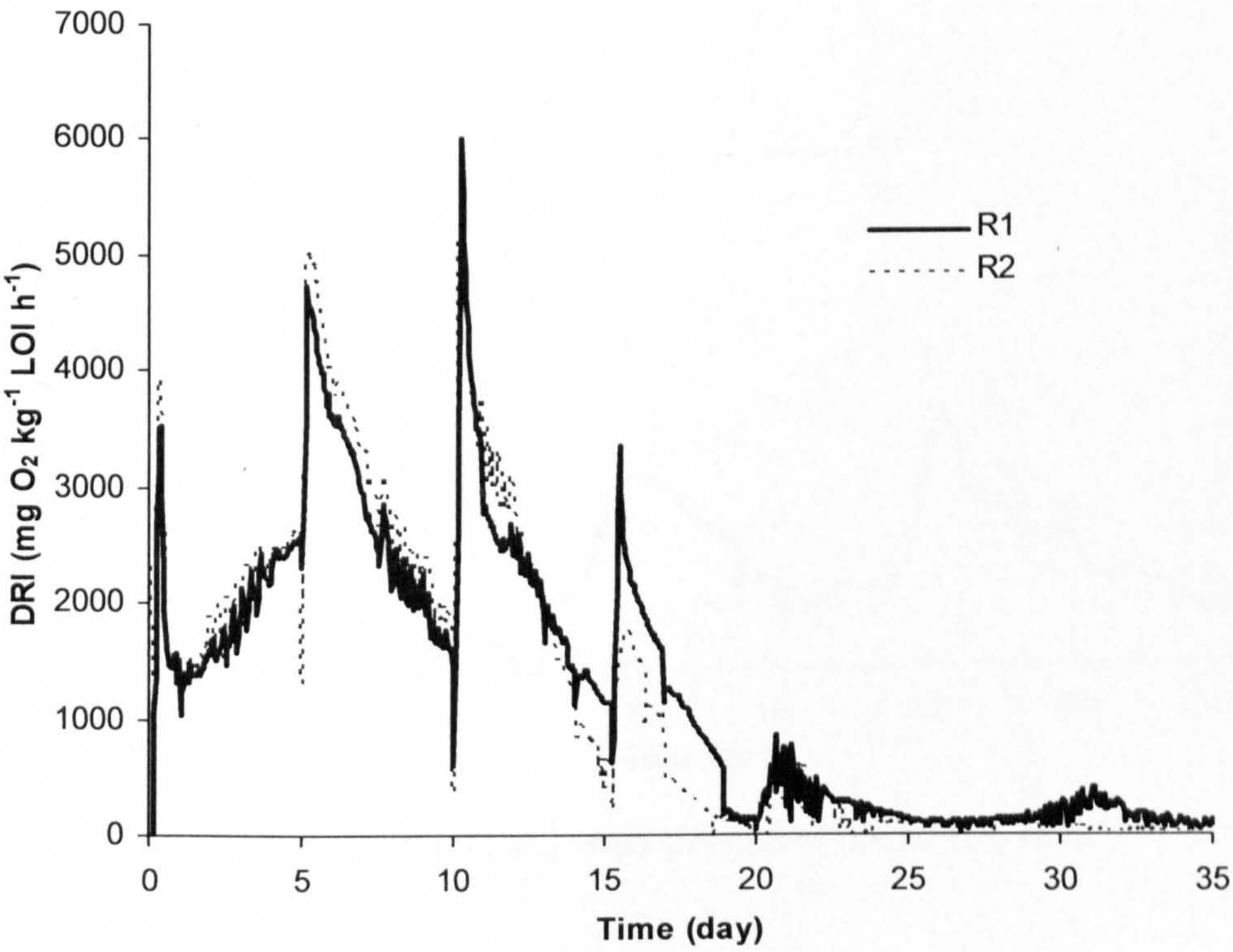


Figure 6.2a. Instantaneous DRI profiles of the B1 two runs

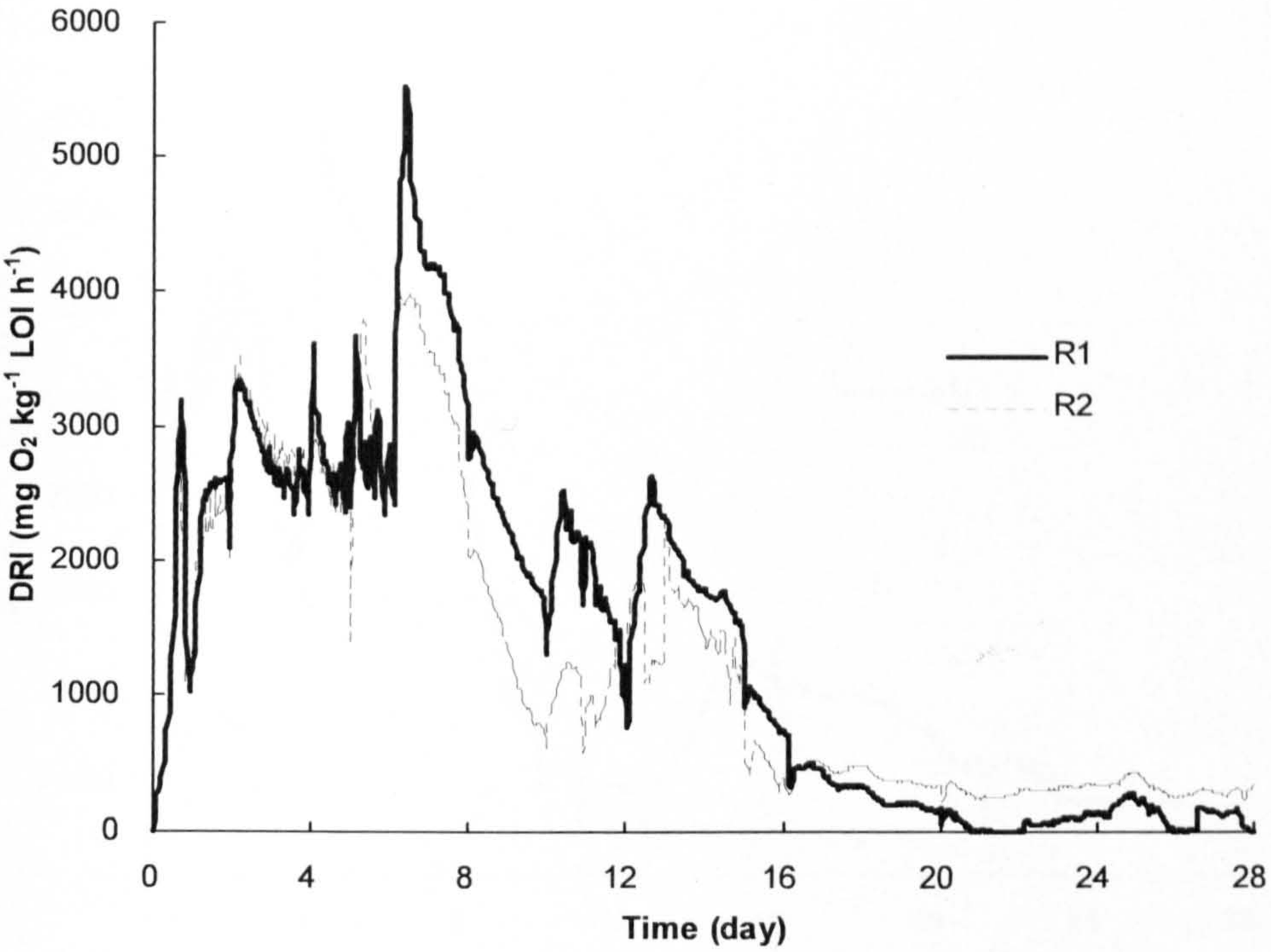


Figure 6.2b. Instantaneous DRI profiles of the B2 two runs

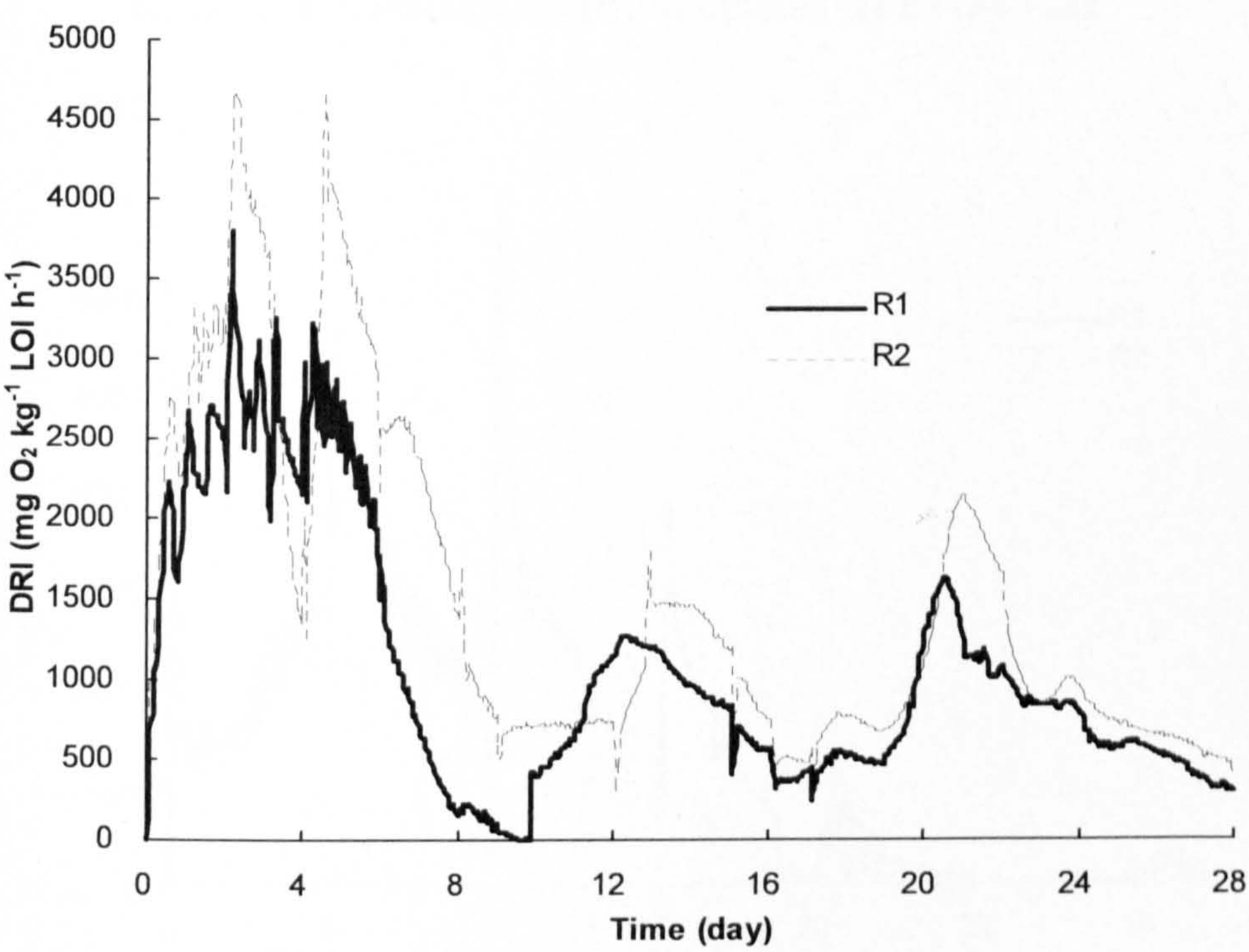


Figure 6.2c. Instantaneous DRI profiles of the B3 two runs

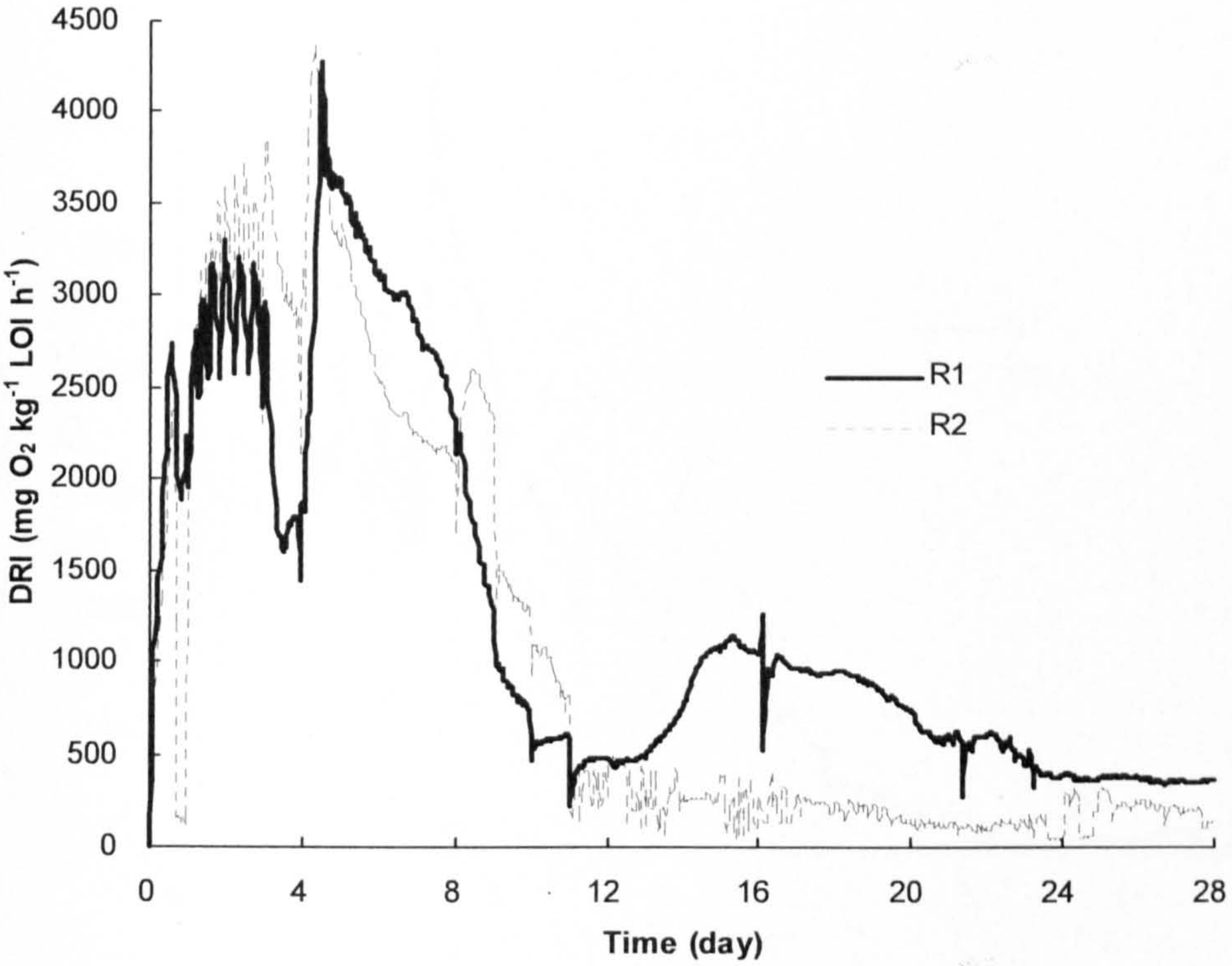


Figure 6.2d. Instantaneous DRI profiles of the B4 two runs

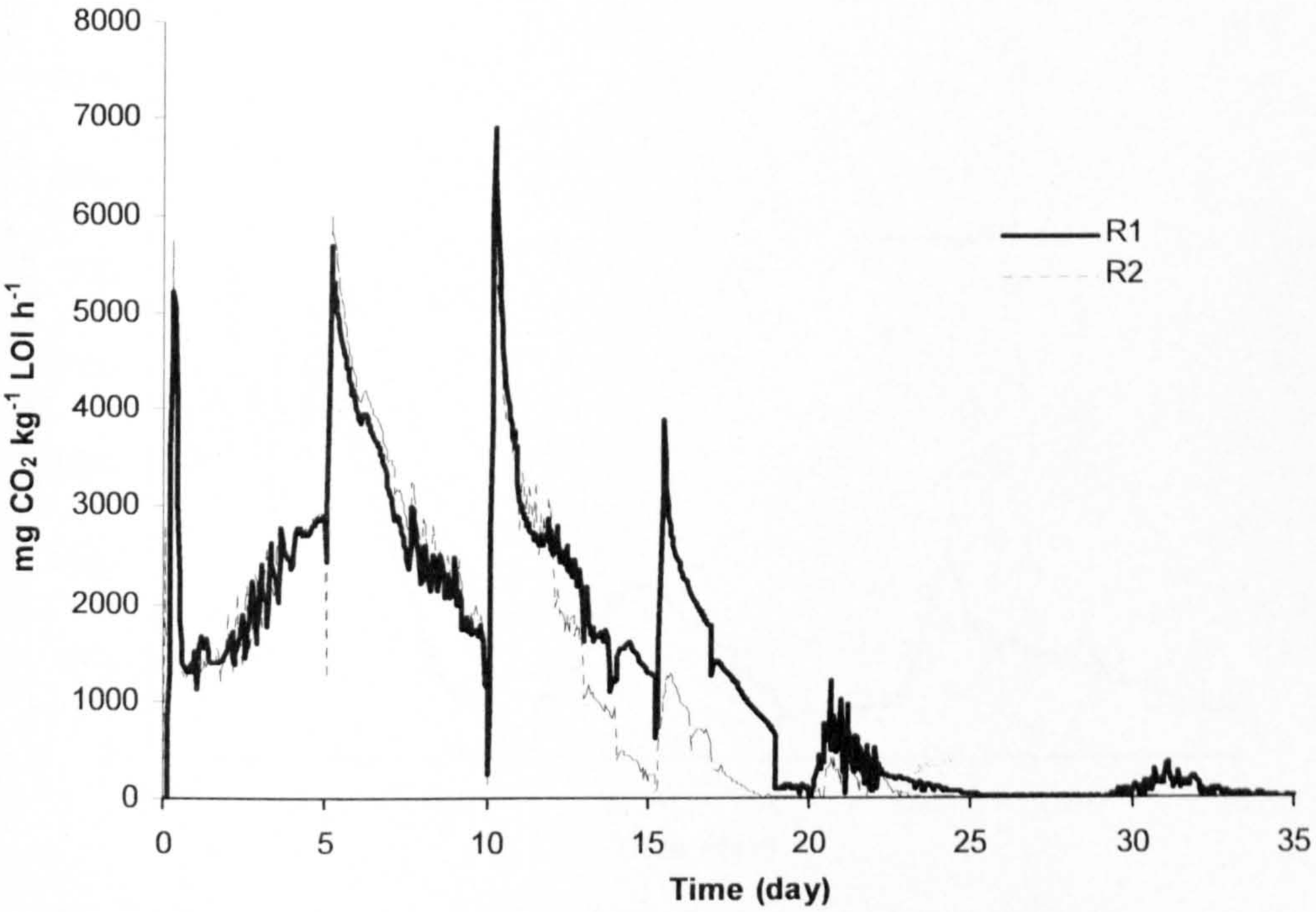


Figure 6.3a. Instantaneous CO_2 production rate profile of B1 two runs

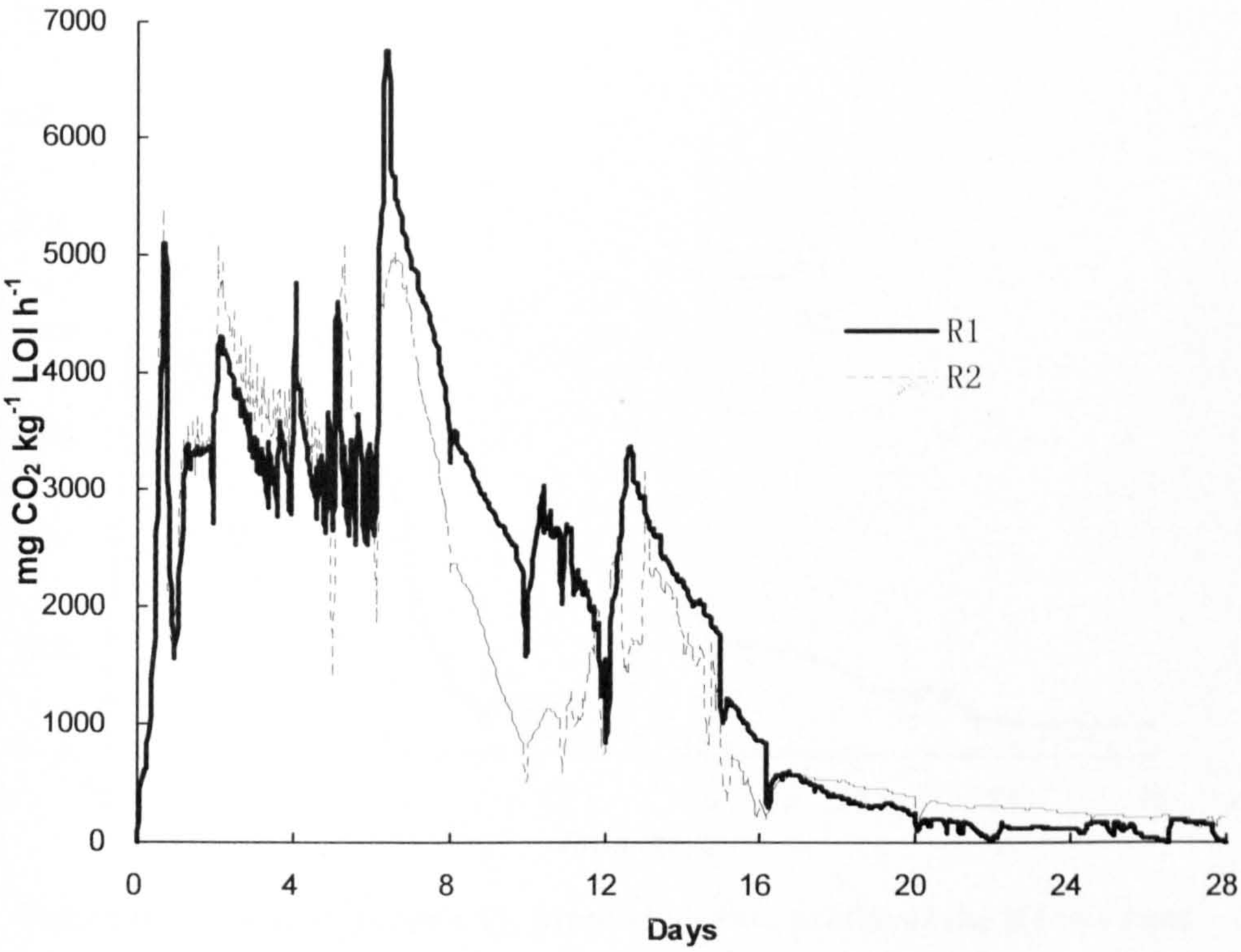


Figure 6.3b. Instantaneous CO_2 production rate profile of B2 two runs

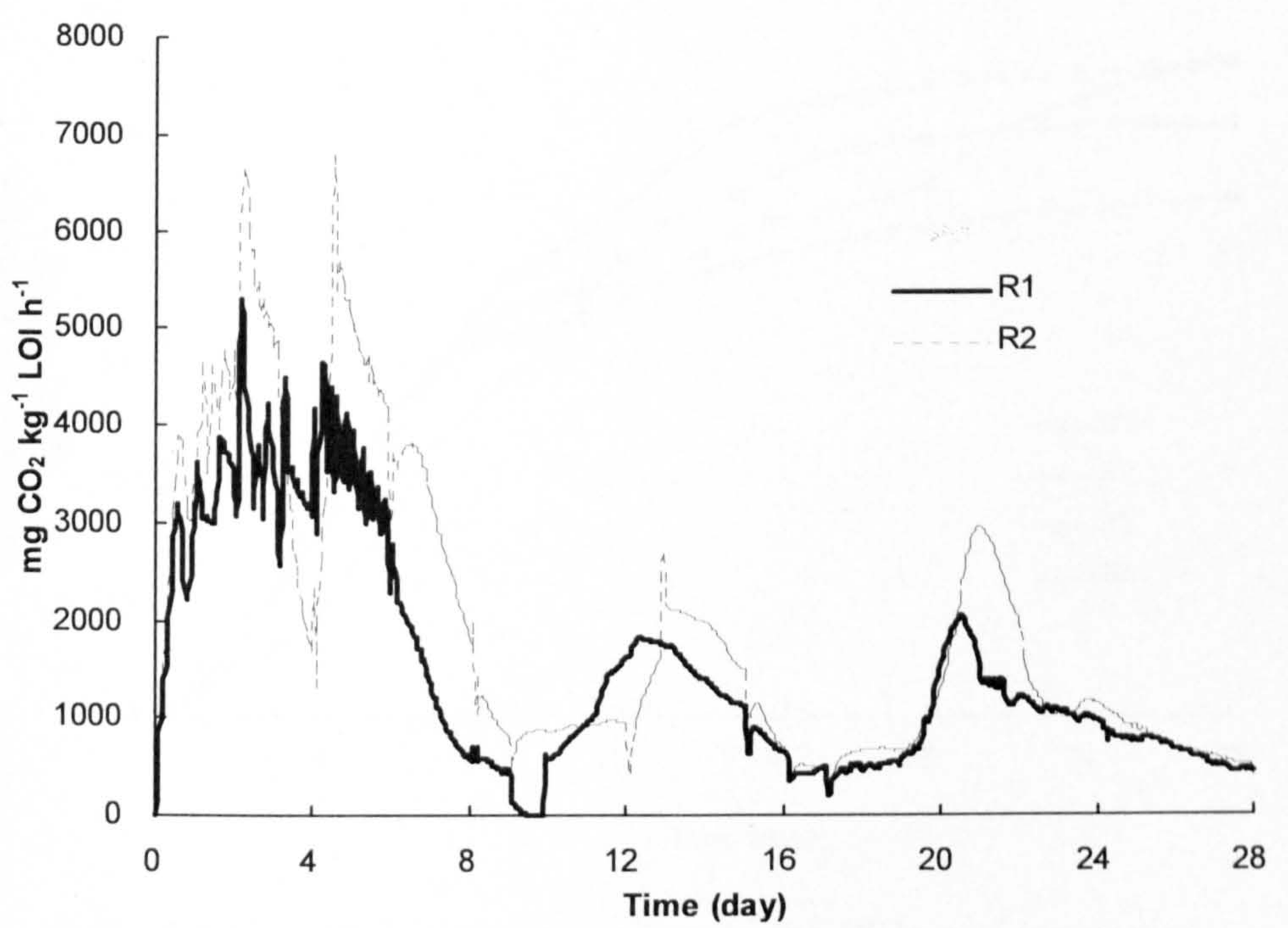


Figure 6.3c. Instantaneous CO_2 production rate profile of B3 two runs

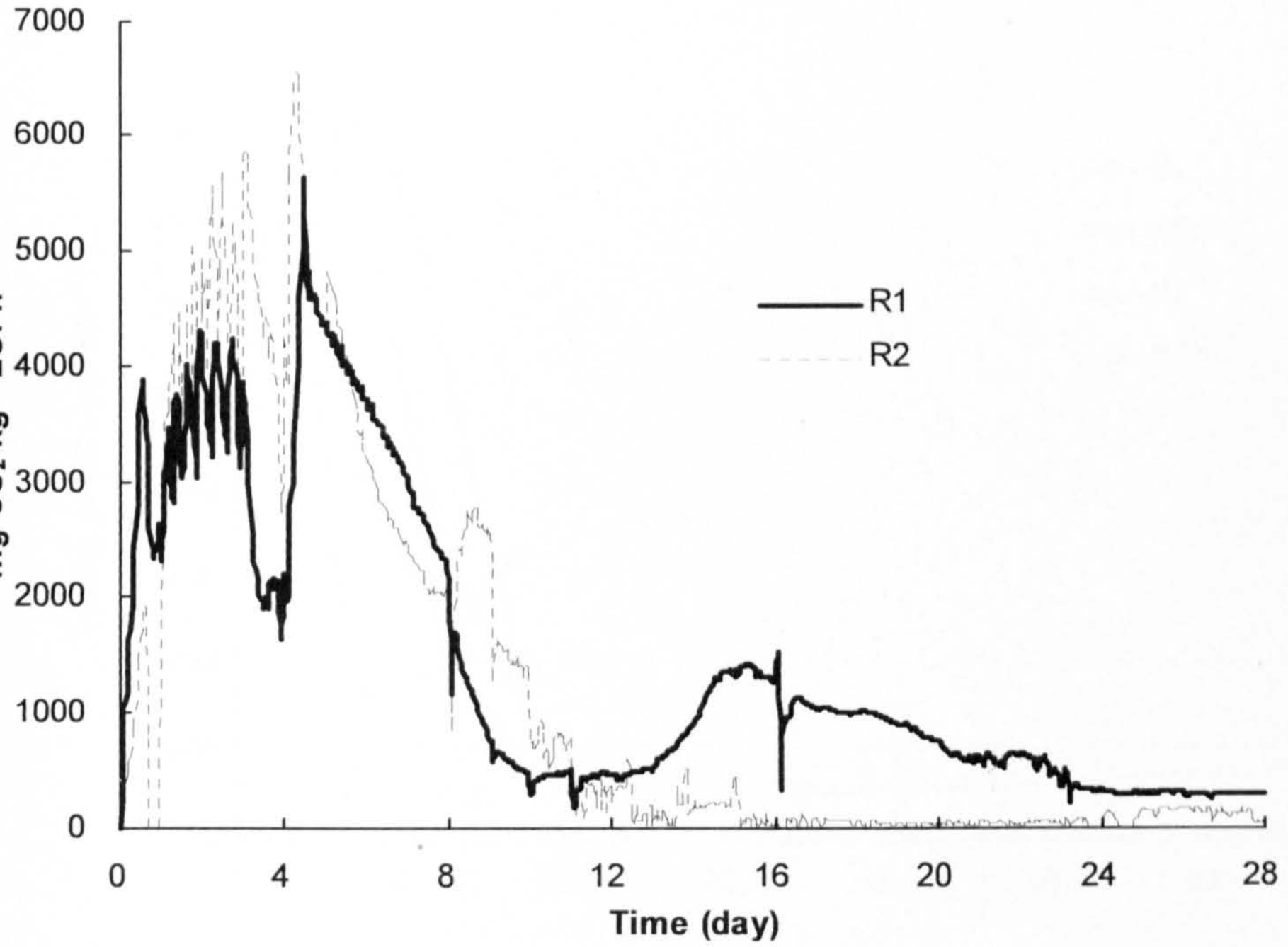


Figure 6.3d. Instantaneous CO_2 production rate profile of the B4 two runs

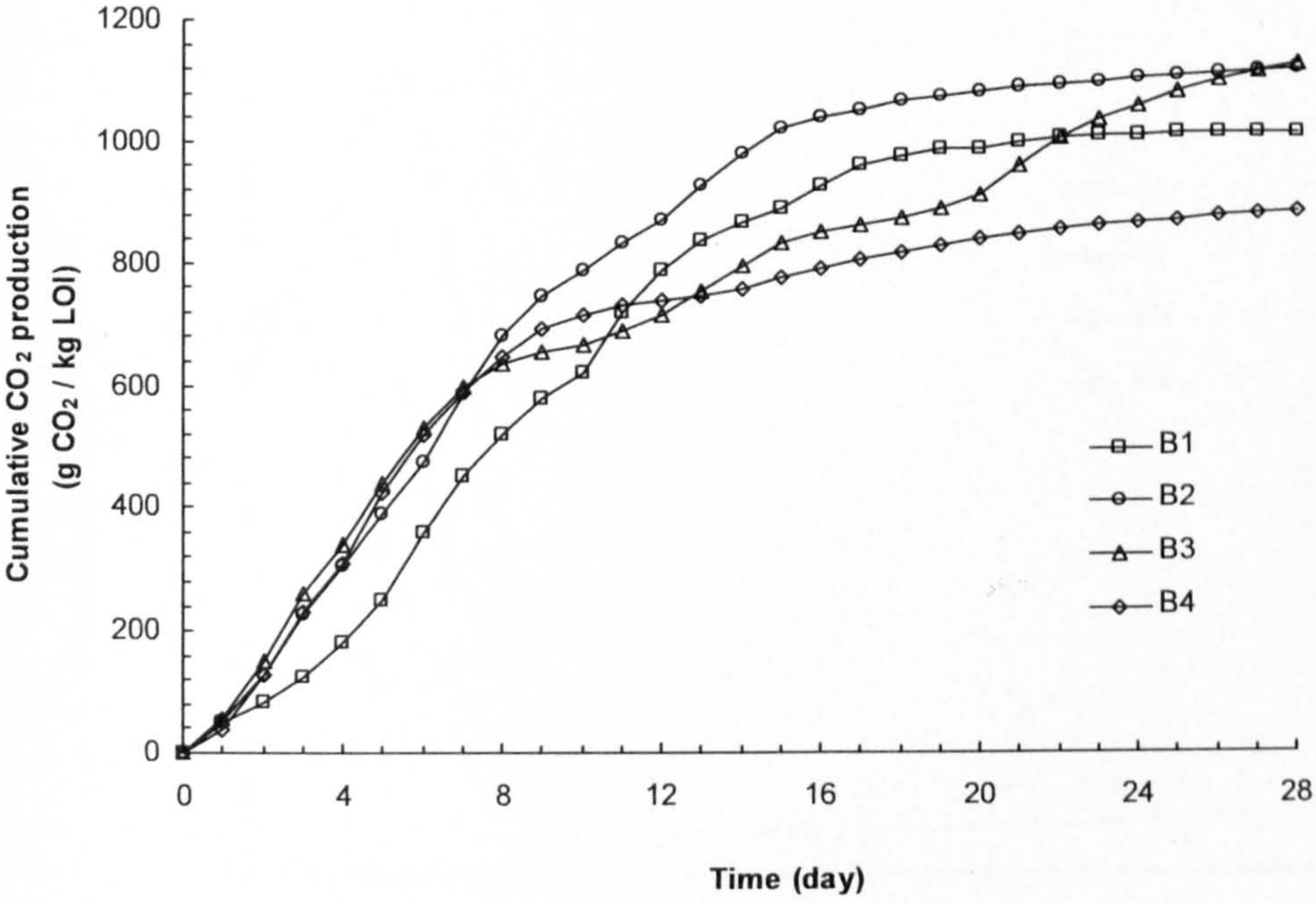


Figure 6.4. Cumulative CO_2 production during 28 days of composting

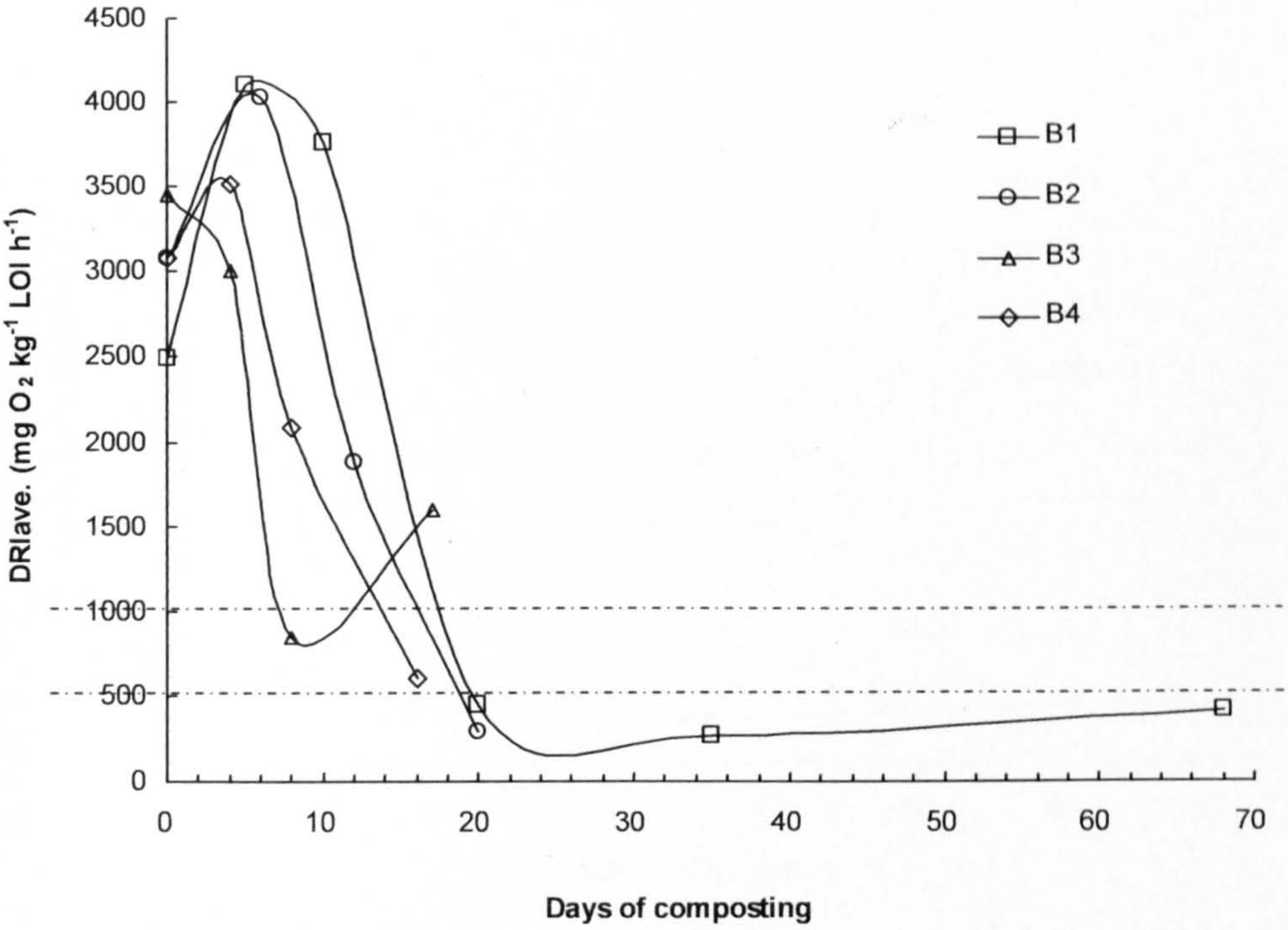


Figure 6.5. Change of stability expressed by $DRI_{ave.}$ during aerobic treatment process for the four batches

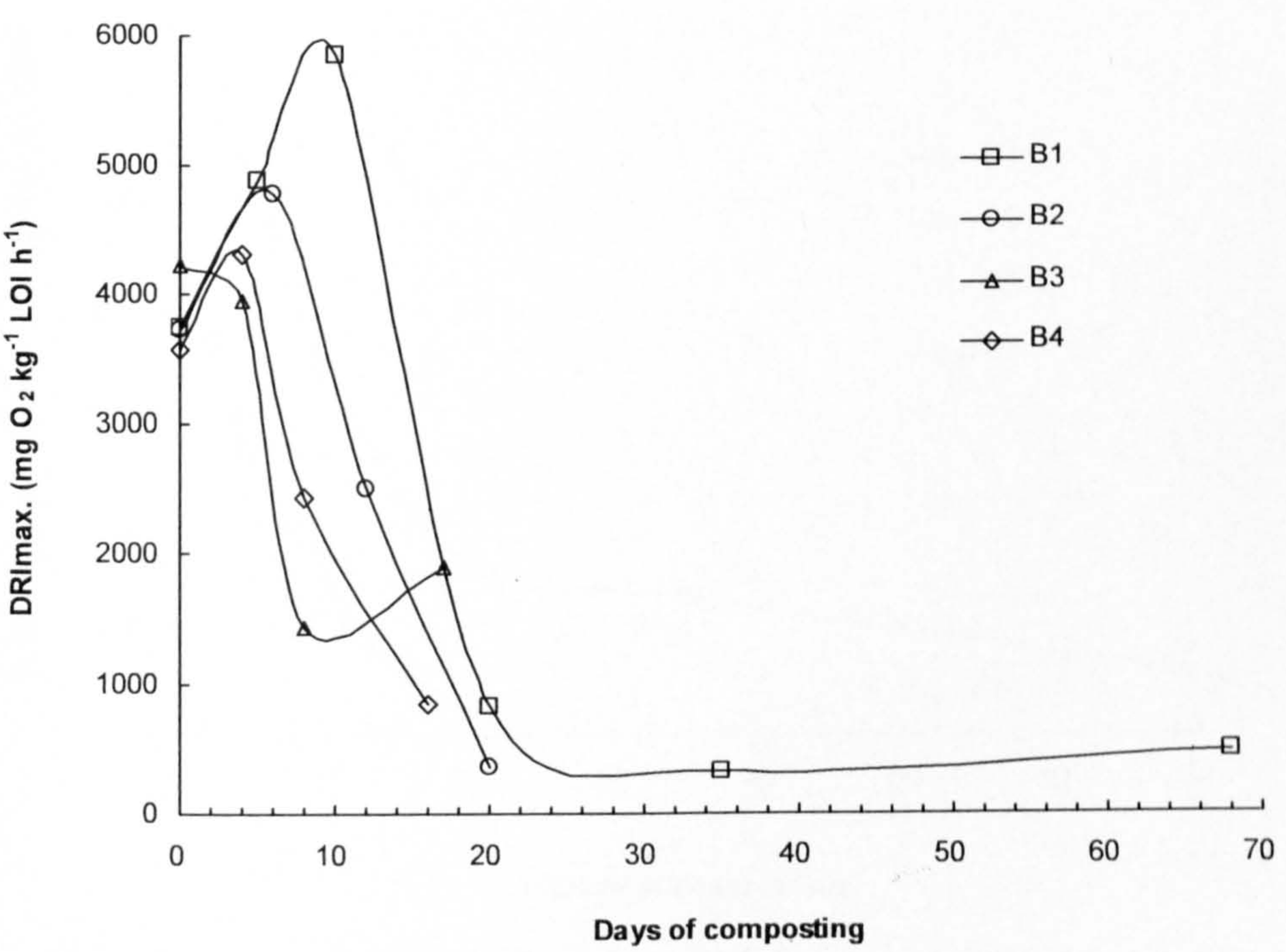


Figure 6.6. Change of stability expressed by $DRI_{max.}$ during aerobic treatment process for the four batches

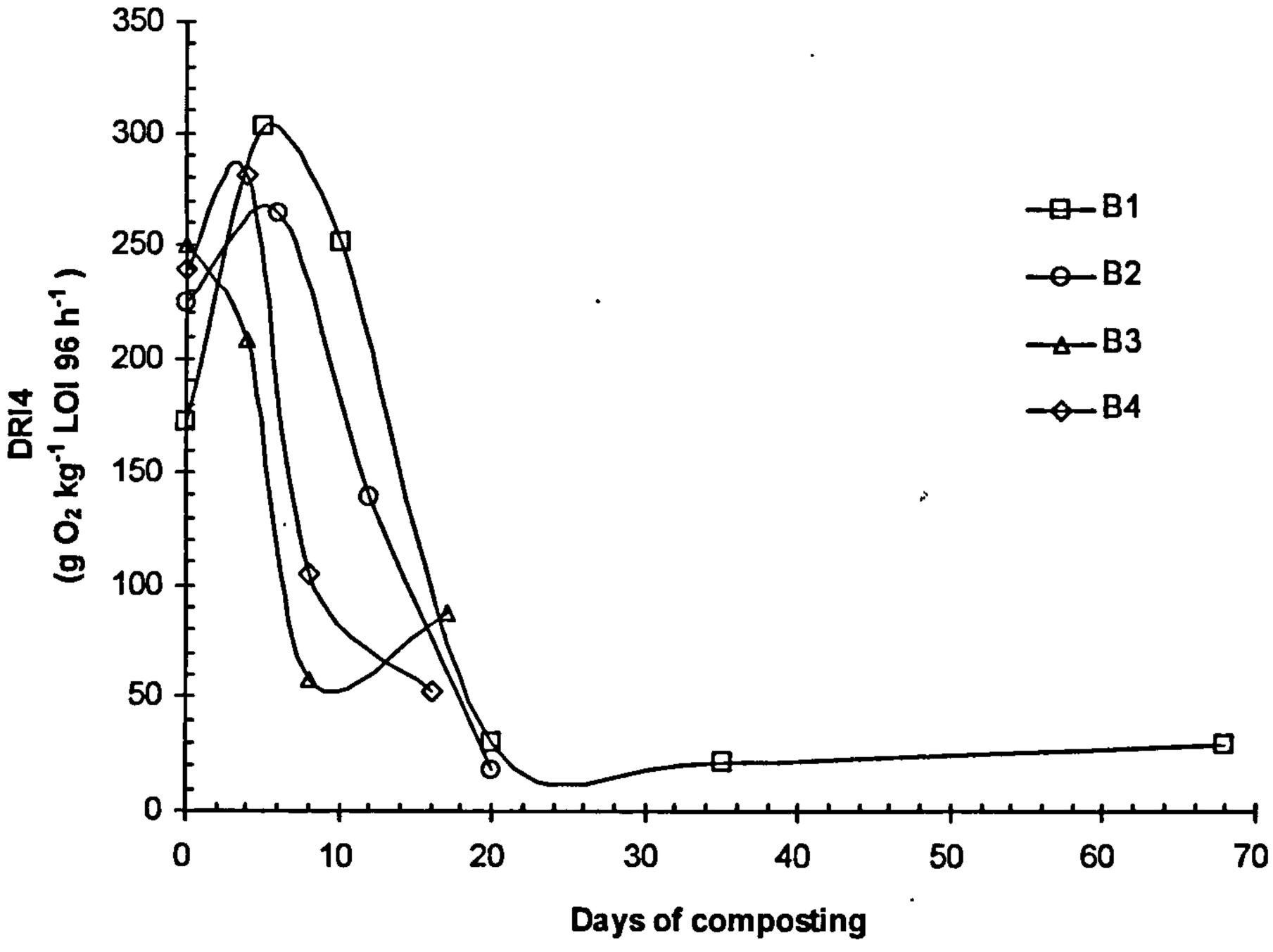


Figure 6.7. Change of stability expressed by DRI4 during aerobic treatment process for the four batches

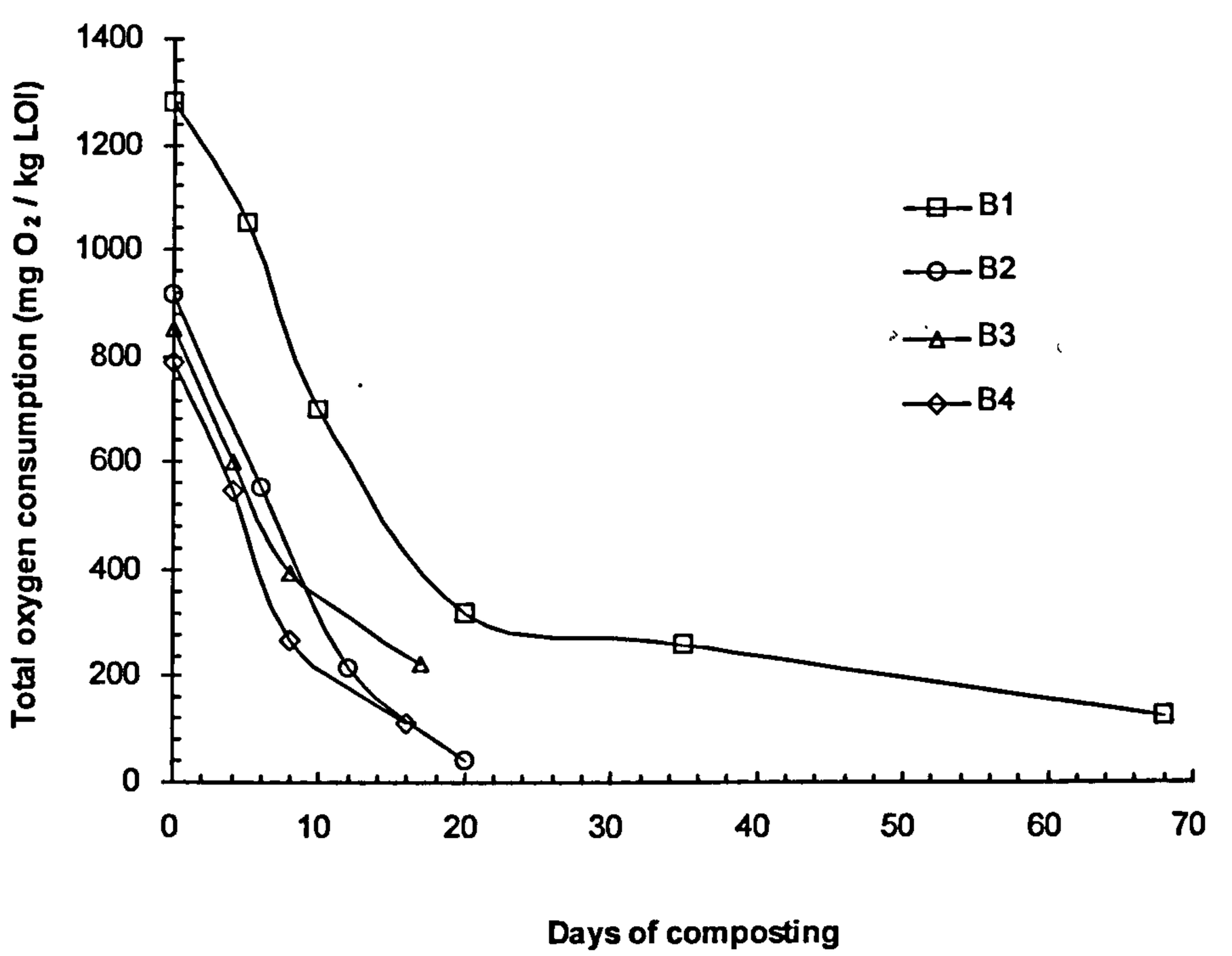


Figure 6.8. Change of stability expressed by $DRI_{tot.}$ during aerobic treatment process for the four batches

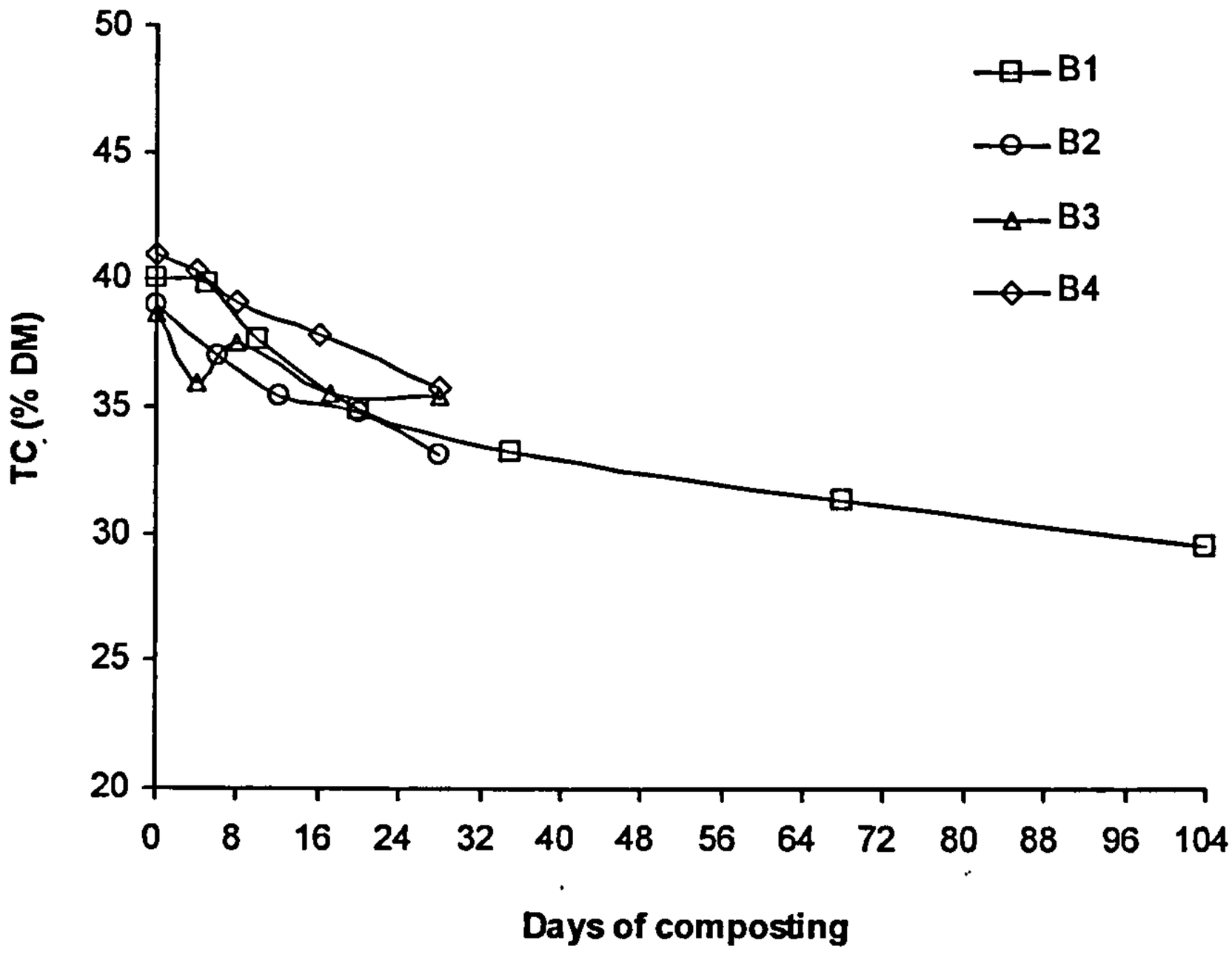


Figure 6.9. Change of total carbon contents during composting for the four batches

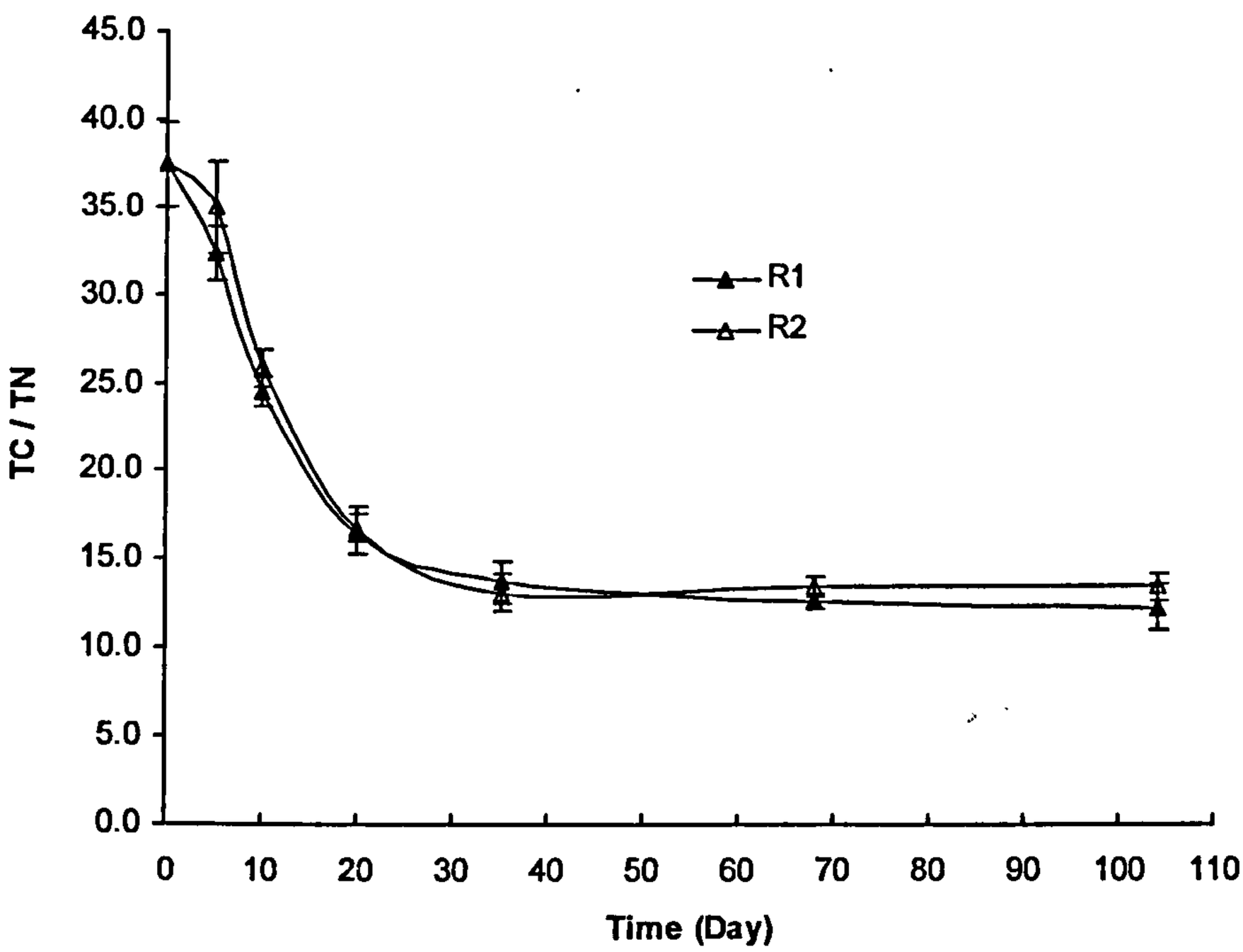


Figure 6.10a. Changes of total carbon to total nitrogen ratio in solids during composting in B1

(error bars represent the standard deviation of ratios)

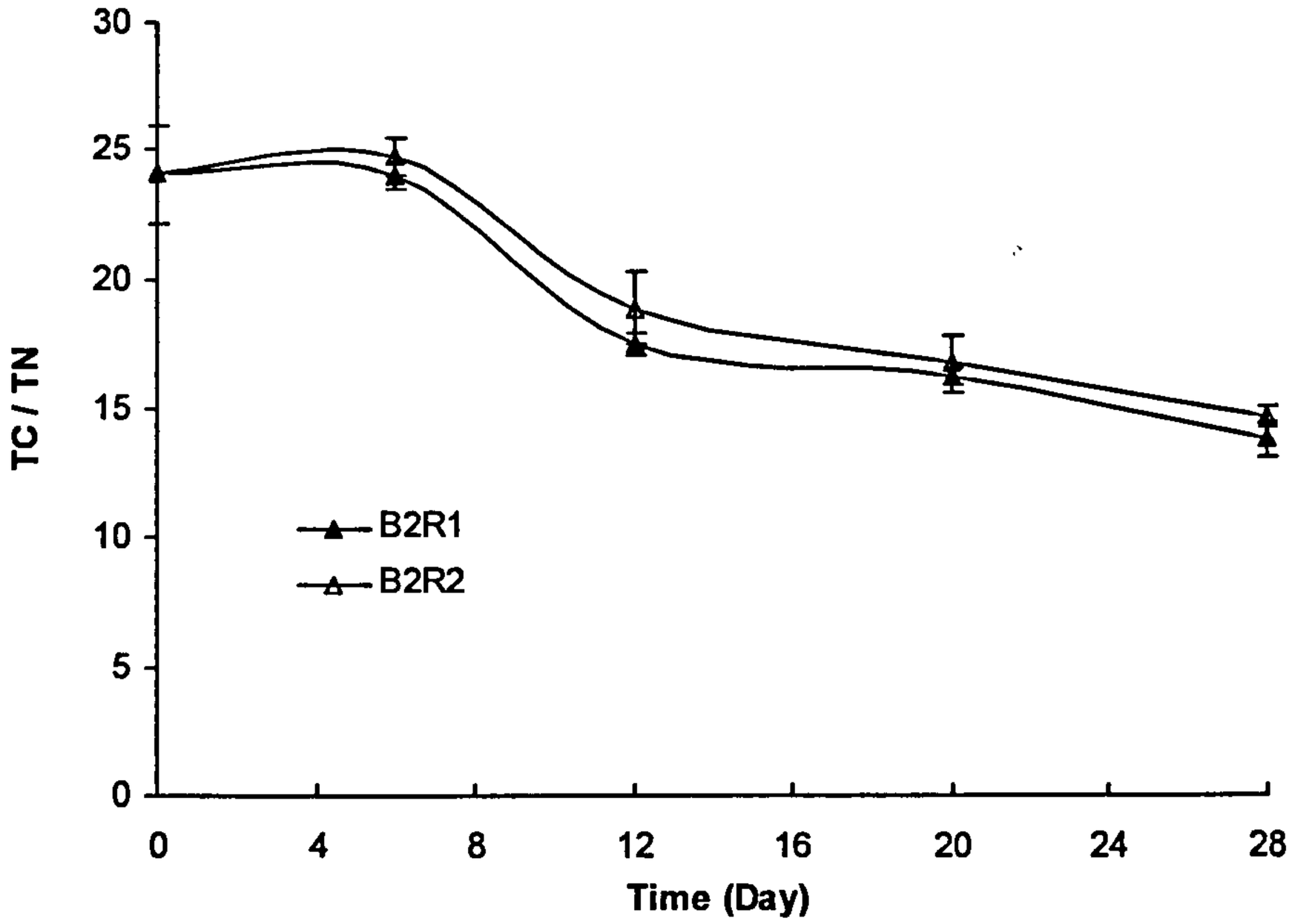


Figure 6.10b. Changes of total carbon to total nitrogen ratio in solids during composting in B2

(error bars represent the standard deviation of ratios)

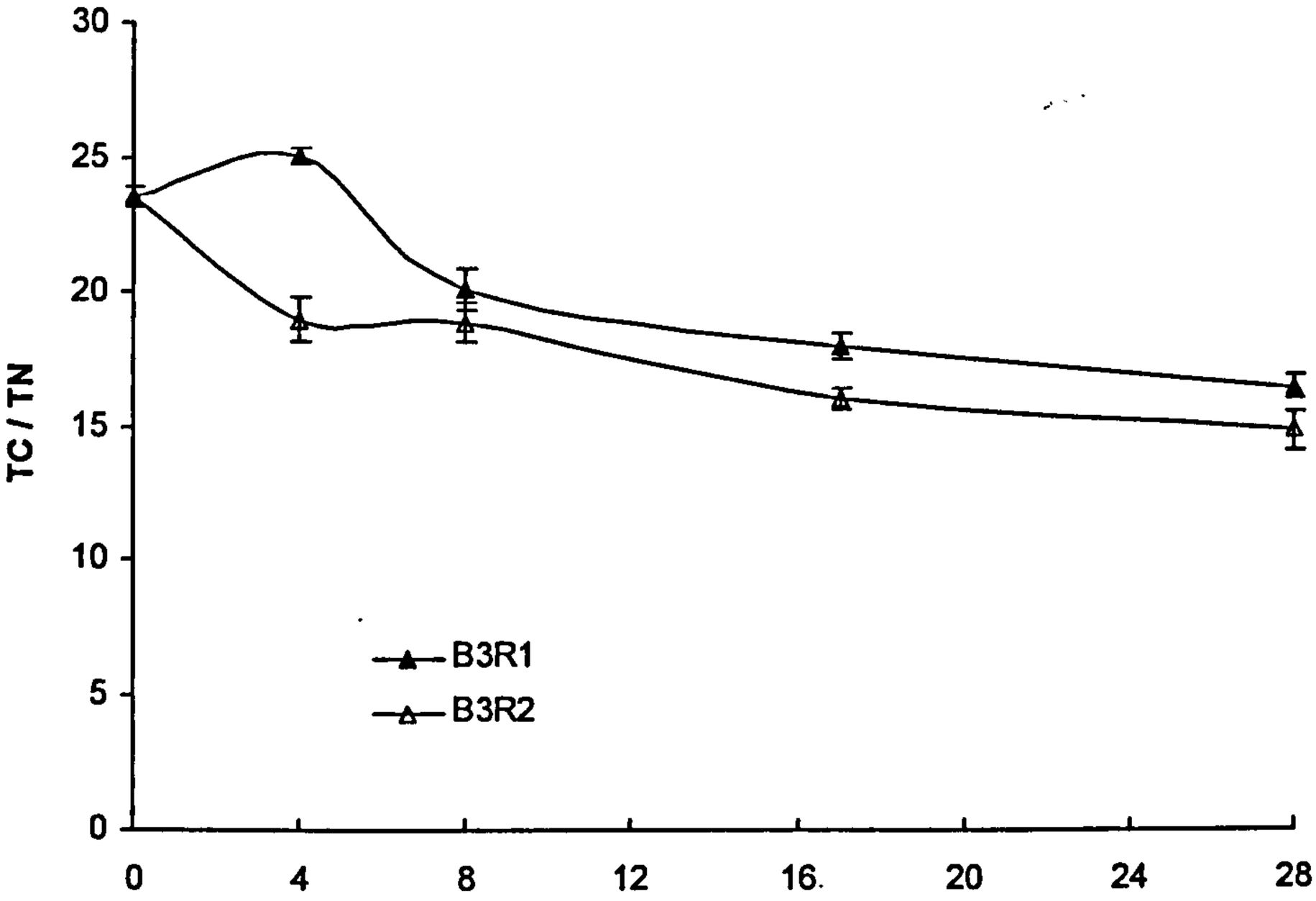


Figure 6.10c. Changes of total carbon to total nitrogen ratio in solids during composting in B3

(B3, error bars represent the standard deviation of ratios)

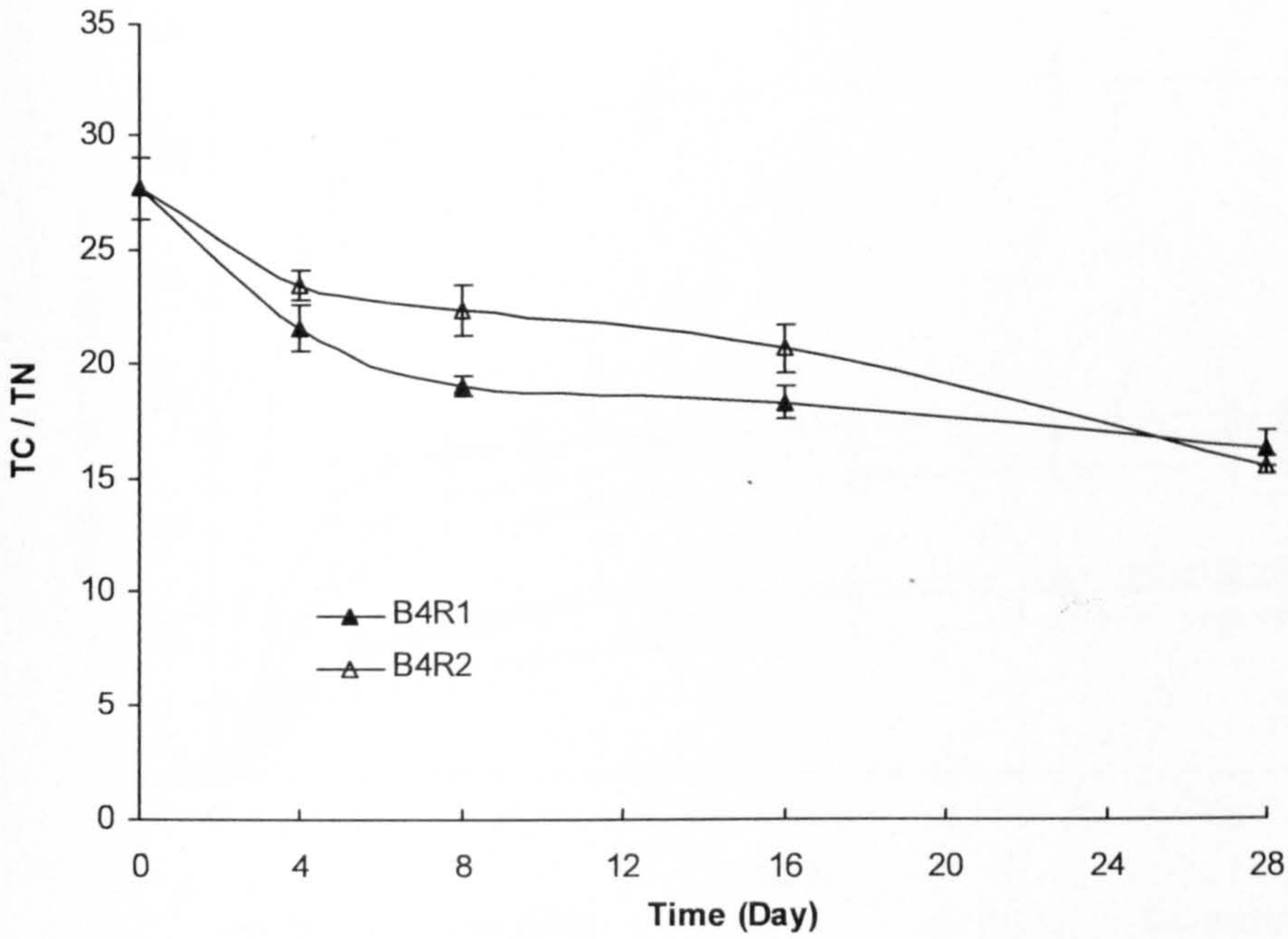


Figure 6.10d. Changes of total carbon to total nitrogen ratio in solids during composting in B4 (error bars represent the standard deviation of ratios)

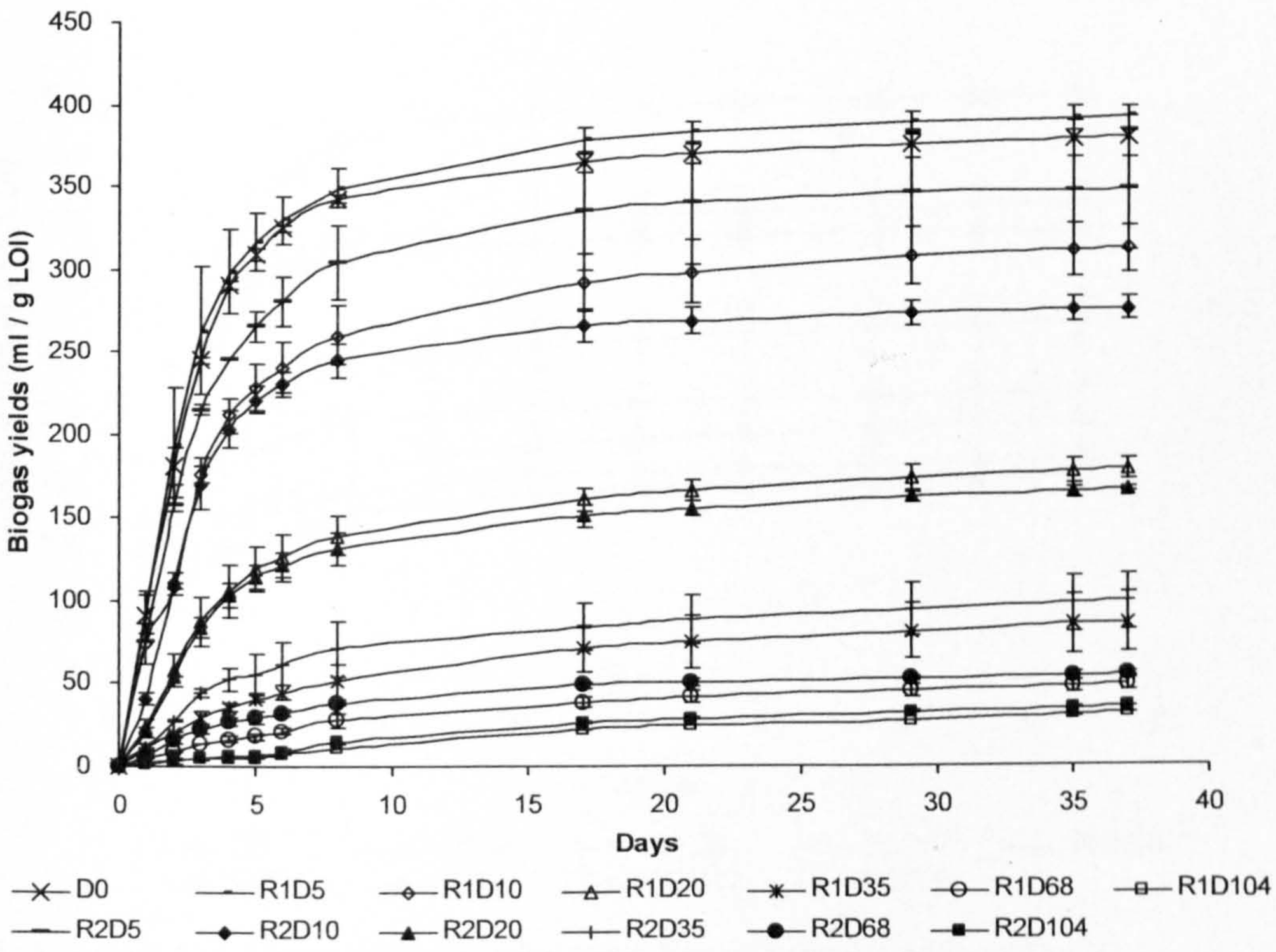


Figure 6.11a. Cumulative biogas production of samples collected in B1

(error bar represented the standard deviation of the two BMP assays for each sample tested)

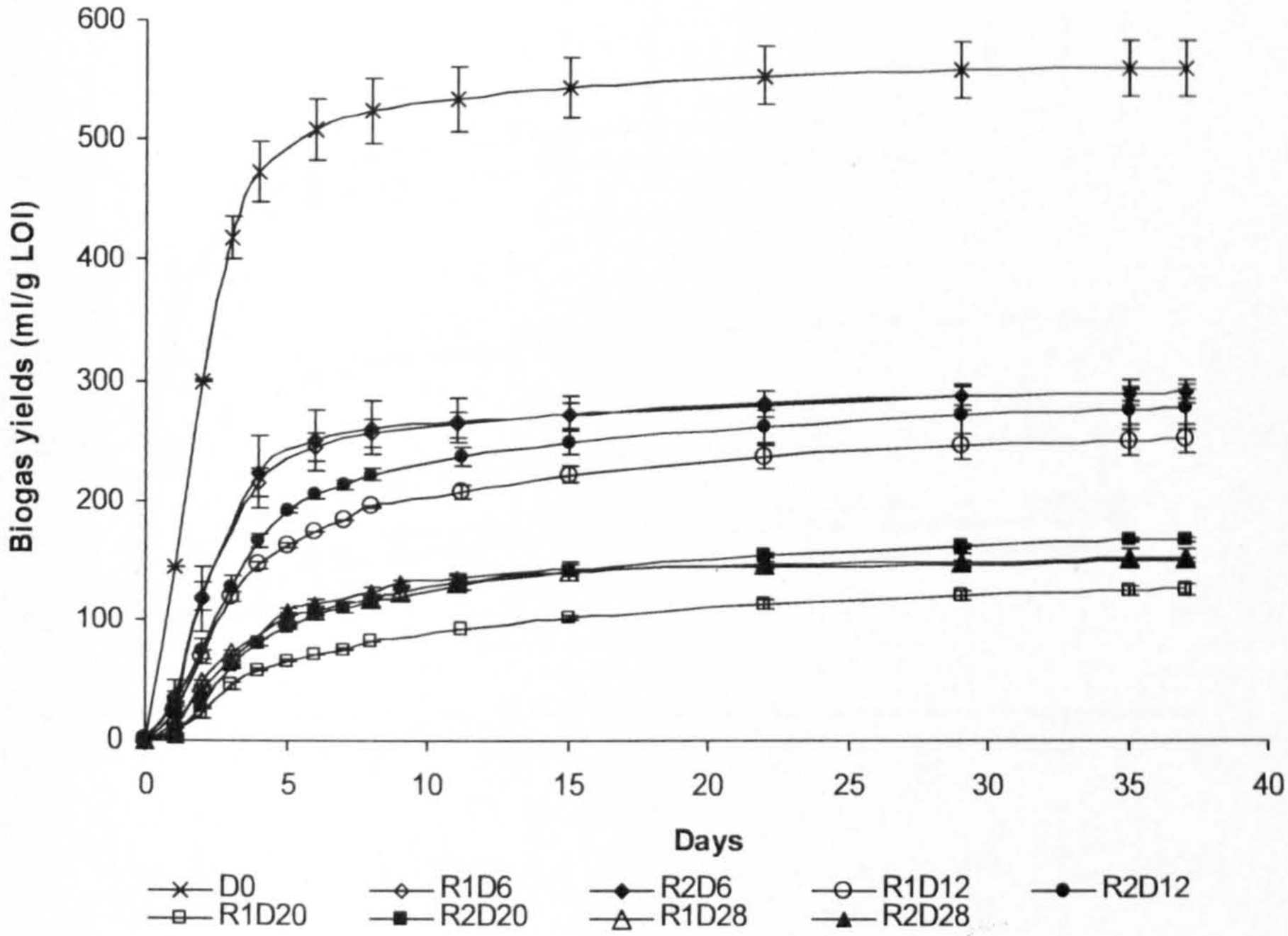


Figure 6.11b. Cumulative biogas production of samples collected in B2

(error bar represents the standard deviation of the two BMP assays for each sample tested)

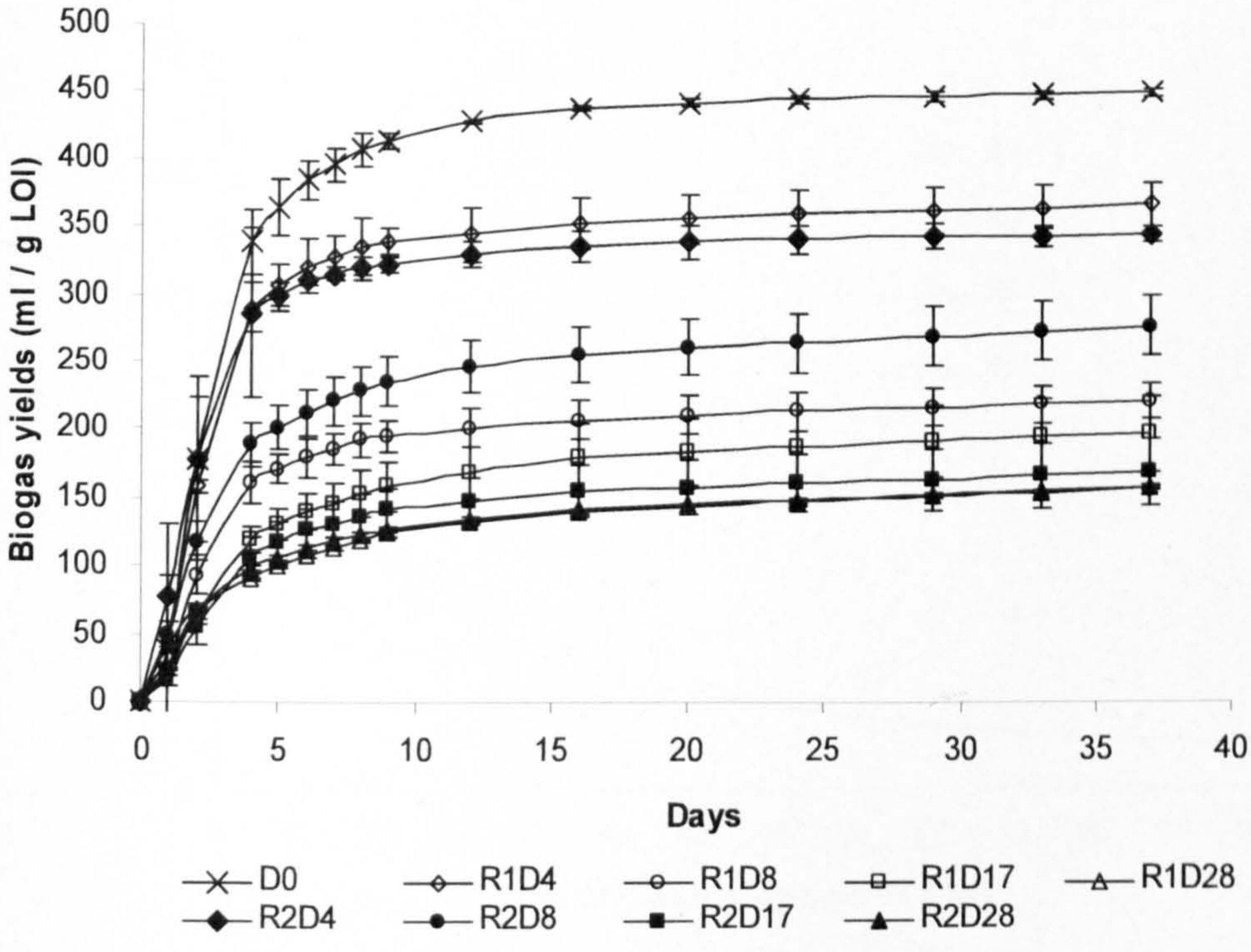


Figure 6.11c. Cumulative biogas production of samples collected in B3

(error bar represents the standard deviation of the two BMP assays for each sample tested)

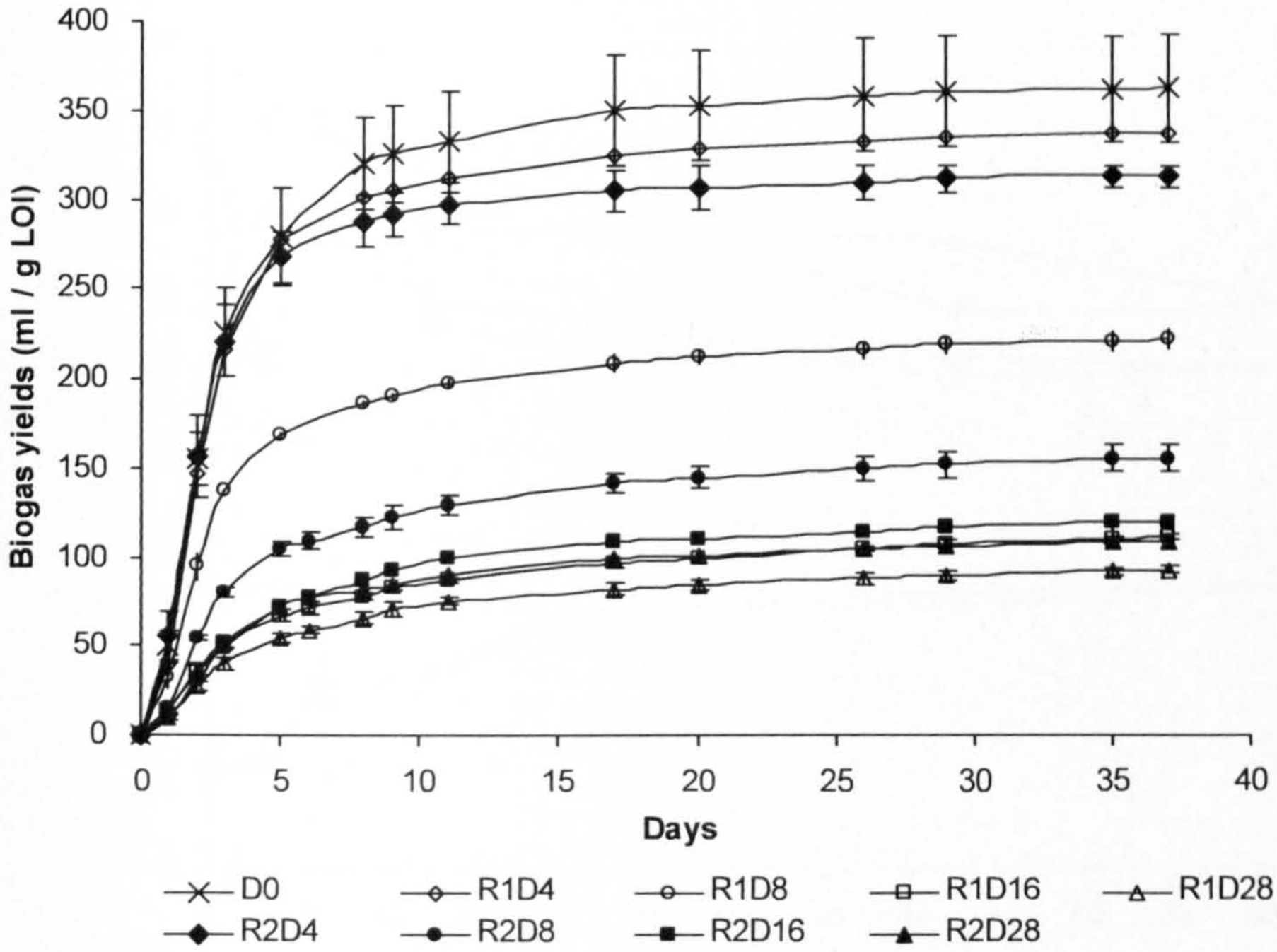


Figure 6.11d. Cumulative biogas production of samples collected in B4

(error bar represents the standard deviation of the two BMP assays for each sample tested)

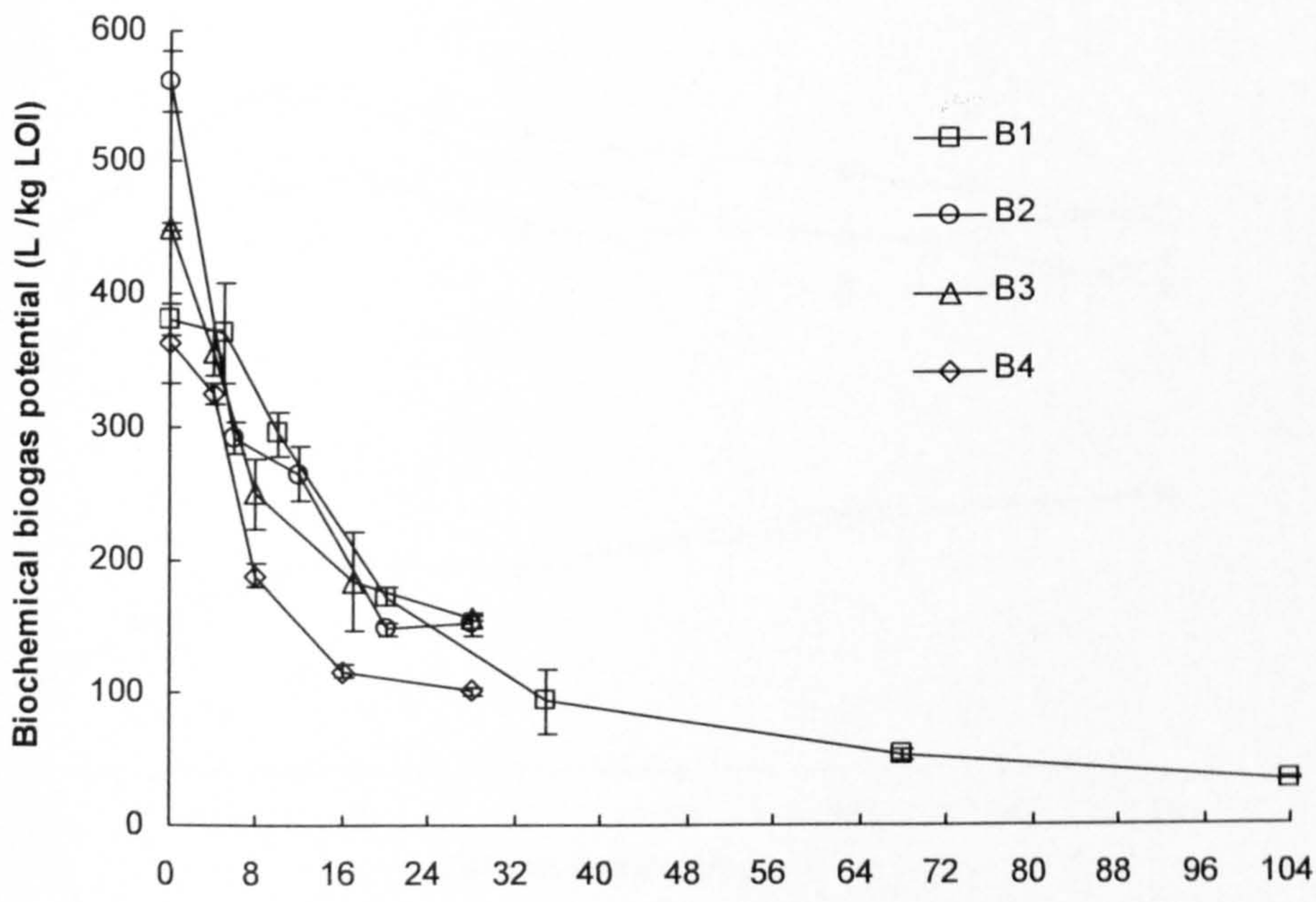


Figure 6.12. Change of biogas potential after aerobic treatment

(error bar represented the standard deviation of the mean value)

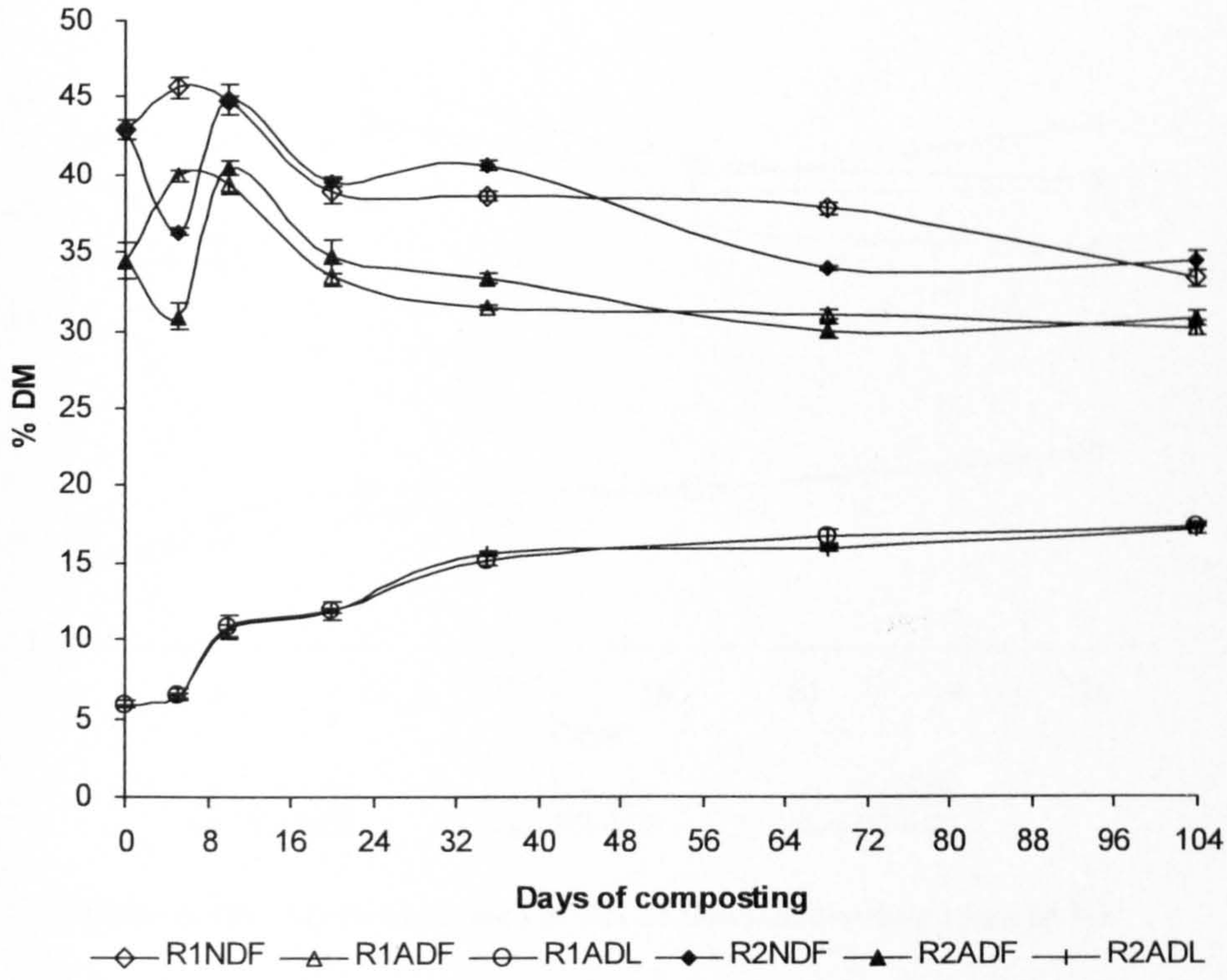


Figure 6.13a. Fibre analysis results of two composting runs in B1

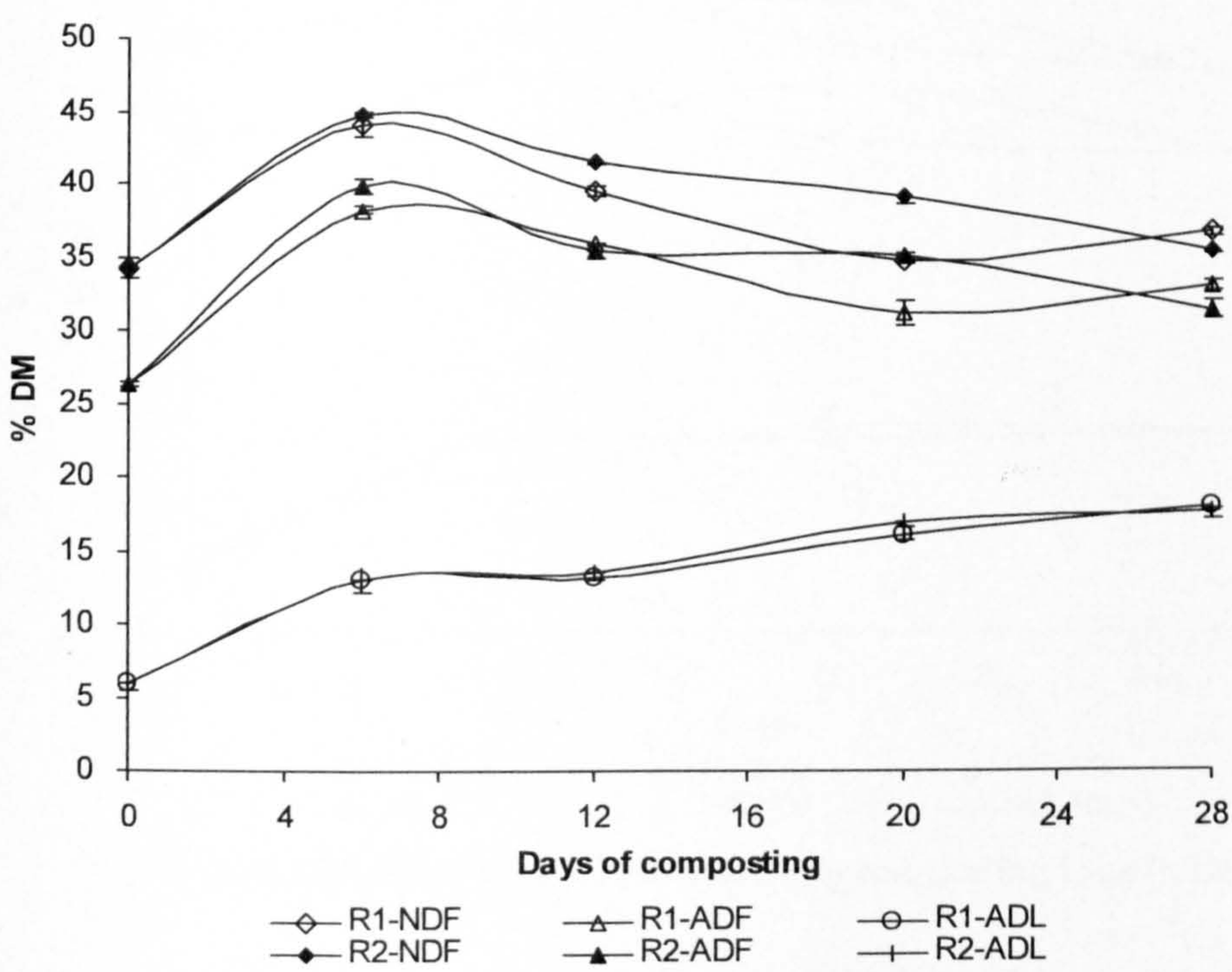


Figure 6.13b. Fibre analysis results of two composting runs in B2

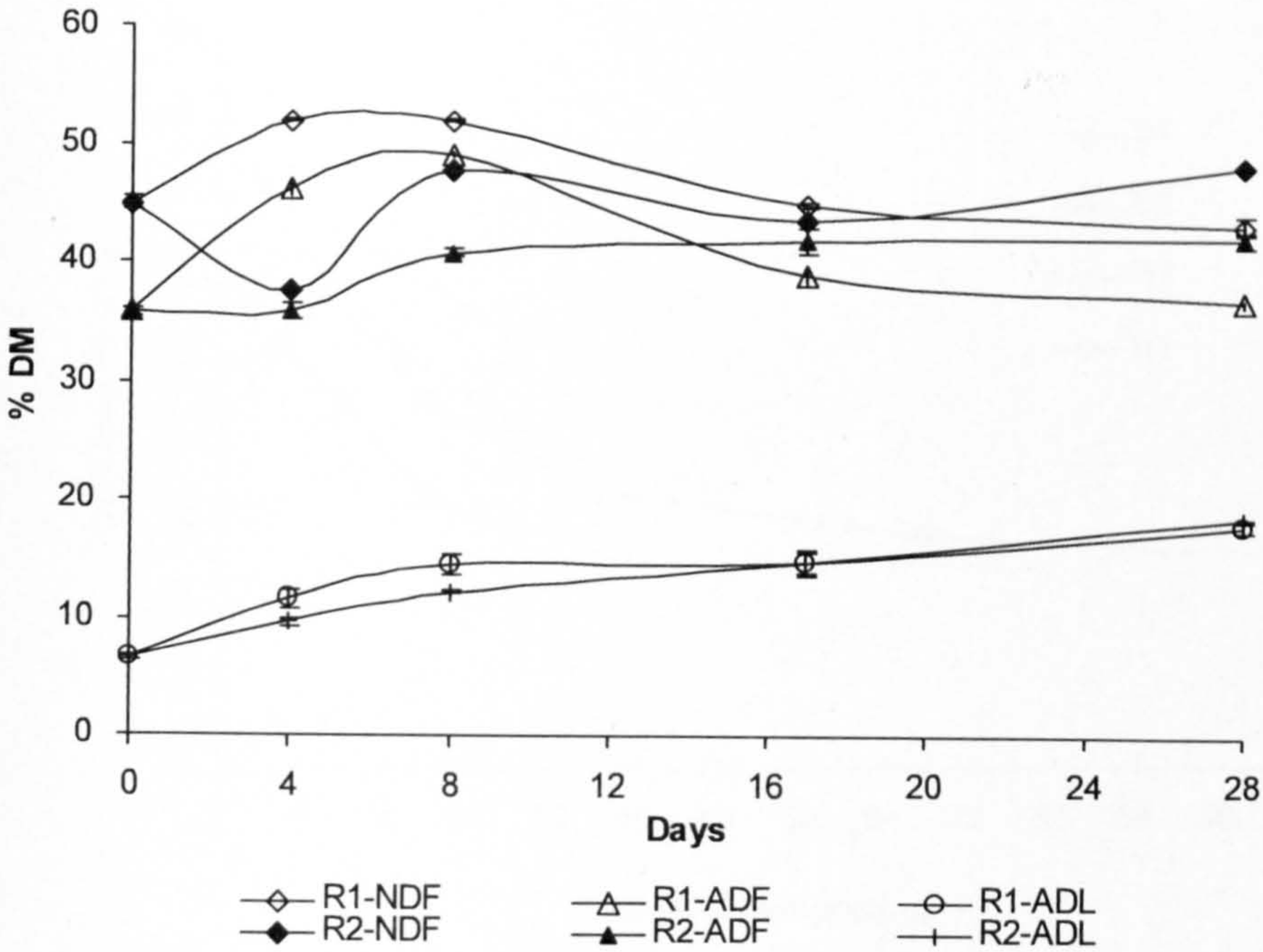


Figure 6.13c. Fibre analysis results of two composting runs in B3

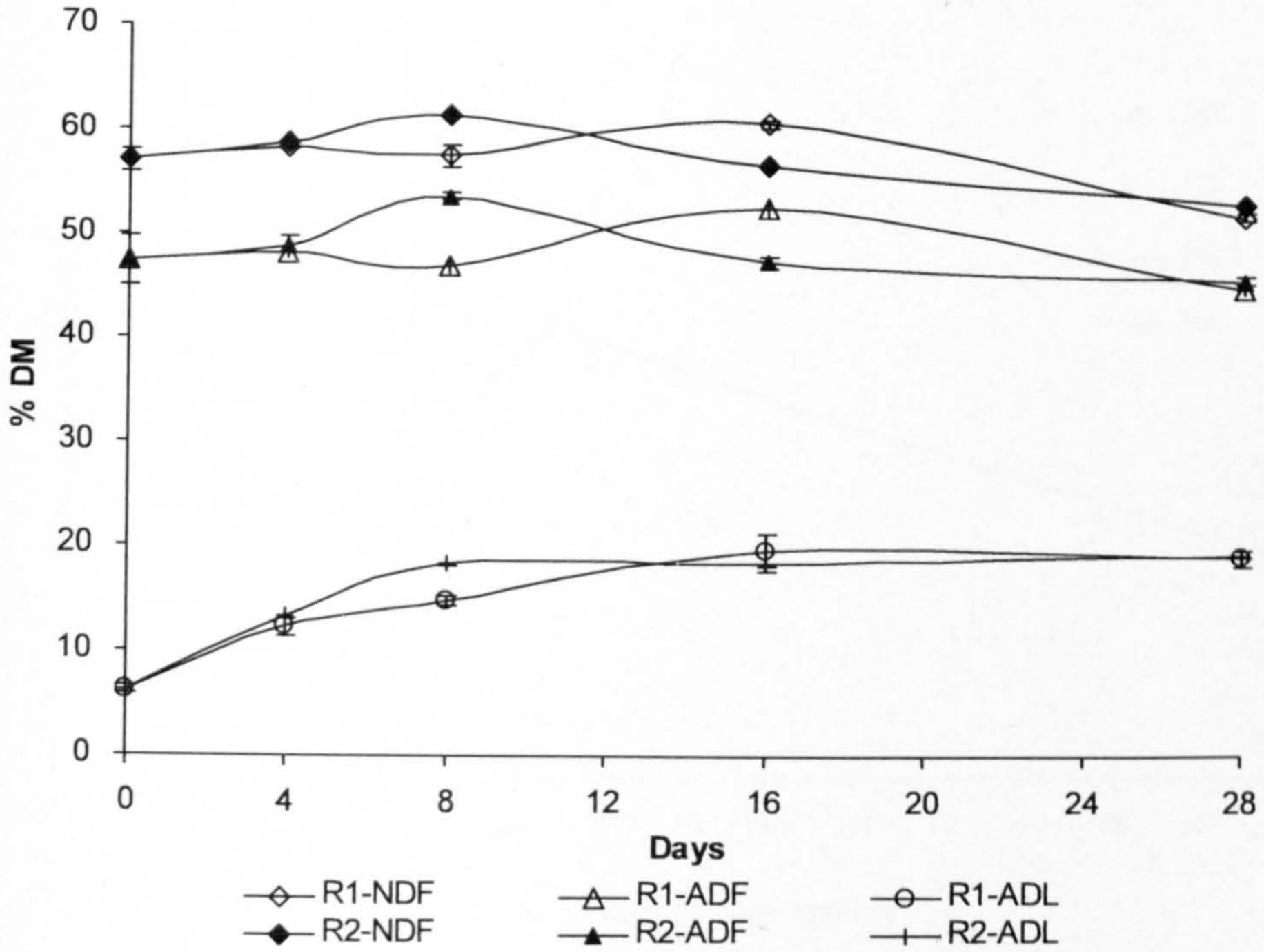


Figure 6.13d. Fibre analysis results of two composting runs in B4

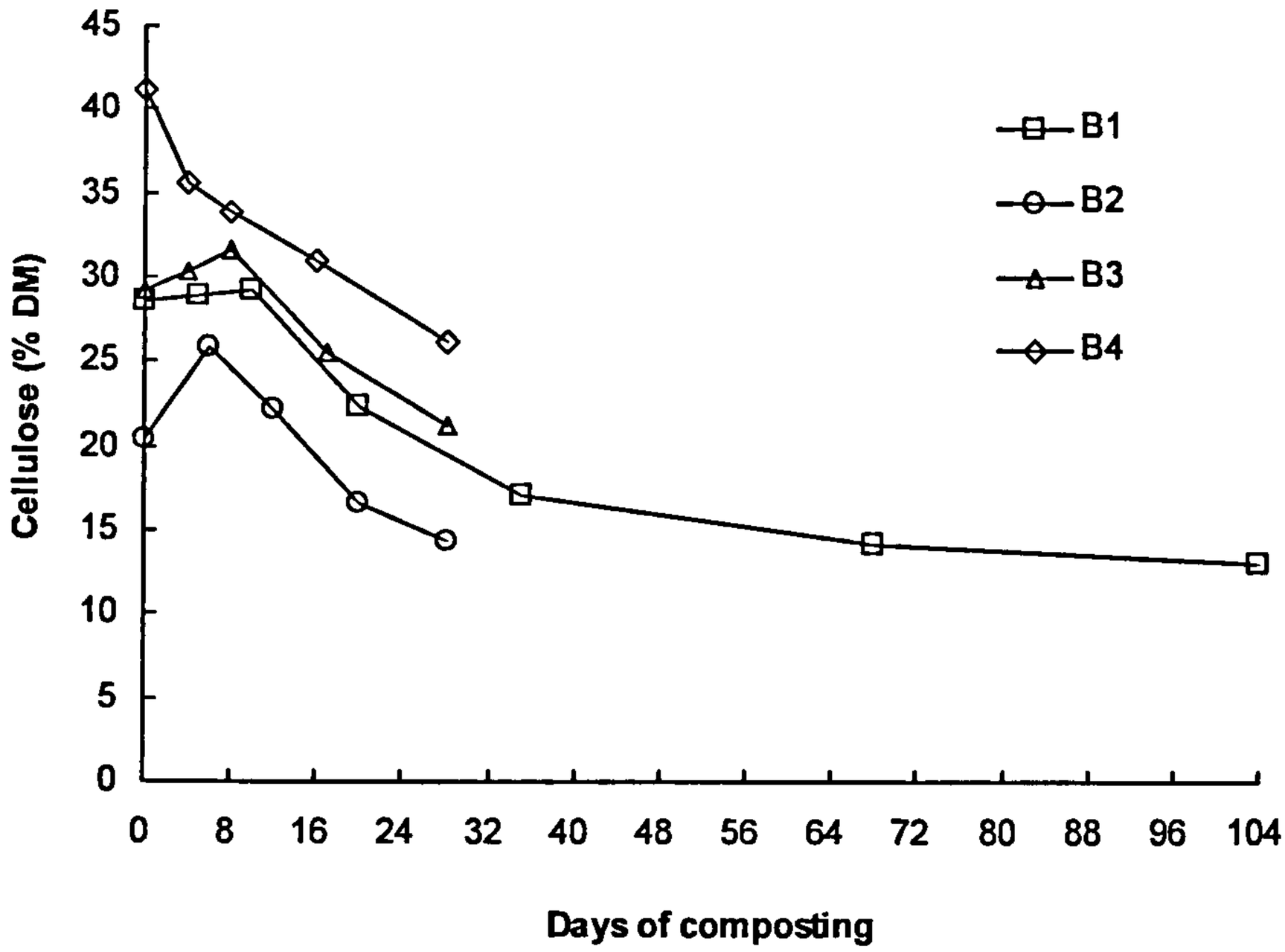


Figure 6.14. Change of cellulose content during composting in in B1-B4

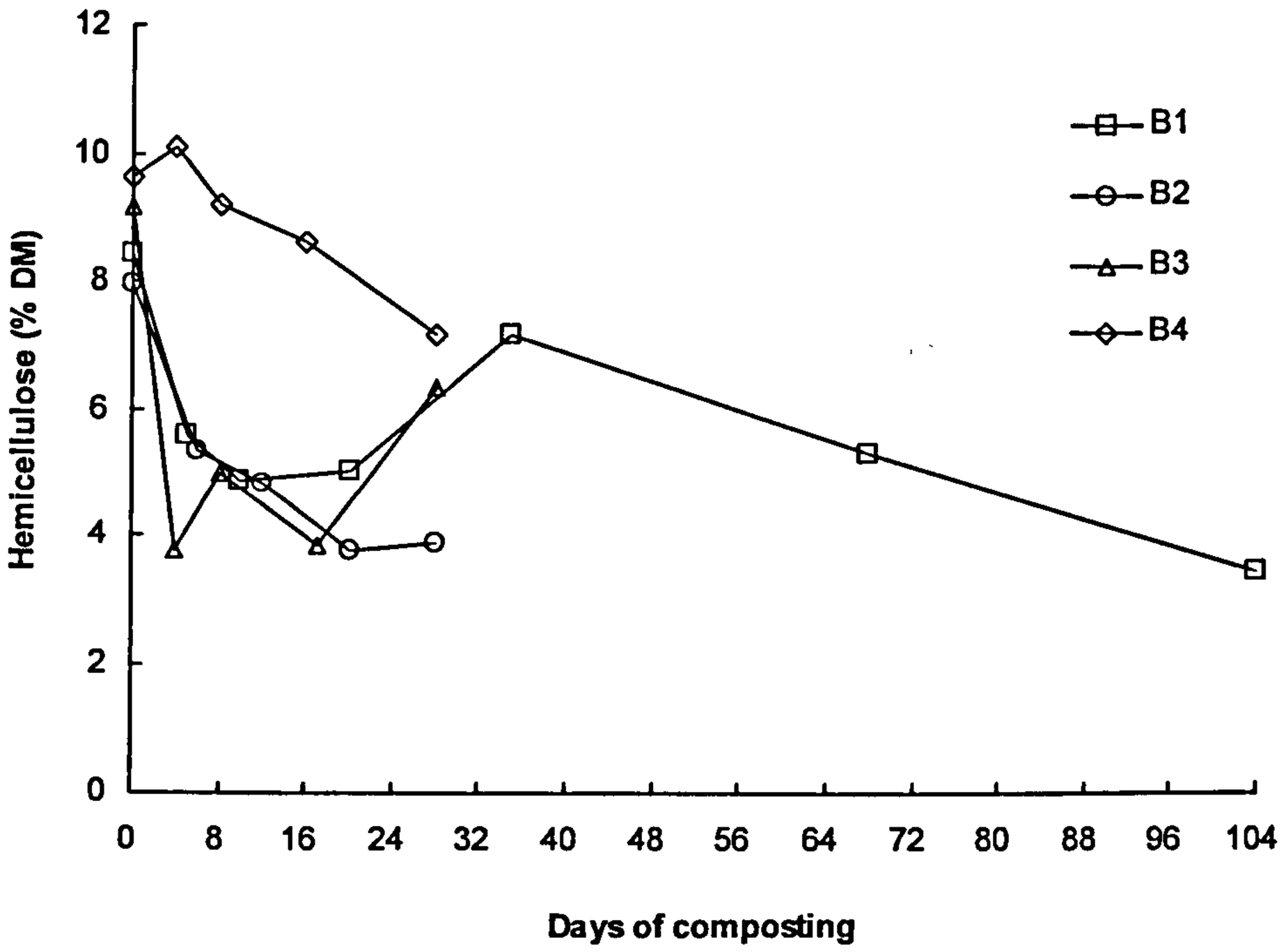


Figure 6.15. Change of hemicellulose content during composting in B1-B4

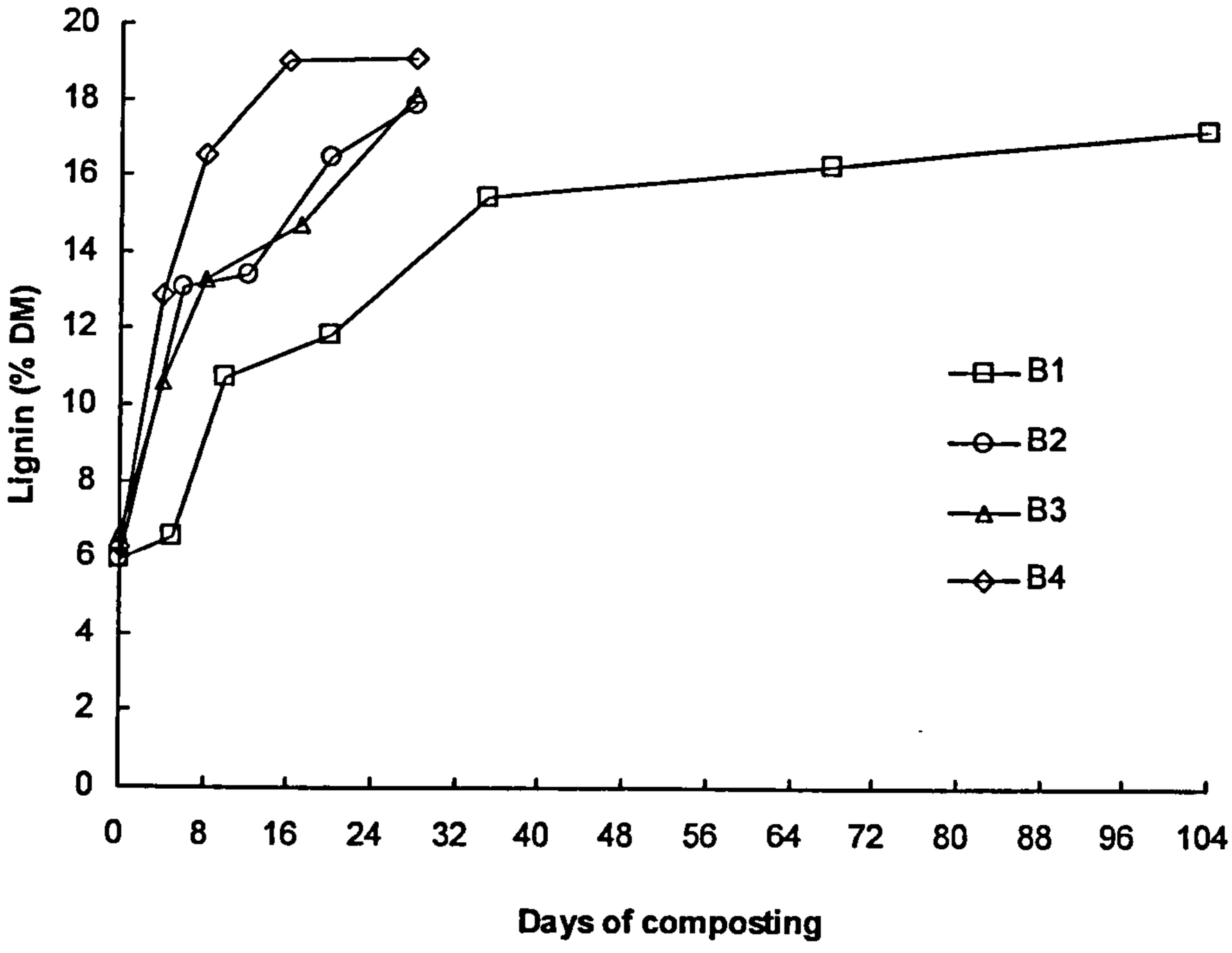


Figure 6.16. Change of lignin content during composting in B1-B4

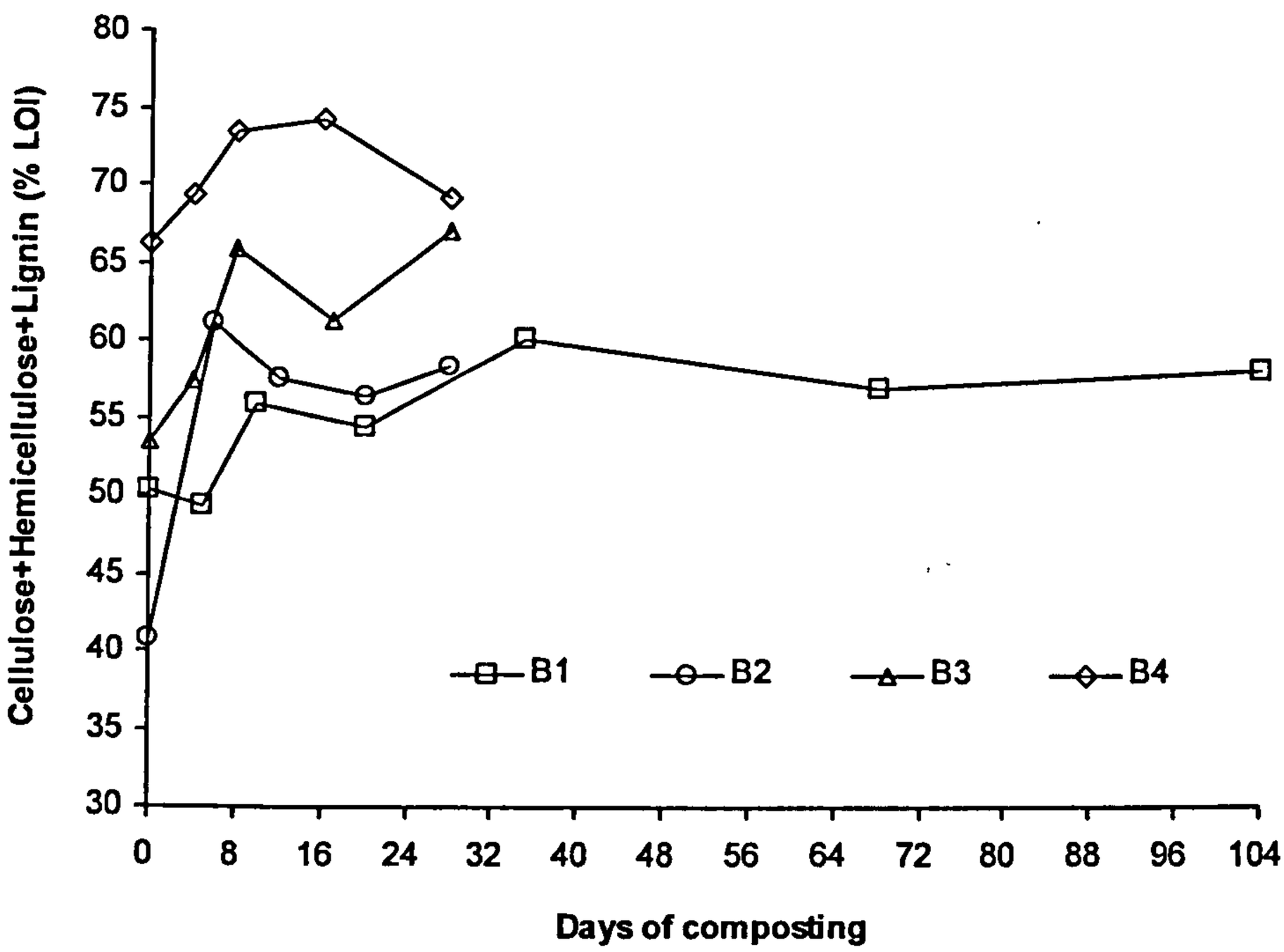


Figure 6.17. Change of NDF (cellulose, hemicellulose and lignin) content (% LOI) during composting in B1-B4

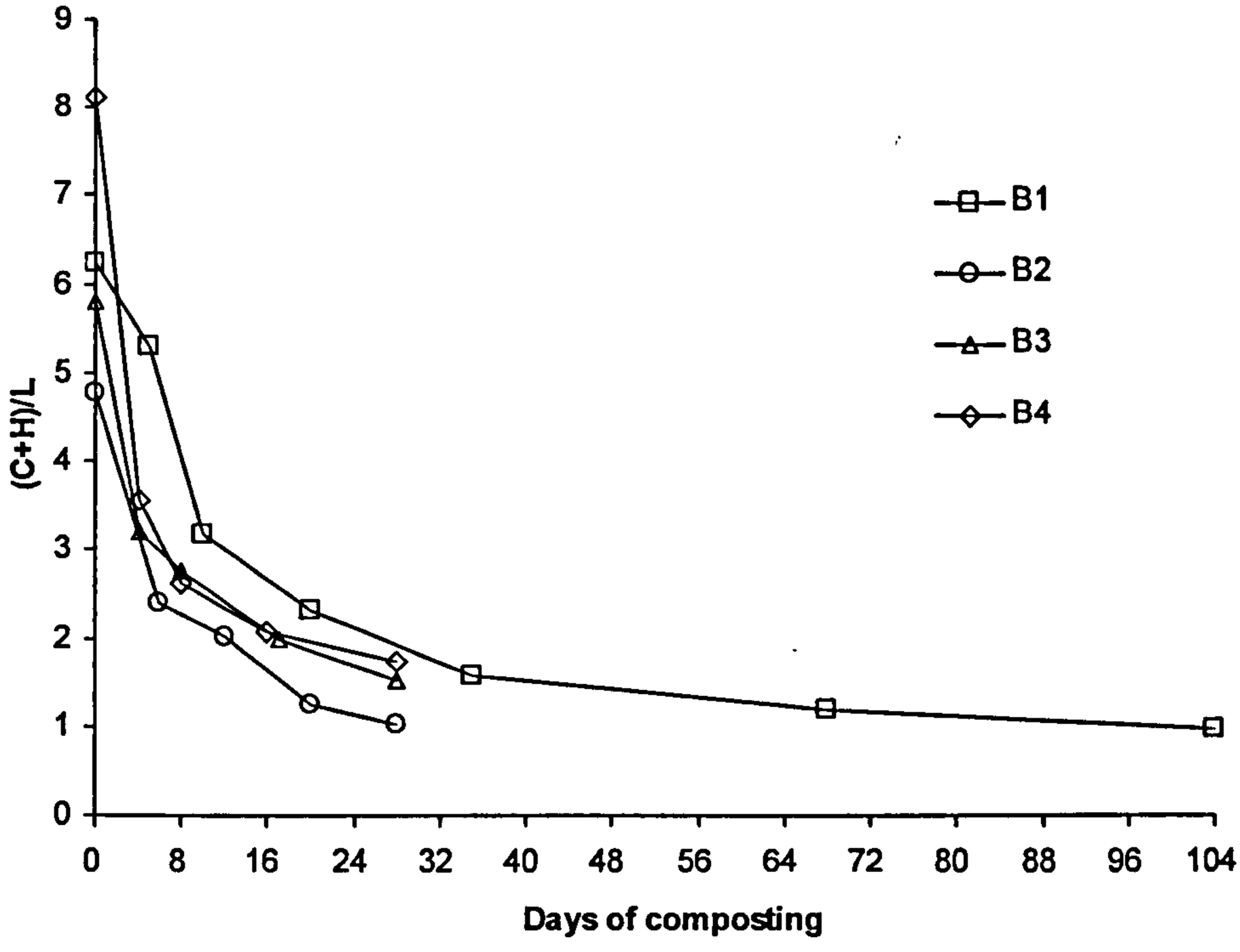


Figure 6.18. Changes of (C+H)/L ratio during composting in B1-B4

CHAPTER 7 CORRELATIONS BETWEEN TESTS

As previously described in Chapter 5, gravimetric, chemical and biological tests were conducted to investigate the degradation of BMW during composting. Chemical methods used included the measurement of the TC/TN ratio and fibre contents; the biological methods included respirometric method (DRI), and the anaerobic methane potential test (BMP). In order to identify the validity of using fibre analysis as a surrogate for assessing the biodegradability of the BMW during or following typical MBT practice, in this Chapter the results from fibre analysis are compared with those from the other tests (BMP, DRI, LOI content and TC/TN ratio) to determine if any correlations between the fibre content and the biodegradation potential exist either for each batch (B1-B4) or for both untreated waste samples and the treated waste samples. At the same time, the results of LOI content and TC/TN ratio are compared with those from BMP and DRI tests to determine if the LOI content or TC/TN ratio correlates with the biodegradation potential or if they could indicate the stability of treated waste. The test results for the untreated waste samples and the treated waste samples have already been presented in Chapter 6, and summarized in Appendix F.

7.1 Correlation between DRI and Fibre Analysis

As presented in Chapter 6, four different methods relating to the respiration index were used to assess the degree of biological stability of the BMW during degradation. These indices were DRI_{ave}, DRI_{max}, DRI4 and DRI_{tot}. In this section,

the results from the DRI tests are compared to the results of the (C+H)/L ratio determined by fibre analysis in order to identify any correlation between the two tests.

7.1.1 $DRI_{tot.}$ and (C+H)/L

Correlations between the (C+H)/L ratio and the total respiration index ($DRI_{tot.}$) of each batch composting runs are presented in Figure 7.1a. The points in the figures are the mean values of two replicate runs in each batch.

In Batch 1, when the (C+H)/L ratio decreased from 6.2 (D0) to 1.2 (D68), the oxygen consumption measured dropped from 1280.6 g O_2 /kg LOI to 127.4 g O_2 /kg LOI after 68 days of composting. During the composting processes the change of the (C+H)/L ratio correlated well with the decrease in oxygen consumption, as there was an obvious linear correlation between these two variables (r^2=0.98). The regression coefficient of the linear correlation meant that for each unit decrease of the (C+H)/L ratio, the decrease in oxygen consumption potential was 225.4 g O_2/kg LOI. The three other batches (B2, B3 and B4) also showed good linear correlations between the change of (C+H)/L ratio and the corresponding oxygen consumption during each aerobic degradation process with r^2 being 0.91 (B2), 0.90 (B3) and 0.83 (B4) (Figure 7.1a). The good correlation existing in each batch suggested that for a certain BMW studied, the change of (C+H)/L ratio during aerobic degradation could be used to evaluate the degree of degradation or the biological stability.

In order to identify if the (C+H)/L ratio correlates with the measured oxygen consumption for both the untreated and treated BMW, the results of fibre analysis for all the tested samples were compared to the corresponding oxygen consumption measured in the four batches (Figure 7.1b). Generally, the figure showed that as the (C+H)/L ratio decreased, the biodegradability in terms of oxygen consumption decreased. For those samples with the (C+H)/L ratio less than 2 the oxygen

consumption was less than 300 g O_2 /kg LOI, which is much lower than those of the untreated waste (D0, above 800 g O_2 /kg LOI), suggesting that the test material became less and less biodegradable which can be indicated by the increasing degree of lignification (lower (C+H)/L ratio). However, the (C+H)/L ratios did not correlate well with oxygen consumption for the untreated wastes (D0), for example, the untreated BMW used in B4 of which the (C+H)/L ratio was highest was not observed to show the highest oxygen consumption. Therefore, it could not be concluded for the untreated waste studied that the higher the (C+H)/L ratio, the more oxygen consumption or higher biodegradability would be expected. A better correlation between the oxygen consumption and the (C+H)/L ratio was found for the treated wastes with r^2=0.85 (Figure 7.1c) compared to the correlation for both the untreated and treated wastes with r^2=0.72 (Figure 7.1b). The failure of the regression line to pass through zero may suggest that factors in addition to cellulose and hemicellulose influence the total oxygen consumption, for example, the interference of lignin in the biodegradation of cellulose and hemicllulose (Wang, et al., 1994).

7.1.2 $DRI_{ave,}$, $DRI_{max.}$, DRI4 and (C+H)/L

The results of DRI_{ave}, DRI_{max}, DRI4 were compared to the fibre analysis in Figures 7.2, 7.3 and Figure 7.4 respectively. The points in the figures are the mean values of two replicate runs in each batch. Evidently, the correlations were poor between these three DRI expressions and the (C+H)/L ratio. No significant relationships were found between DRI4 and the (C+H)/L ratio, similar to what was observed by Godley *et al.* (2005).

As discussed in Chapter 6, the DRI_{ave}, DRI_{max}, DRI4 tests did not provide results sufficient to demonstrate biodegradability due to the limitation of the time scales and the incubation conditions, which do not allow complete decomposition of the

test material in the test period and therefore underestimate the overall biodegradability potential. To some extent, they represent more the biodegradation rate. Because $DRI_{tot.}$ was expressed as the total DRI_i from the start to the very end of the composting experiments, it was much closer to the true biodegradability of the test material than the methods of DRI_{ave}, DRI_{max}, DRI4. As described in Section 7.1.1, the (C+H)/L ratio correlated well with the total oxygen consumption for all the treated waste samples tested, and in each batch of composting runs the change of (C+H)/L ratio correlated well with the change of total oxygen consumption. This suggests that as far as the biodegradability of a test material is concerned, the fibre analysis data (such as (C+H)/L ratio) provides a quantification of the biodegradation potential or the overall biodegradability rather than the biodegradation rate.

7.2 Correlation between BMP and Fibre Analysis

7.2.1 Anaerobic Biogas Potential and Fibre Contents

The cellulose, hemicellulose and lignin contents of DM were compared with the corresponding anaerobic biogas potential respectively for the tested samples collected at various stages from each batch composting runs, as shown in Figure 7.5, 7.6 and 7.7. It was observed that the cellulose contents correlated well with the anaerobic biogas potential remaining during composting treatment in B1 ($r^2 = 0.94$) and B4 ($r^2 = 0.84$), whilst the linear correlations were poor for B2 ($r^2 = 0.18$) and B3 ($r^2 = 0.42$). Good correlations were also found between hemicellulose contents the anaerobic biogas potential remaining during composting treatment in B2 ($r^2 = 0.996$) and B4 ($r^2 = 0.71$), whilst the correlations were very poor for B1 ($r^2 = 0.22$) and B3 ($r^2 = 0.27$). Therefore, the cellulose or hemicellulose content alone during composting cannot indicate the degree of decomposition (Wang *et al.*, 1994; Eleazer, *et al.*, 1997). Good correlations were found between the lignin content and the anaerobic biogas potential remaining during composting treatment in each batch

with r^2 being 0.97 (B1), 0.98 (B2), 0.95 (B3) and 0.89 (B4). This suggests that the higher the lignin content, the more stable the waste would be, as observed by Komilis and Ham (2003). Therefore, the lignin content is a promising indicator of the degree of decomposition for BMW.

7.2.2 Anaerobic Biogas Potential and (C+H)/L Ratio

The comparison between (C+H)/L and anaerobic biogas potential of the test samples collected from each batch of composting runs are presented in Figure 7.8a. The points in the figure are the mean values of two replicate runs in each batch. It was clear that as the (C+H)/L decreased through the composting process, the anaerobic biogas potential also decreased.

In Batch 1, as the (C+H)/L ratio decreased from 6.2 to less than 2.0 after 35 days of composting, the corresponding anaerobic biogas potential dropped from 381 L/kg LOI to below 93 L/kg LOI. When the composting process of Batch 1 was terminated after 104 days, the (C+H)/L ratio of the compost was 1.0 and the anaerobic biogas yield remaining was 34 L/kg LOI. Additionally, a good linear relationship (r^2=0.92) was found between the (C+H)/L ratio and the corresponding anaerobic biogas potential remaining during composting. According to the linear regression, each unit decrease of the (C+H)/L ratio indicated that the BMW was removed by 68.89 L/kg LOI in terms of anaerobic biogas potential during aerobic treatment. In the other batches (B2, B3 and B4), good linear correlations also existed between the (C+H)/L ratio and the corresponding oxygen consumption during each aerobic degradation process with r^2 being 0.99 (B2), 0.90 (B3) and 0.71 (B4) (Figure 7.8a). The good correlation existing in each batch suggested that for a certain BMW studied, during aerobic degradation the degree of decomposition or the change of biodegradability can be evaluated by the change of (C+H)/L.

In order to identify if (C+H)/L ratios correlate with the anaerobic biogas potential for both the untreated and treated waste, the results of fibre analysis from the four batches were compared to the corresponding biogas potential (Figure 7.8b). Generally, there was a trend of higher biogas potential in the less lignified test material (higher (C+H)/L ratio) (r^2=0.63). However, this relationship was not observed amongst the four untreated wastes (D0). For example, B4D0 with highest (C+H)/L ratio (8.1) showed the lowest biogas potential (362.7 L/kg LOI), and B2D0 with the lowest (C+H)/L ratio (4.8) showed the highest biogas potential (561.1 L/kg LOI). This could be explained by examining the lignocellulose contents in the untreated waste samples. The lignin contents of B1D0 - B4 D0 were close to each other ranging between 5.9-6.6%DM, and the highest (C+H)/L ratio of B4D0 was the consequence of its highest cellulose plus hemicellulose content (50.8%DM). The reason for the relatively low biodegradability of B4D0 may be that the presence of lignin can interfere or hinder the degradation of cellulose and hemicellulose (Micales and Skog, 1997; Stinson and Ham, 1995), although the cellulose and hemicellulose are biodegradable constituents. Therefore, for the untreated waste, it cannot be concluded that the biogas potential correlate directly with the (C+H)/L ratio.

A better correlation between the anaerobic biogas potential and the (C+H)/L ratio was found for the treated wastes with r^2=0.71 (Figure 7.8c) compared to the correlation for both the untreated and treated wastes, which indicates that for the treated waste the more lignified or the lower (C+H)/L ratio, the less anaerobic biogas potential. Those samples with the (C+H)/L ratio less than 2 were found with the anaerobic biogas potential less than 200 L/kg LOI, which is much lower than those of the untreated waste (above 360 L/kg LOI). Therefore, for the treated BMW studied, the (C+H)/L ratio can be used to evaluate the extent of decomposition. Again, the failure of the regression line to pass through zero may suggest that factors apart from cellulose and hemicellulose influence the anaerobic biogas potential.

7.2.3 Anaerobic Biogas Potential and C/L Ratio

The cellulose to lignin ratio (C/L) had been used to evaluate the stability of compost and the waste in landfills in some studies, for example, Bookter and Ham (1982) and Komilis and Ham (2003).

The comparison of the change of C/L ratio and the anaerobic biogas potential in each batch of composting treatment (Figure 7.9a) showed good correlations between the change of C/L ratio and the biogas potential for each batch. For both untreated and treated waste of all four batches (Figure 7.9b), there was also a general trend of higher anaerobic biogas potential in the test material with a higher C/L ratio except for points associated with the untreated waste, which was similar to those points in Figure 7.8b. A better correlation between the C/L ratio was found for the treated wastes with r^2=0.75 (Figure 7.9c) compared to the correlation for both the untreated and treated wastes with r^2=0.62 (Figure 7.9b).

Compared to Figures 7.8a – 7.8c, it was evident that the relationships between C/L ratio and the biogas potential were similar to the relationship between (C+H)/L ratio and the biogas potential. Therefore, in common with the (C+H)/L ratio, during the degradation process of BMW the change of C / L ratio can also be a useful indicator for the anaerobic biogas potential. Additionally, no correlation was found between the hemicellulose to lignin ratio (H/L ratio) and the anaerobic biogas potential for treated waste in all four batches (r^2=0.34, plots not shown). This suggests that the degradation of cellulose could play more important role for the test material in the change of the biodegradability than the degradation of hemicellulose.

7.3 Correlation between BMP and DRI

A comparison of anaerobic BMP method and aerobic DRI method is illustrated in Figure 7.10a, where the expression of total oxygen consumption ($DRI_{tot.}$) was used. It shows that the change of anaerobic biogas potential correlated well with the total oxygen consumption of the waste samples collected in each batch. In order to make the comparison for both untreated and treated waste samples of all four batches, the total duration used to calculate $DRI_{tot.}$ in B1 was normalized to be 28 days for consistency with B2, B3 and B4. Calculation and results are given in Appendix B. There was a significant linear correlation between the aerobic and anaerobic biodegradation tests (Figure 7.10b, r^2=0.83). In general, the higher the oxygen consumption of the test material, the more biogas potential it had. The failure of the regression line to pass through zero may suggest that, after aerobic treatment cellulose and hemicellulose becomes more readily available for methanogenic conversion in anaerobic conditions (Latham, 1979; Komilis *et al.*, 1999).

The methods of DRI_{ave}, DRI_{max}, DRI4 were also compared to the BMP method (the plots are not shown). Significant relationships were also found for both untreated and treated waste samples from all four batches with decreased correlation coefficients (r^2=0.61, 0.59 and 0.64, respectively) compared to that between BMP and $DRI_{tot.}$(r^2=0.83). This further confirms that $DRI_{tot.}$ is closer to the true overall biodegradability of the test material than the methods of DRI_{ave}, DRI_{max}, DRI4 because the time scale of $DRI_{tot.}$ allowed longer for more complete degradation. The correlation between DRI4 and BMP is similar to that found by Godley *et al.* (2007a) which was developed from the results of 96 treated and untreated BMW samples and specific waste components.

7.4 Correlations between LOI and Biological Tests

As reviewed in Section 3.3.1, LOI content has been used as a general criterion to characterize the total organic solid in solid wastes. In order to identify if it could be a suitable criterion to evaluate biodegradability of BMW, the LOI contents of the waste samples are compared with the results from BMP and DRI either for each batch or for both untreated and treated waste from all four batches.

The LOI content of the samples in each batch correlated well with the anaerobic biogas potential and oxygen consumption measured ($r^2 > 0.91$, Figure 7.11a and Figure 12). For both untreated waste and treated waste, relatively poor relationships were observed between LOI contents and anaerobic biogas potential ($r^2 = 0.61$, Figure 7.11b), and oxygen consumption ($r^2 = 0.59$, plot not shown) The biogas potential of the untreated waste used in the four batches followed a decreasing sequence of B2D0, B3D0, B1D0 and B4D0, of which the LOI contents were 83.9%, 84.2%, 85.3% and 86.1%, respectively. This confirmed that although LOI represents the organic solids in the waste tested it does not provide a suitable indicator of biodegradation potential as it also includes organic parts that do not degrade. The relationship between LOI contents and anaerobic biogas potential and oxygen consumption for the treated waste was even poorer ($r^2 = 0.5$ and $r^2 = 0.43$, plots not shown). Therefore, although during composting treatment of one certain BMW, the change of LOI contents indicates the degree of BMW degradation, the LOI content alone cannot be used to indicate the biodegradation potential for either untreated waste or treated waste.

7.5 Correlations between TC/TN Ratio and Biological Tests

The TC/TN ratio was compared with the results from BMP and DRI tests either for each batch or for both untreated and treated waste from all four batches. The

TC/TN ratios of the samples in each batch correlated well with their anaerobic biogas potential, and oxygen consumption (r^2 = 0.63 - 0.99, Figure 7.13a and 7.14a). The relationships between TC/TN ratio and anaerobic biogas potential (Figure 7.13b) for both untreated waste and treated waste were relatively poor (r^2 =0.58), whilst a good correlation existed between TC/TN ratio and oxygen consumption (r^2 =0.83, Figure 7.14b). The correlations for the treated waste were good (r^2 =0.72, Figure 7.13c; r^2 =0.80, plot now shown). This suggested that TC/TN ratio was a good indicator of aerobic degradation potential as far as BMW was concerned.

7.6 Discussion and Summary

In the comparisons above, good relationships existed among the tests evaluated. More specifically, during the process of aerobic degradation in each batch, the change of the (C+H)/L ratio correlated well with the results from other biological or chemical tests in each batch, including degradable carbon content, oxygen consumption, anaerobic biogas potential, TC/TN ratio and LOI content. Therefore, for certain BMW studied the change of (C+H)/L ratio is capable of evaluating the change of biodegradability or the degree of decomposition.

For the untreated wastes used in the four batches (B1D0-B4D0), it was not found that the higher the cellulose plus hemicellulose content, the more biogas potential, which is consistent with the results of Eleazer *et al.* (1997) and Barlaz *et al.* (1997). Additionally, amongst the untreated wastes it was not the case that the higher (C+H)/L ratio, the higher oxygen consumption or anaerobic biogas potential. One reason may be that the presence of lignin interferes or hinders the degradation of cellulose and hemicellulose (Micales and Skog, 1997; Stinson and Ham, 1995), although the cellulose and hemicellulose are biodegradable constituents. As reviewed in Section 3.2.2, to what extent this interference or impedance would be differs from material to material and is still unknown yet.

The non-fibre organic parts have been defined as the cell soluble matter by Chandler *et al.* (1980), which includes the organic matter soluble in neutral detergent. By comparing the cell soluble contents of the untreated waste with their corresponding biodegradable carbon contents (Figure 7.15) and anaerobic biogas potential (Figure 7.16), there was a general trend that the higher the cell soluble contents, the higher the biodegradable carbon contents or anaerobic biogas potential. This may suggest that in the untreated waste, the non-fibre parts also play an important role in waste biodegradability, which is consistent with what has been reviewed in Section 3.1.2, i.e., in BMW the cellulose, hemicellulose and lignin are less readily biodegradable compounds compared to other organic constituents such as sugars, starch.

Additionally, when comparing the (C+H)/L ratios with oxygen consumption or anaerobic biogas potential for both day 0 (untreated) and other times (treated) waste, the derived correlations in most cases showed a high degree of discrepancy at day 0 compared to the other times. This is almost certainly due to some adaptation of the waste occurring both chemically and biologically, particularly with respect to the development of a microbial population and with associated extracellular hydrolytic or enzyme production activity within the biomass. It is to be expected that a steady state would not be attained during the initial phases of the composting or any other biological degradation process. The associated correlations therefore showed good correlation coefficient for the waste after some time periods of biological treatment (such as after day 4, 5 or 6) for most the different tests evaluated (Table 7.1). Therefore, as far as untreated BMW is concerned, it would be more difficult to establish precise correlations between the chemical parameters (such as (C+H)/L ratio) and their biodegradability.

Table 7.1. Summary of linear correlation coefficients (r^2) for the tests on treated waste samples

	(C+H)/L[a]	C/L[a]	BMP[a]	$DRI_{tot.}$[b]	LOI[a]	TC/TN[a]
(C+H)/L	1	--	0.71	0.85	--	0.87
C/L		1	0.75	0.79	--	--
BMP			1	0.86	0.50	0.72
$DRI_{tot.}$				1	0.43	0.80
LOI					1	--
TC/TN						1

[a]18 data points (36 waste samples); [b]14 data points (28 waste samples)

For the BMW treated by MBT, the (C+H)/L ratio correlated with oxygen consumption or anaerobic biogas production. Therefore, the fibre analysis is capable to demonstrate the change of biodegradability during MBT by directly measuring the change of lignocellulosic content in the target material. Based on the correlation analysis as shown in Figures 7.1c, 7.8c and 7.9c, three linear model equations were suggested to predict the biodegradability upon fibre analysis for the BMW during or after MBT process (Table 7.2). For example, according to Model 1, for the BMW samples treated by MBT of which the (C+H)/L ratio is 1, the anaerobic biogas potential will be in the range of 69.4-123.2 L/kg LOI with 95% confidence level. Furthermore, if the (C+H)/L ratio of BMW decreases by 1 unit during the pretreatment of MSW in MBT processes prior to landfill, the corresponding amount of BMW diverted from landfill estimated as the reduction in anaerobic biogas potential will be in the range of 53.0-106.8 L/kg LOI with 95% confidence level.

Because the limit value of BMP on the stability of treated MSW was expressed in terms of DM instead of LOI (Section 3.4.2), Model 4 and 5 are given in order to predict the limit value of (C+H)/L ratio, the anaerobic biogas potential in which is in terms of DM. According to Model 4, the limit values of (C+H)/L ratio are in the range of 0.4-0.6 (C/L ratio: 0.3-0.4, based on Model 5), which correspond to the BMP of 20 L/kg DM and is acceptable for landfilling. The limit value of C/L ratio

predicted is quite close to that suggested by Komilis and Ham (2003) for indicating maturity for most of MSW substrates (C/L ratio < 0.5).

Using the results of cellulose, hemicellulose and lignin contents analyzed by Godley *et al.* (2007b), anaerobic biogas potentials can be predicted by Model 1 and Model 3 for the five BMW samples after MBT in their study. The results of prediction is presented in Table 7.3 and compared with their data of the measured BM100. As the models are based on the lab experimental data, before fully relied on for the purpose of monitoring BMW diversion from landfills, they need further validation by testing a variety of waste samples from operations of MBT plants.

Table 7.2. Linear models for predicting biodegradability for treated waste

No.	Model	r^2	Significance F	Confidence intervals[a]
1	$B= 79.9X_1+ 16.39$	0.71	1.1E-05	(53.02-106.82)
2	$O= 238.55X_1- 217.39$	0.85	2.4E-06	(176.34-300.77)
3	$B=94.52X_2+ 22.66$	0.75	3.4E-06	(65.59-123.44)
4	$B^b= 71.41X_1-13.10$	0.80	5.2E-07	(52.60-90.20)
5	$B^b= 80.70X_2- 6.10$	0.83	1.6E-07	(63.53-103.90)

X_1: (C+H)/L ratio

X_2: C/L ratio

B: Anaerobic biogas potential, L/kg LOI.

O: Oxygen consumption, g O_2/kg LOI.

[a] Confidence intervals of regression coefficients at 95% confidence level.

[b] The unit is L/kg DM.

Table 7.3. Comparison of predicted biogas potentials and measured BM100 using data from Godley *et al.* (2007b)

Waste tested	BM100[a] (L/kg LOI)	(C+H)/L[a]	Predicted BMP[b] (L/kg LOI)	C/L[a]	Predicted BMP[c] (L/kg LOI)
BMW (A) (fully composted)	105.0	2.2	195.5	0.8	101.9
BMW (B) (14 days composted)	293.0	1.9	165.9	1.2	138.4
BMW (B) (composted, screened<8mm)	314.0	1.8	160.2	1.0	113.9
Kitchen/green waste (fully composted)	99.0	1.5	136.2	1.0	117.2
Green waste (partially composted, screened)	221.0	1.5	136.2	0.9	103.5

BMW(A) and BMW(B): two different BMW

[a] test results from Godley *et al.* (2007b).

[b] predicted biogas potential by (C+H)/L ratio, using Model 1 in Table 7.2.

[c] predicted biogas potential by C/L ratio, using Model 3 in Table 7.2

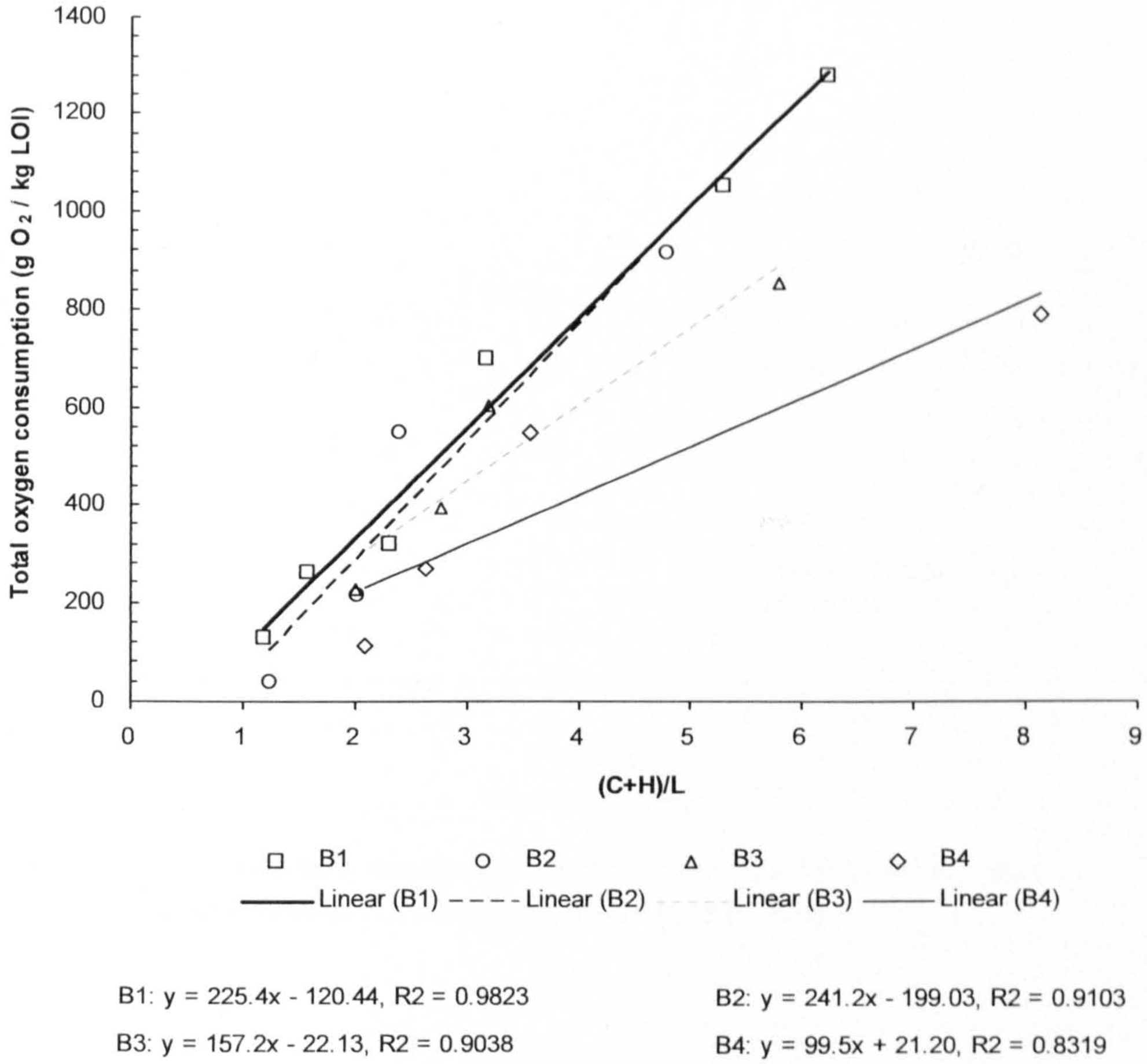


Figure 7.1a Correlation between (C+H)/L ratio and $DRI_{tot.}$ in each batch of composting treatment

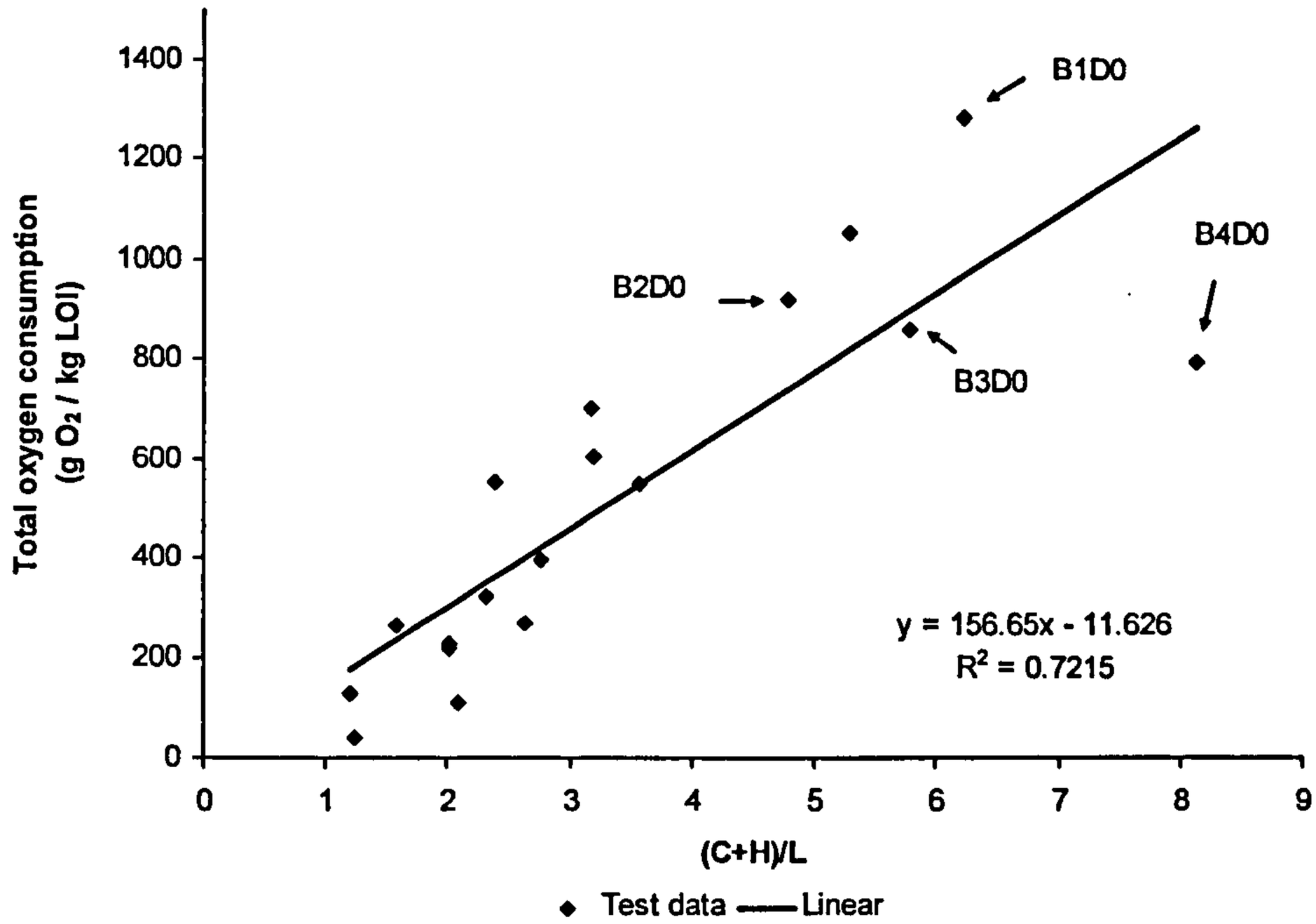


Figure 7.1b Correlation between (C+H)/L ratio and $DRI_{tot.}$ for untreated and treated wastes (Combination of the four batches)

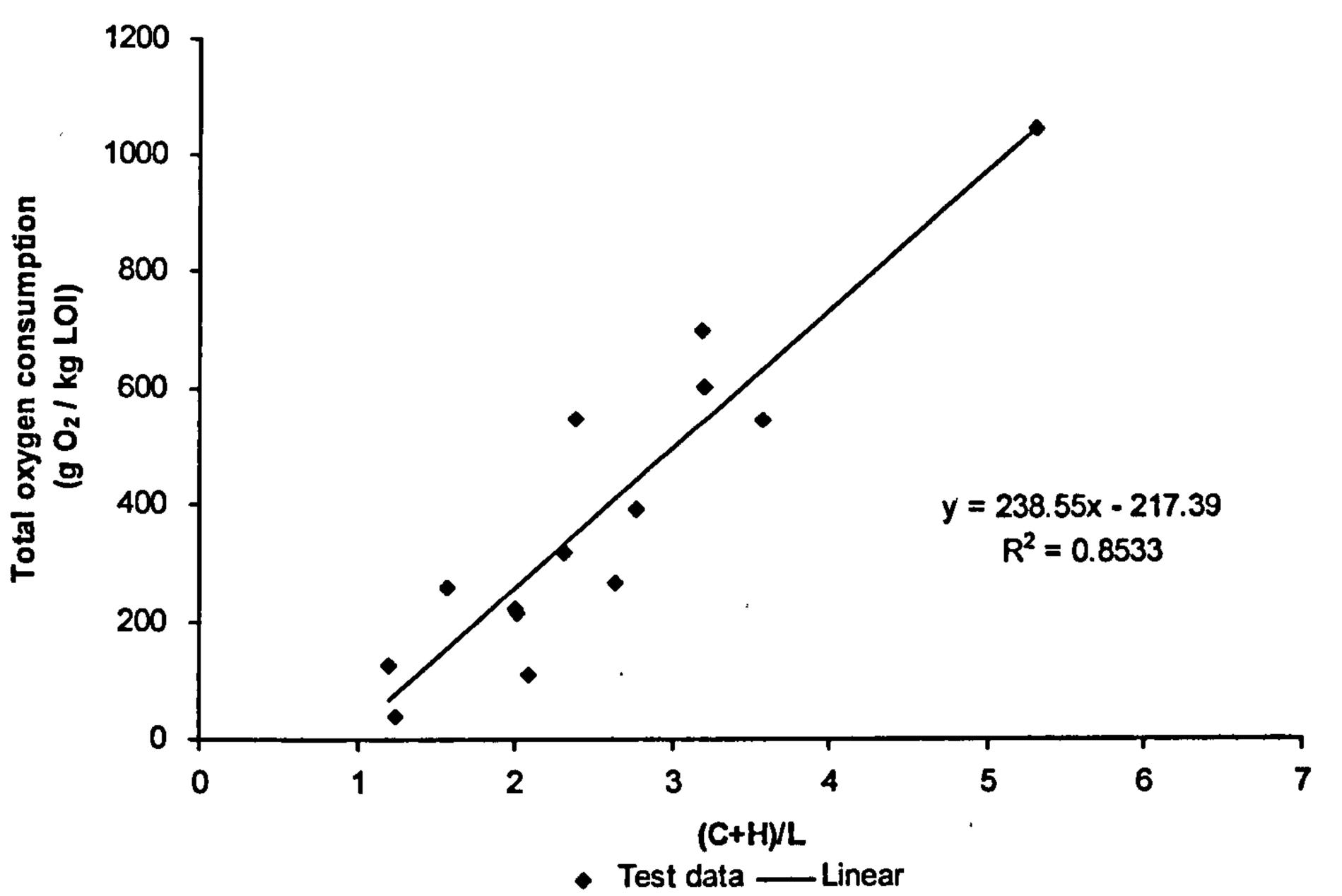
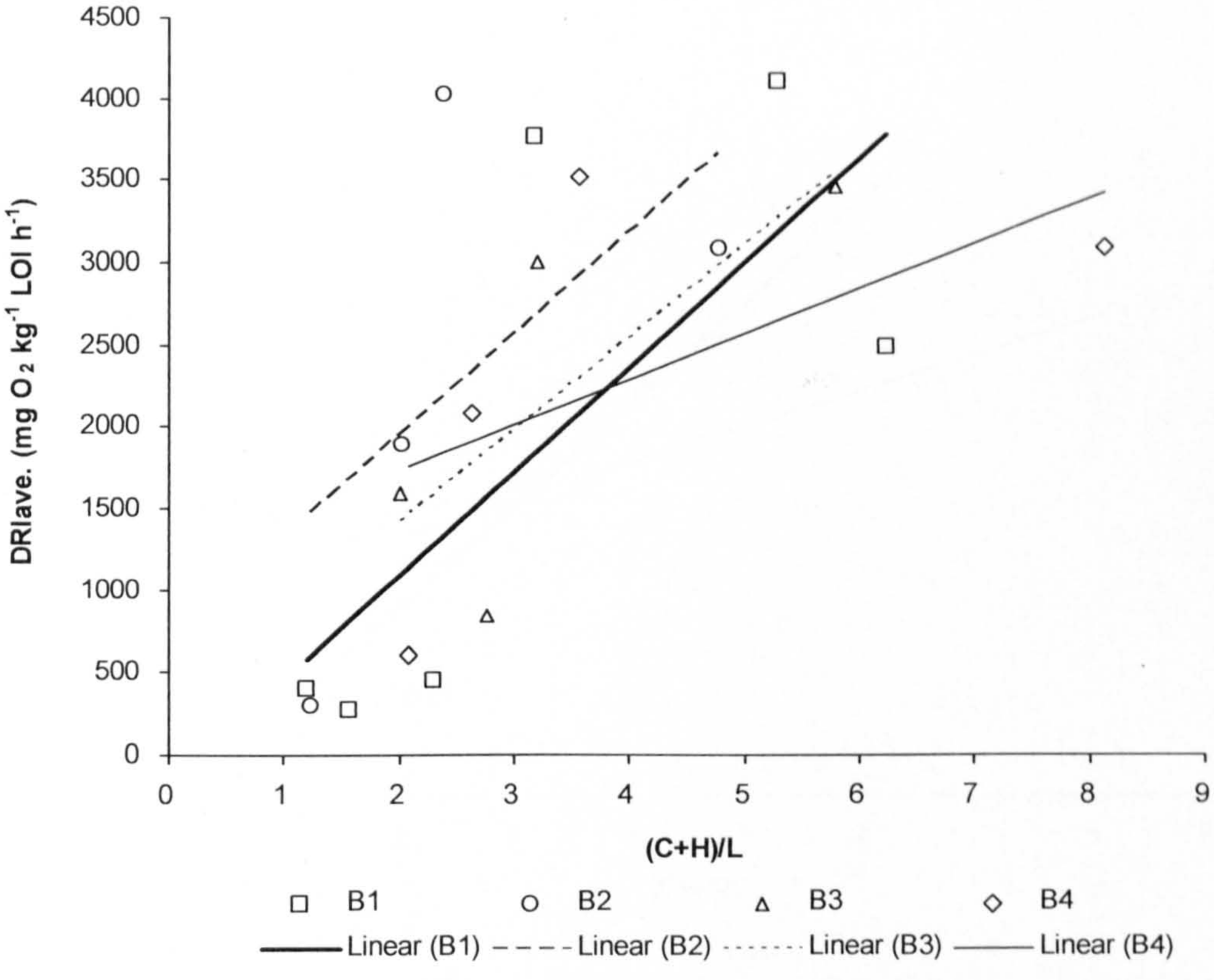


Figure 7.1c Correlation between (C+H)/L ratio and $DRI_{tot.}$ for treated waste (Combination of the four batches)



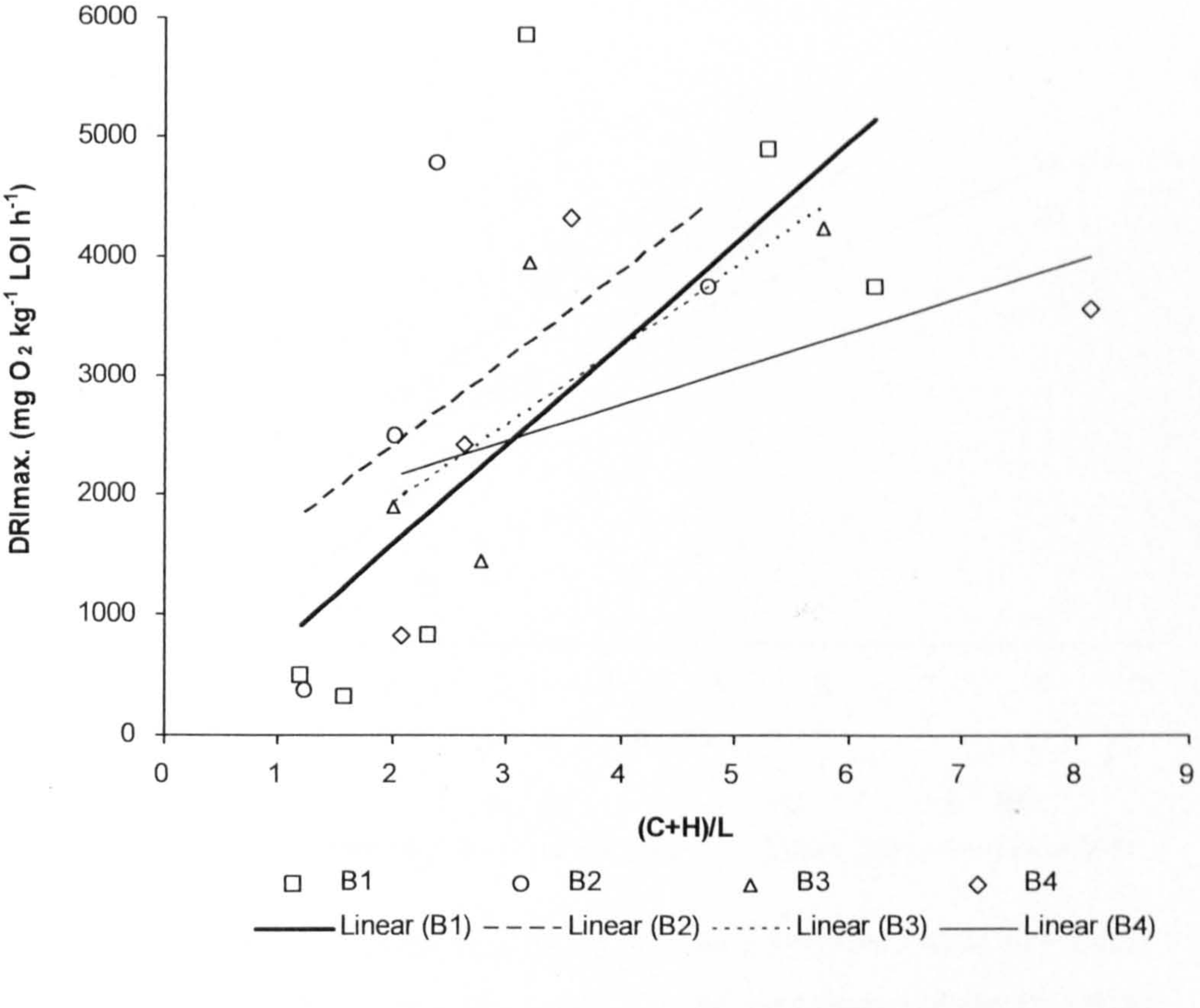
B1: y = 632.42x - 173.85, R2 = 0.5384

B2: y = 614.78x + 717.56, R2 = 0.34

B3: y = 558.16x + 304.8, R2 = 0.574

B4: y = 275.13x + 1191, R2 = 0.3442

Figure 7.2. Relationship between (C+H)/L ratio and $DRI_{ave.}$ in each batch of composting treatment



B1: y = 839.97x - 90.739, R2 = 0.4989

B2: y = 730.86x + 933.58, R2 = 0.3479

B3: y = 653.13x + 628.69, R2 = 0.5732

B4: y = 299.84x + 1553.5, R2 = 0.2963

Figure 7.3. Relationship between (C+H)/L ratio and $DRI_{max.}$ in each batch of composting treatment

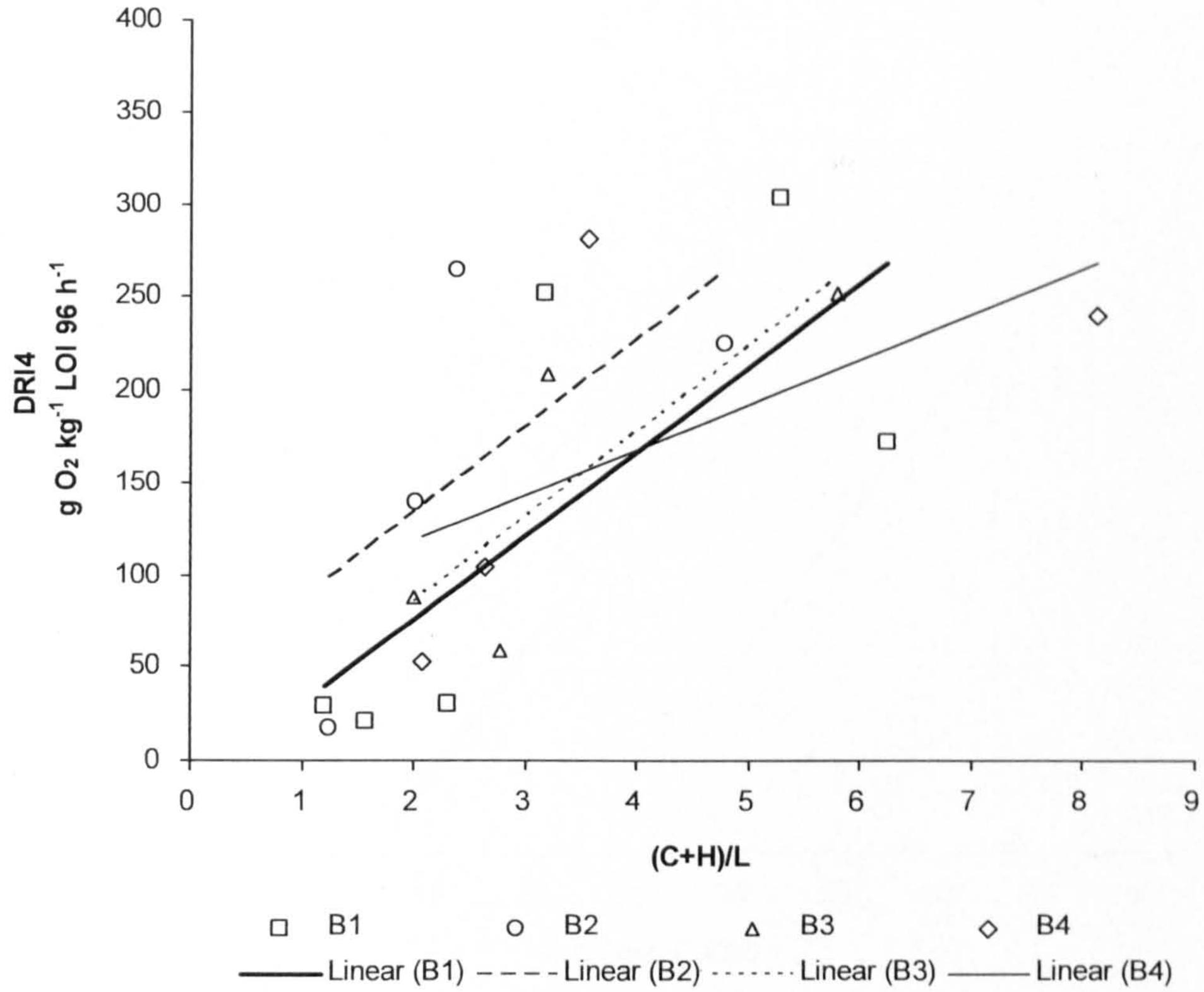


Figure 7.4. Relationship between (C+H)/L ratio and DRI4 in each batch of composting treatment

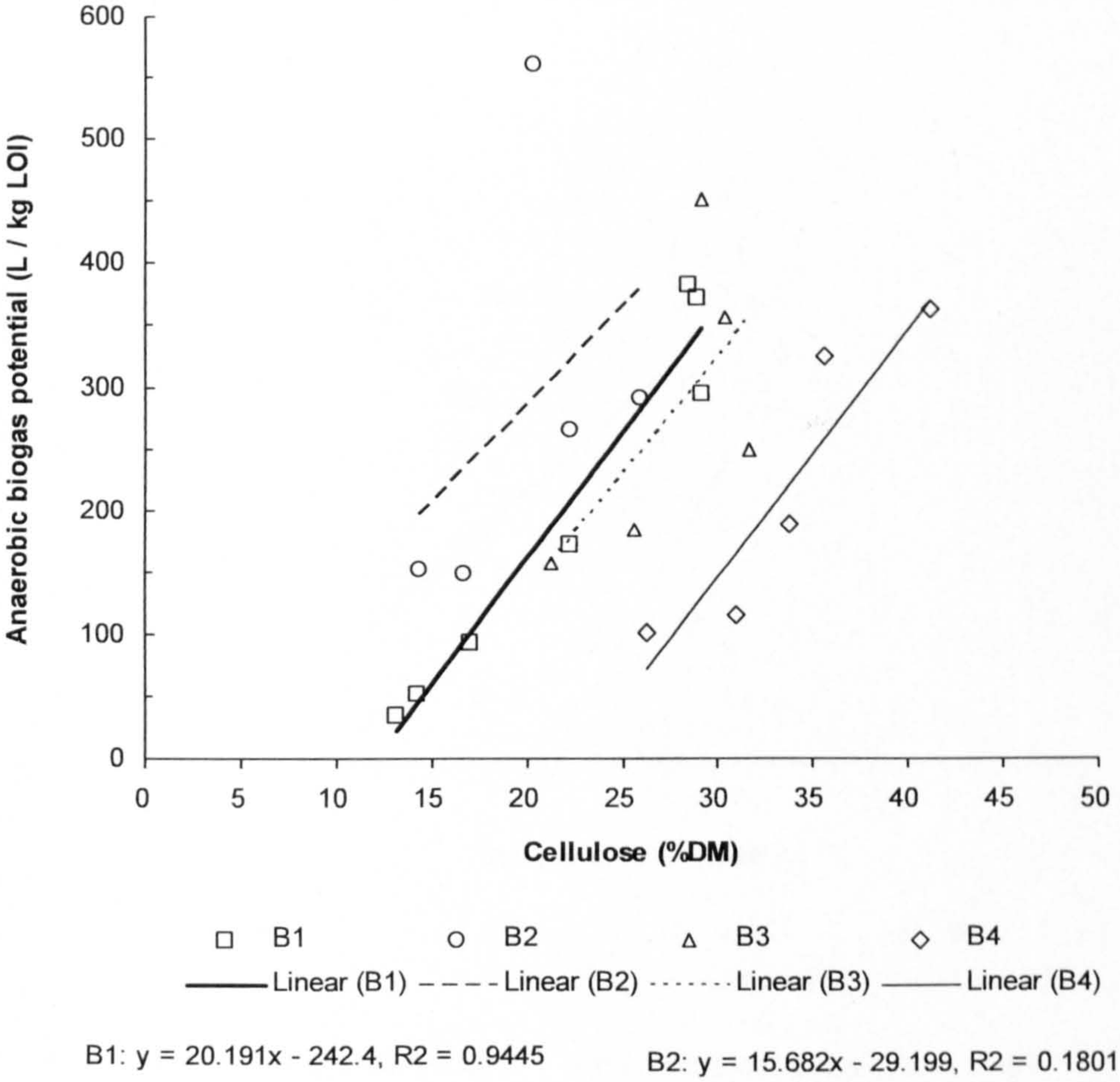
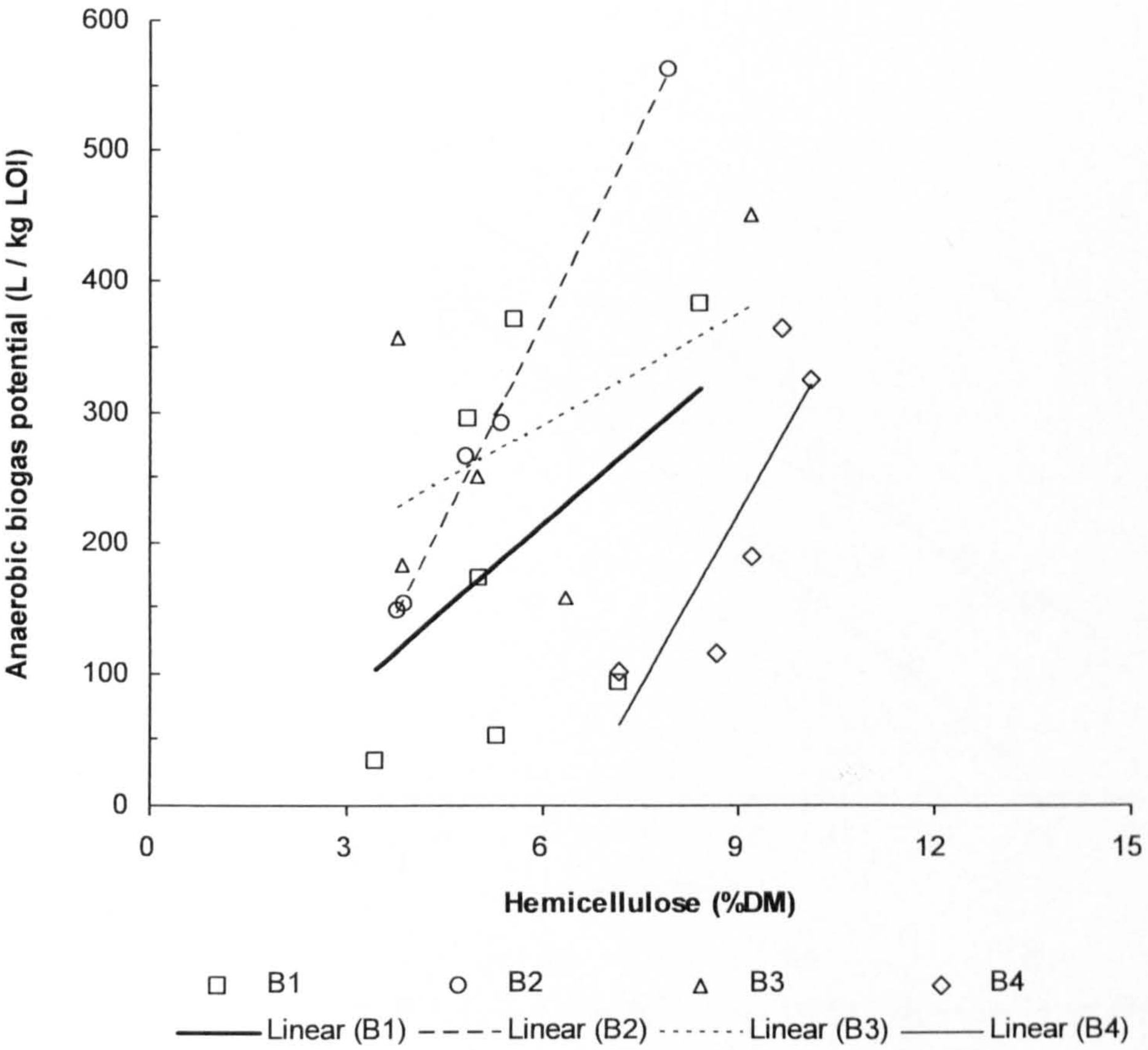


Figure 7.5. Relationship between cellulose content and anaerobic biogas potential for each batch of composting treatment



B1: y = 42.978x - 44.863, R2 = 0.2222 B2: y = 99.684x - 230.97, R2 = 0.9964

B3: y = 28.332x + 119.43, R2 = 0.2667 B4: y = 88.877x - 576.91, R2 = 0.7115

Figure 7.6. Relationship between hemicellulose content and anaerobic biogas potential for each batch of composting treatment

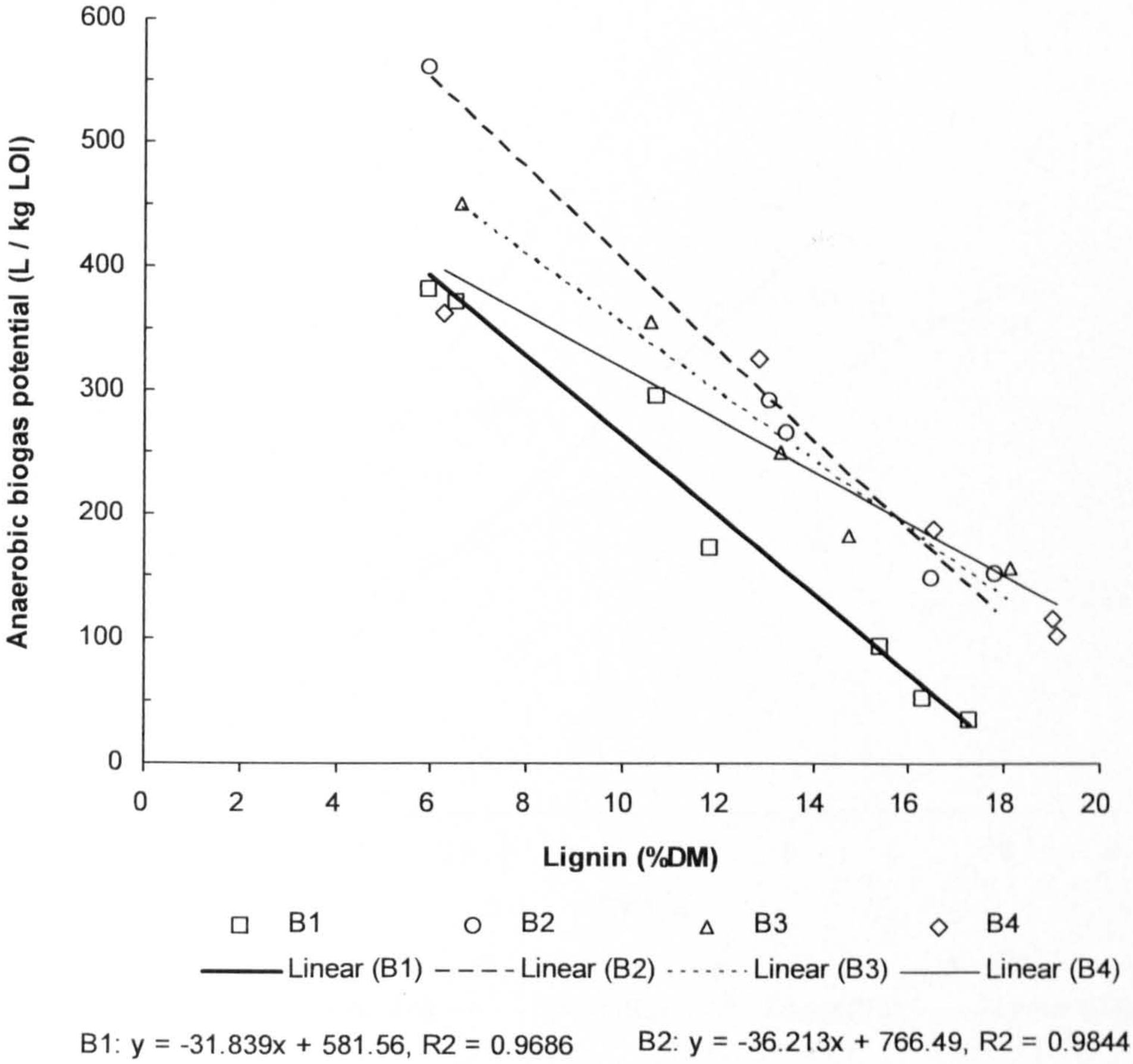


Figure 7.7. Relationship between lignin content and anaerobic biogas potential for each batch of composting treatment

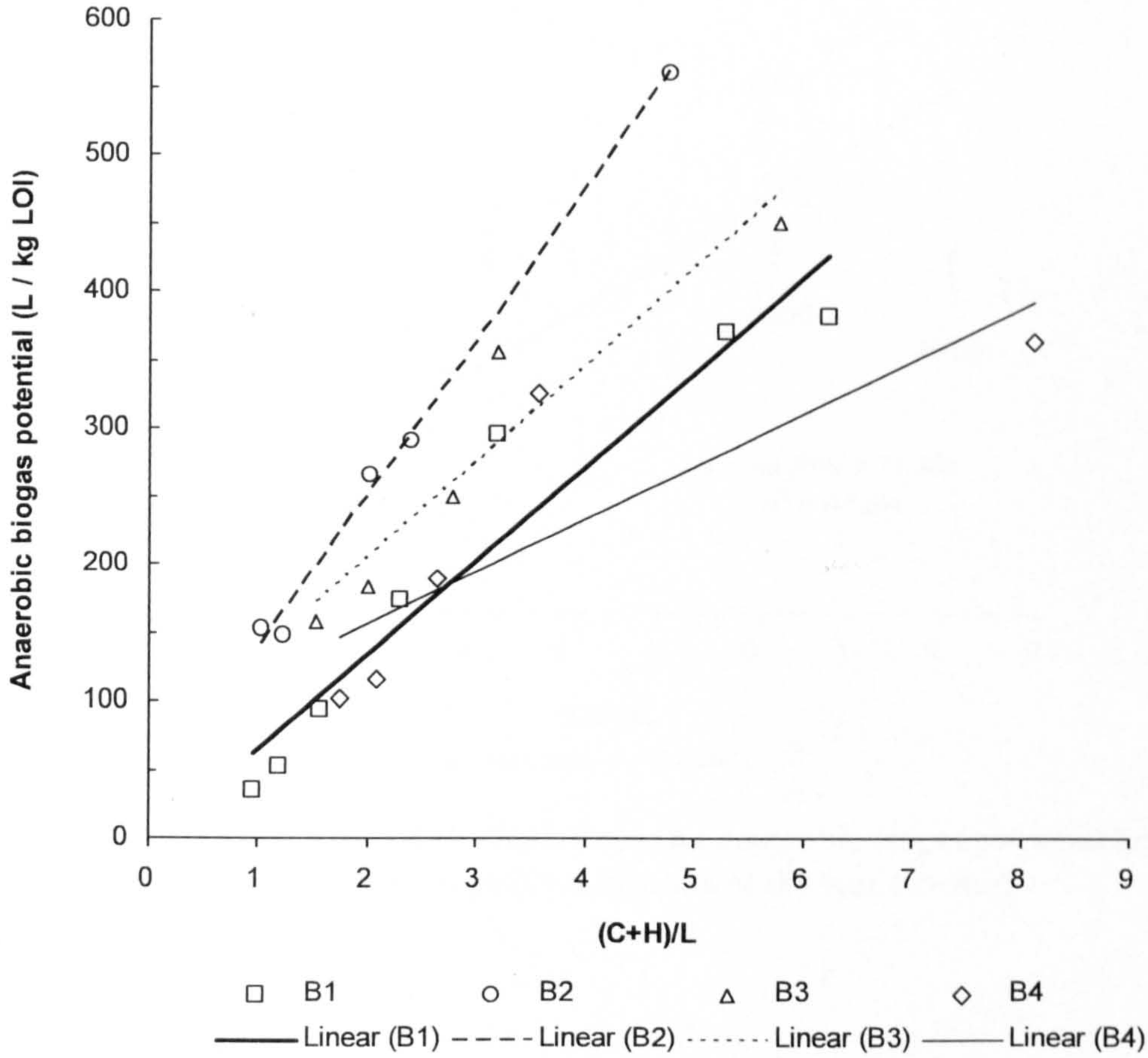


Figure 7.8a Correlation between (C+H)/L ratio and anaerobic biogas potential for each batch of composting treatment

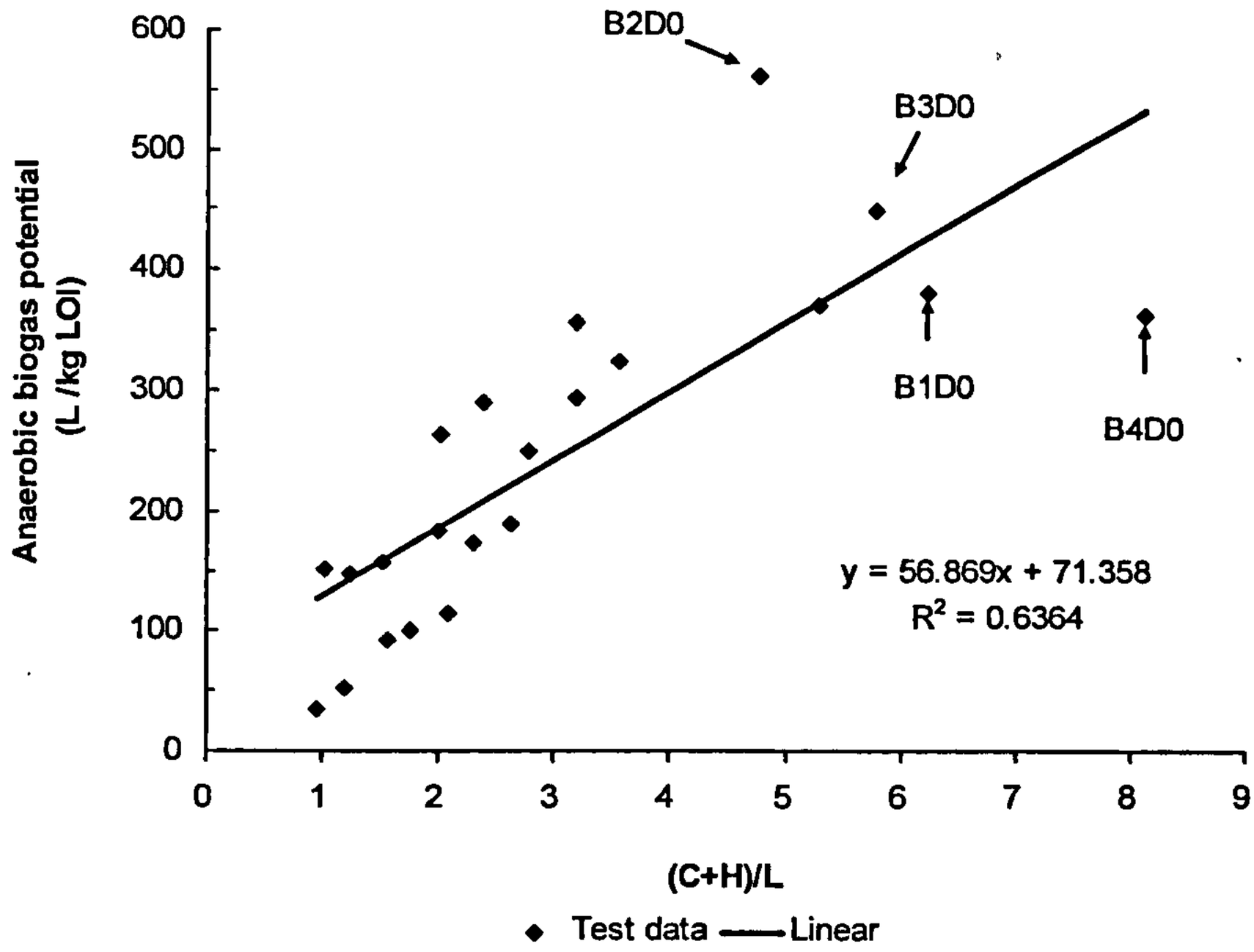


Figure 7.8b Correlation between (C+H)/L ratio and anaerobic biogas potential for untreated and treated waste (Combination of the four batches)

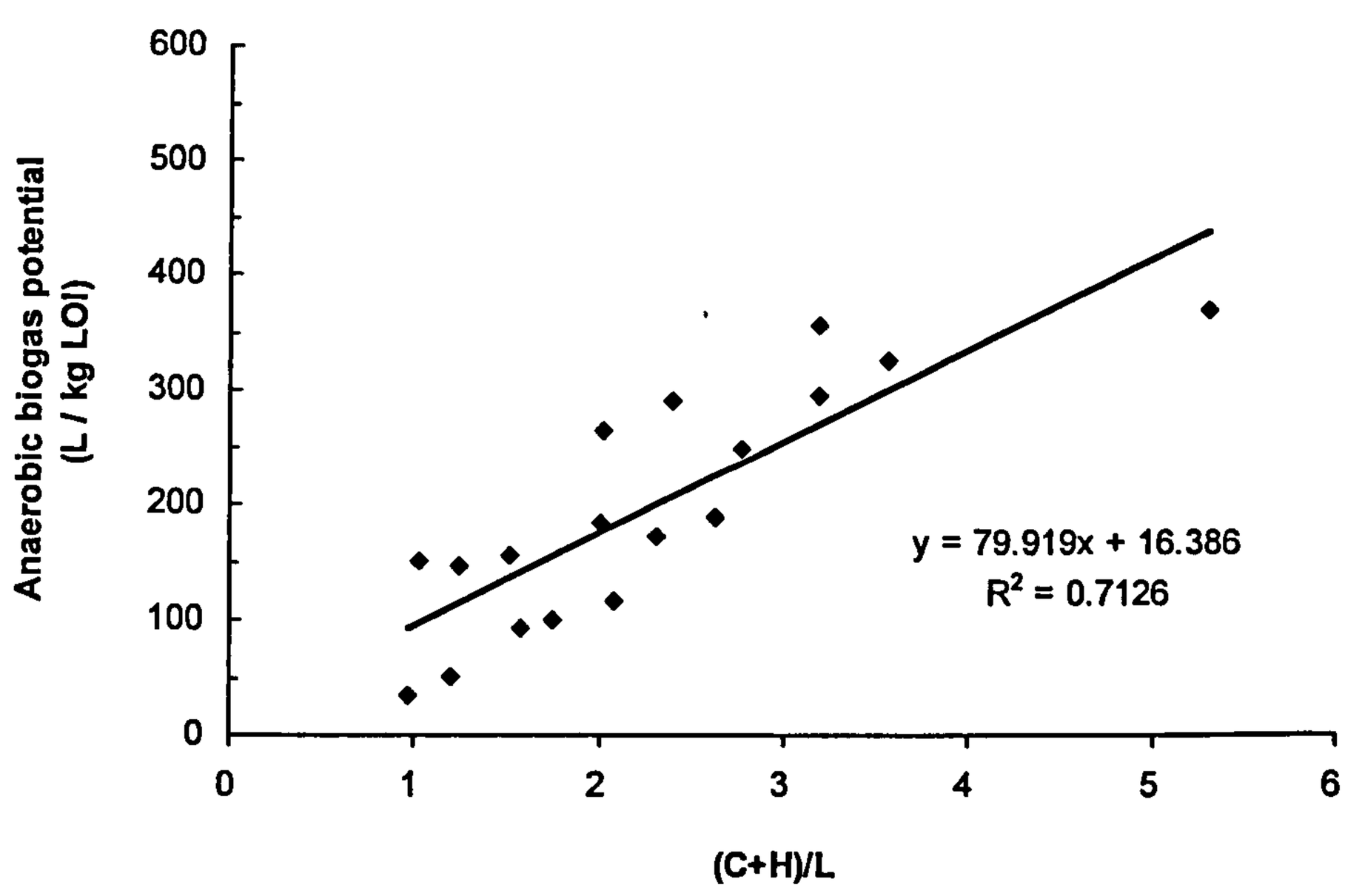
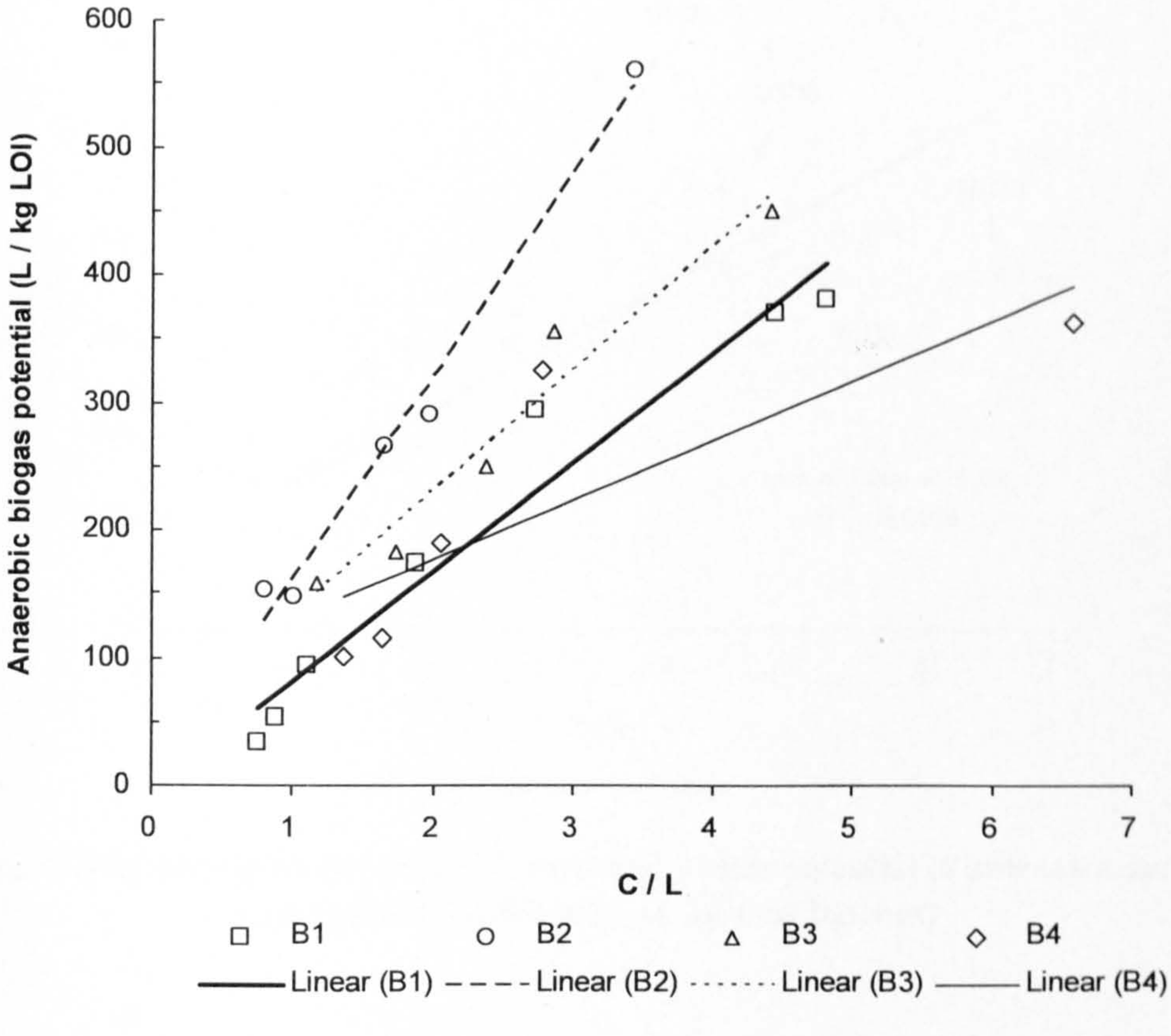


Figure 7.8c Correlation between (C+H)/L ratio and anaerobic biogas potential for the treated waste (Combination of the four batches)



B1: y = 86.056x - 4.3033, R2 = 0.9531 B2: y = 160.2x - 1.6223, R2 = 0.9866

B3: y = 96.637x + 36.513, R2 = 0.9525 B4: y = 46.748x + 83.677, R2 = 0.6959

Figure 7.9a Correlation between C/L ratio and biogas potential for each batch of composting treatment

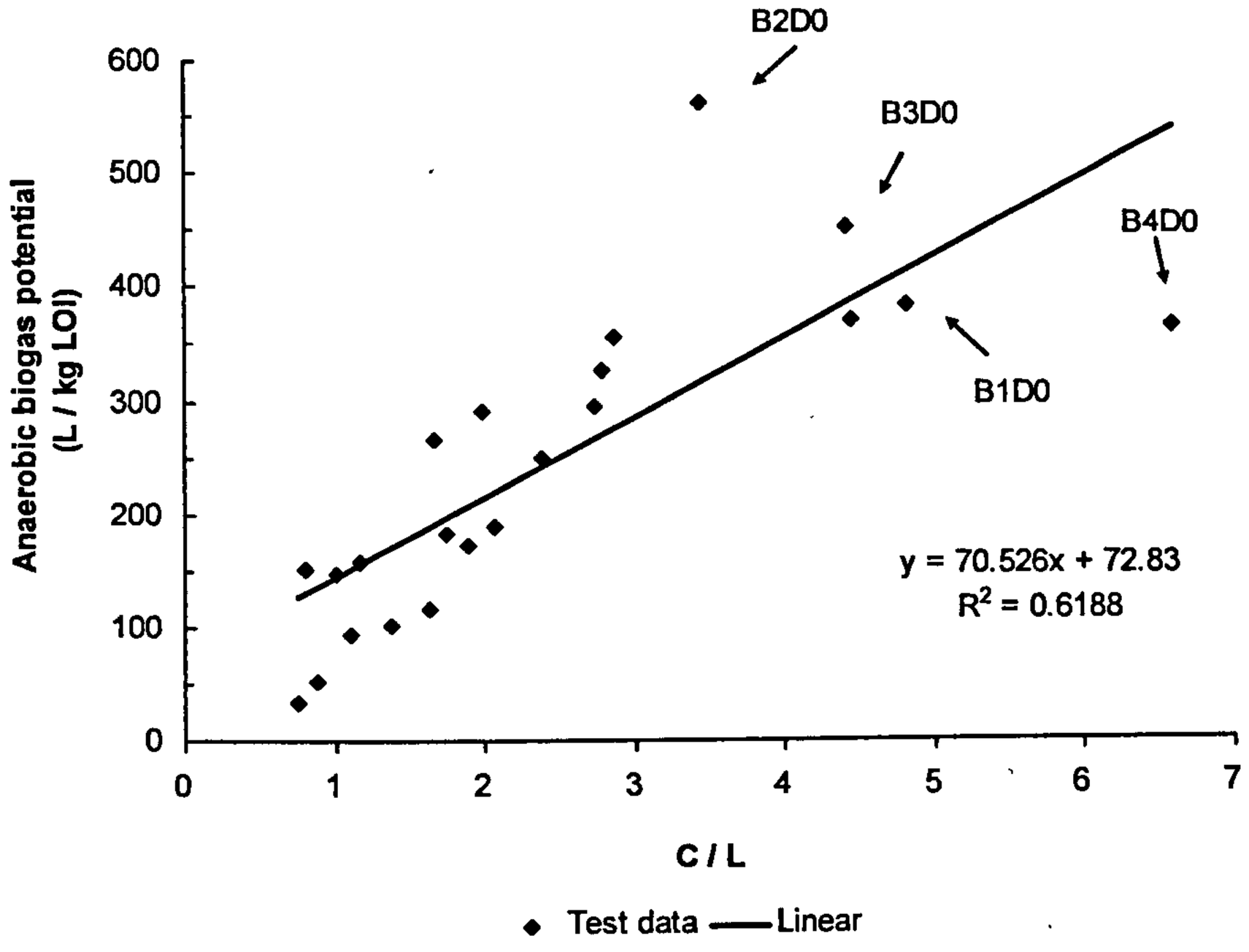


Figure7.9b Correlation between C/L ratio and biogas potential of untreated and treated waste (Combination of the four batches)

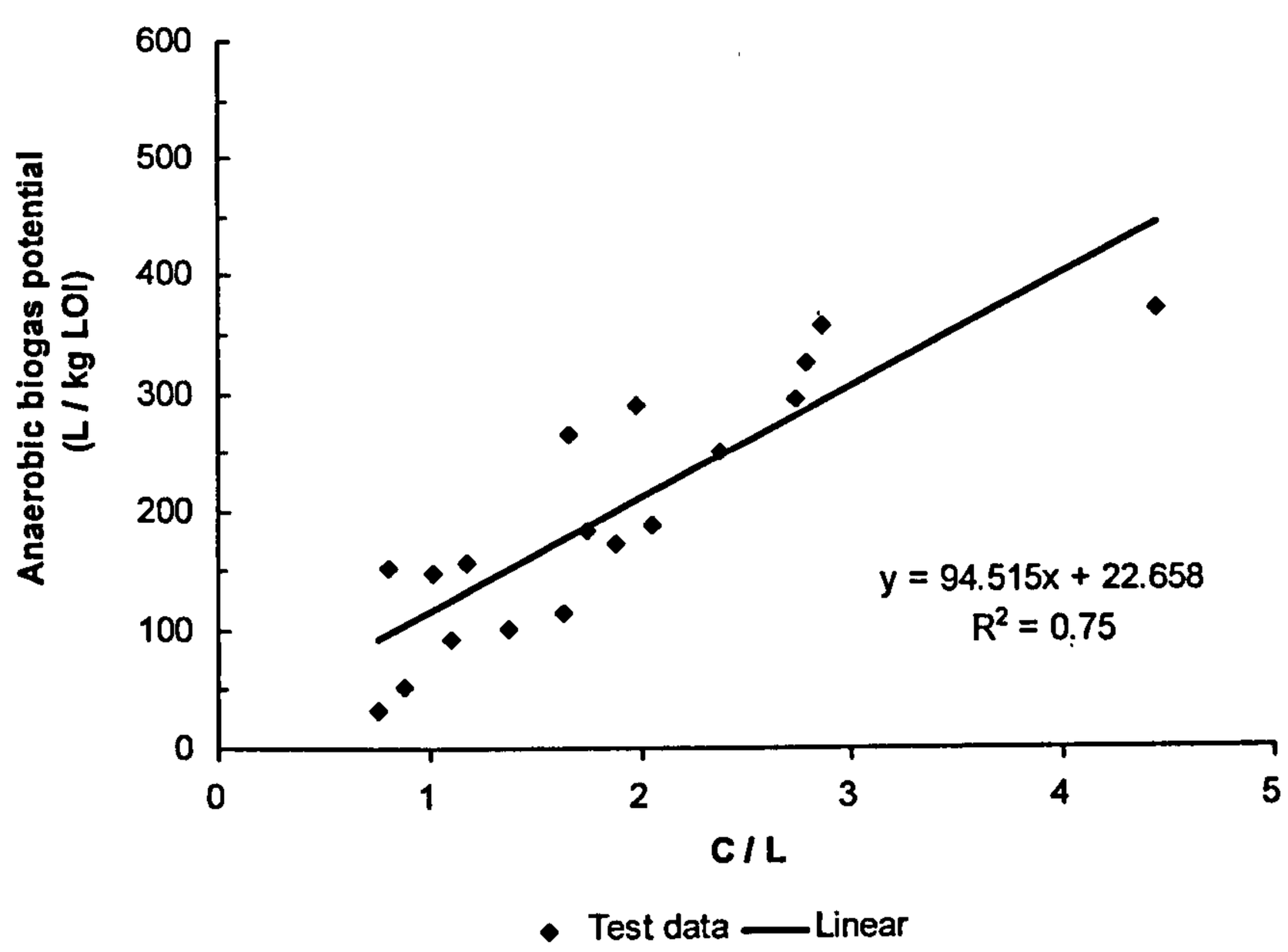


Figure 7.9c Correlation between C/L ratio and biogas potential of treated waste (Combination of the four batches)

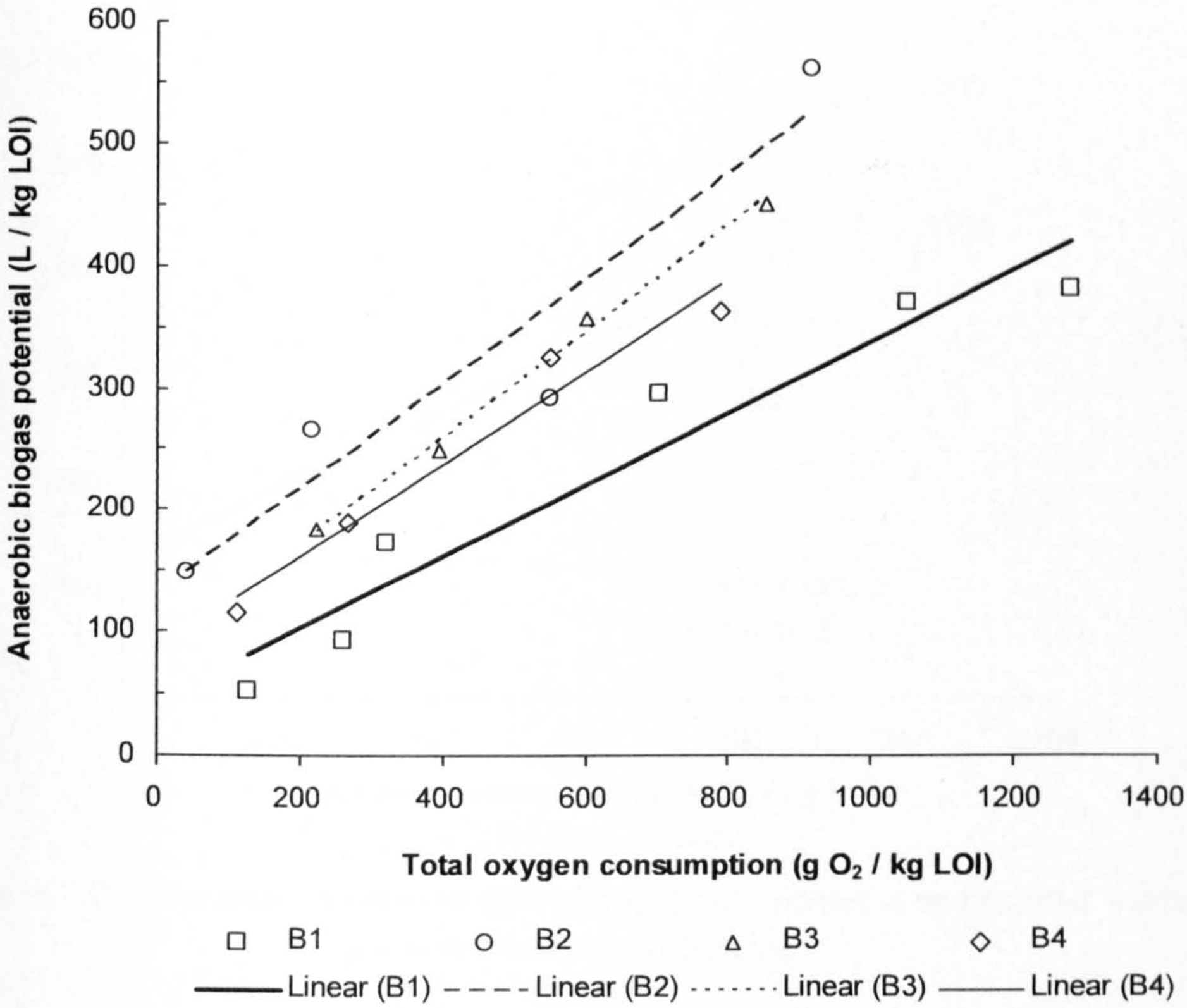


Figure 7.10a Correlation between $DRI_{tot.}$ and BMP in each batch

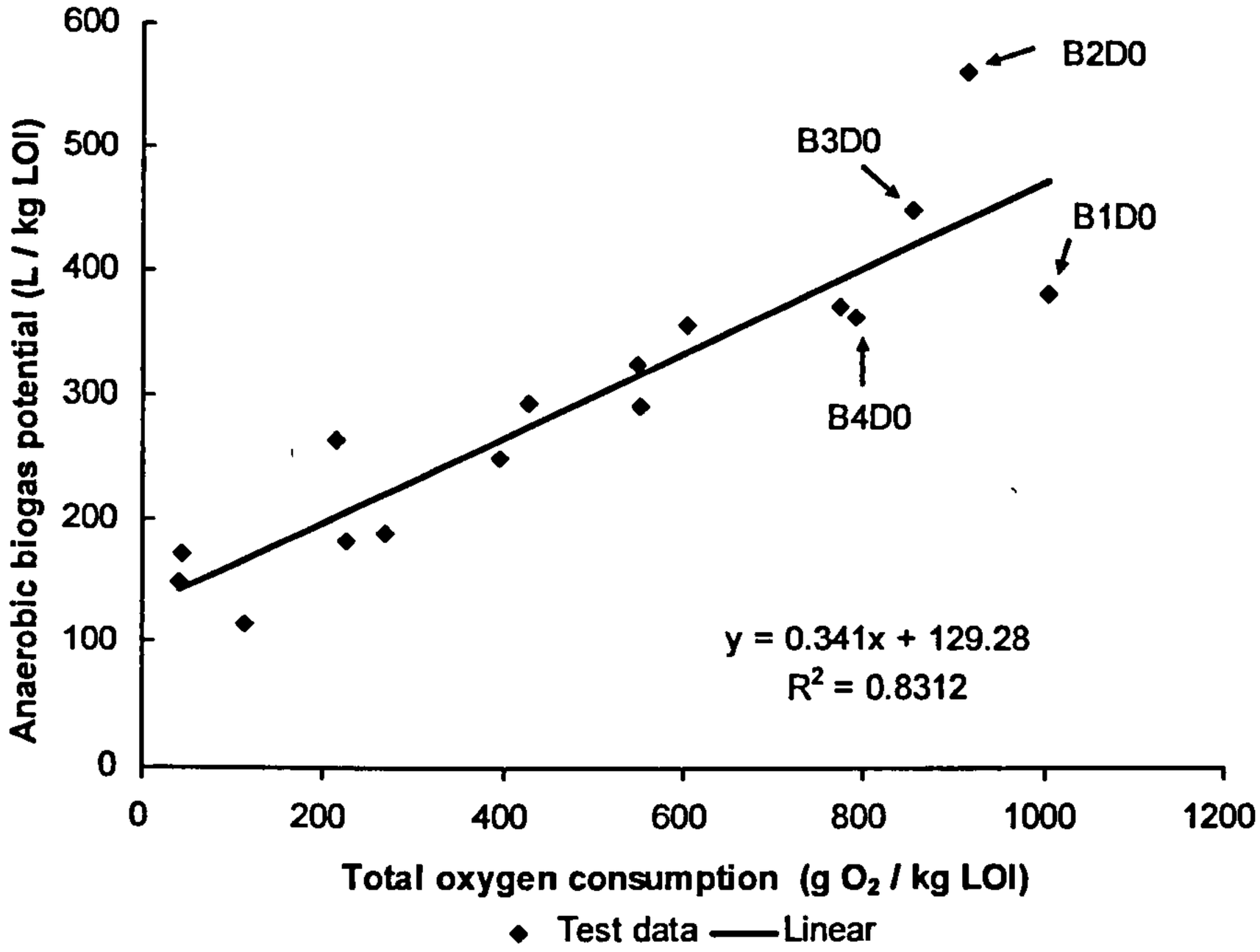


Figure 7.10b Correlation between DRI and BMP of untreated and treated wastes (Combination of four batches)

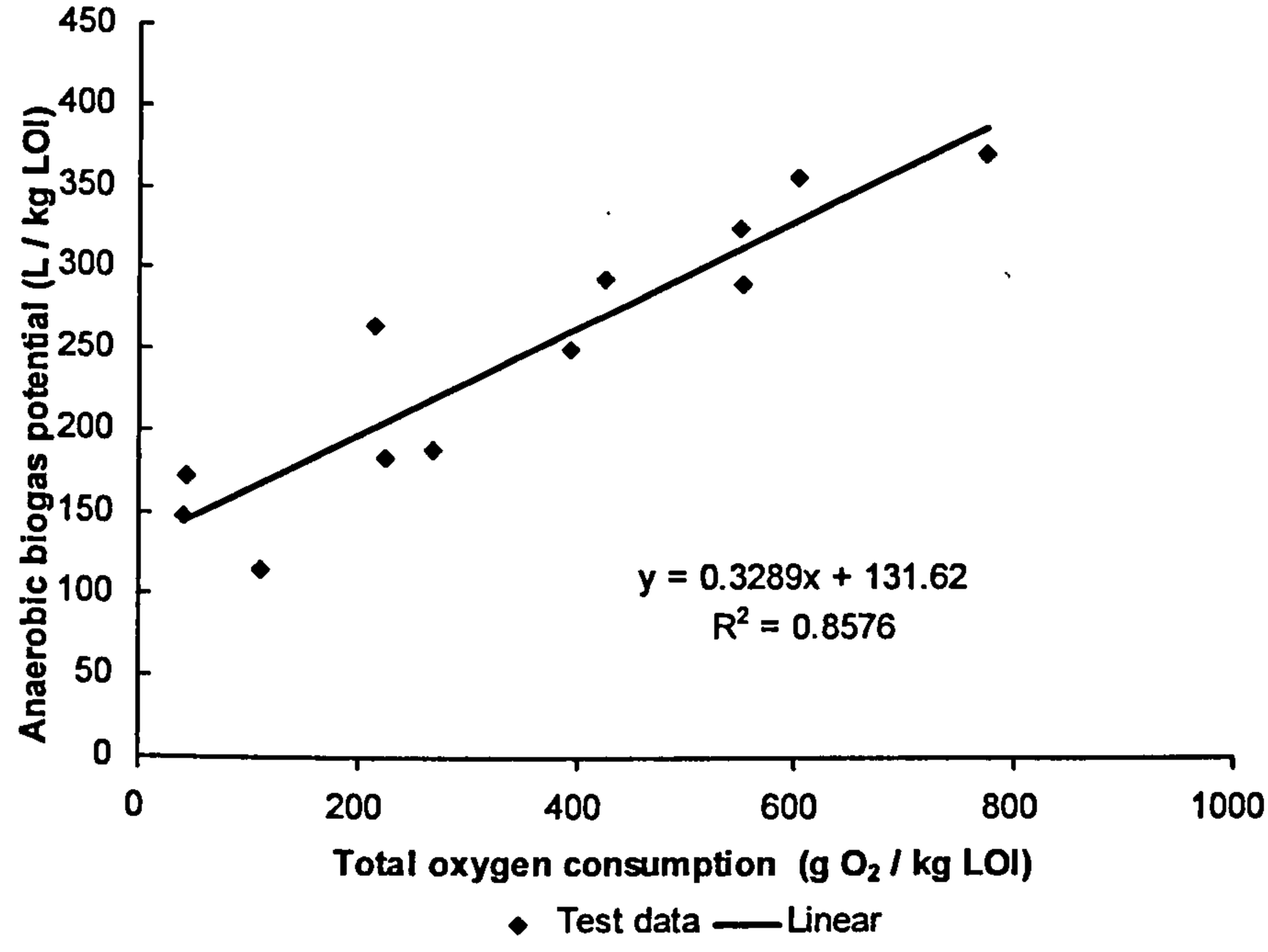
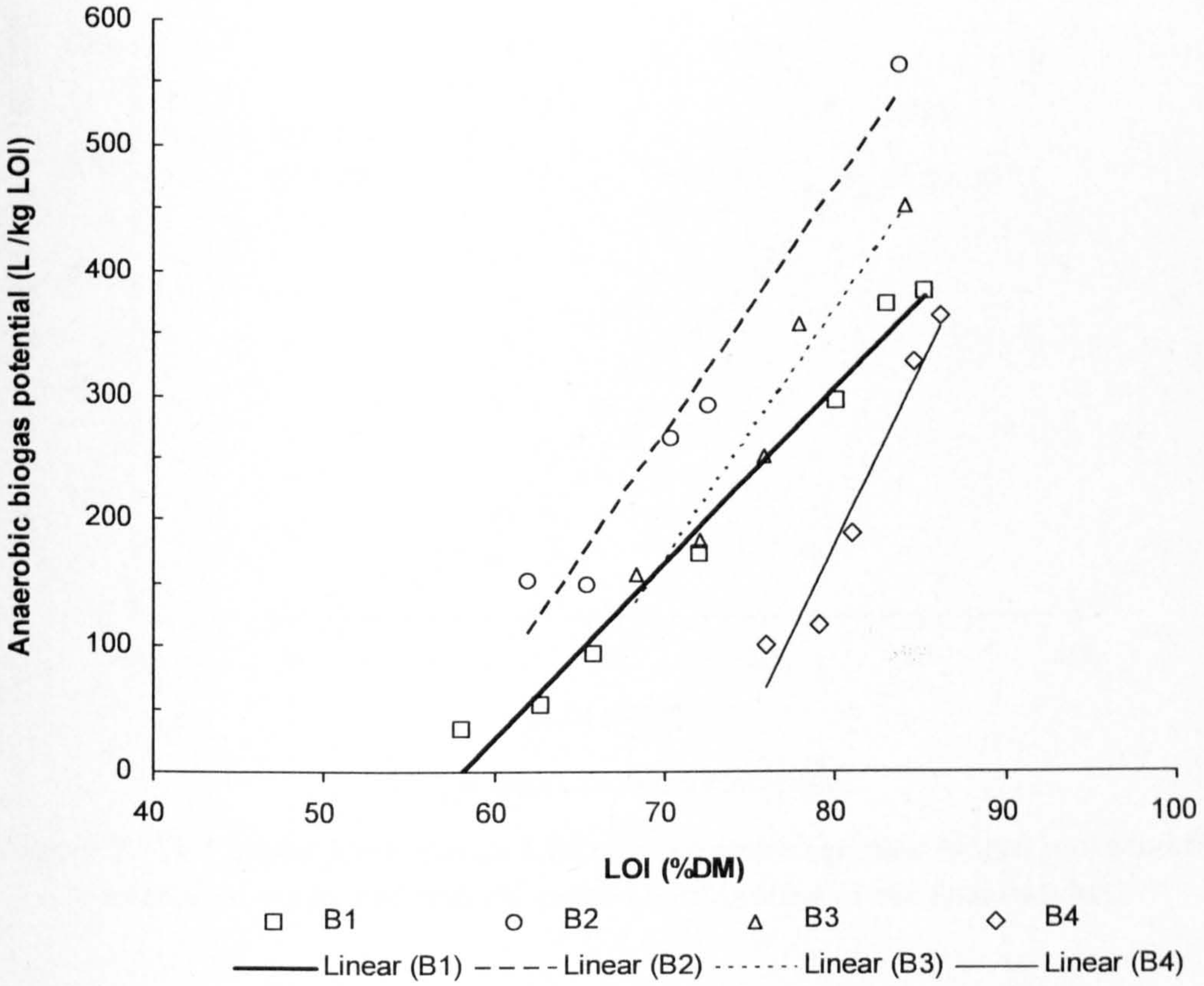


Figure 7.10c Correlation between DRI and BMP of treated waste (Combination of four batches)



B1: y = 13.835x - 802.83, R2 = 0.9795

B2: y = 19.696x - 1112.1,R2 = 0.965

B3: y = 19.899x - 1226.1, R2 = 0.9456

B4: y = 28.104x - 2066.5, R2 = 0.9409

Figure 7.11a Correlation between LOI content and anaerobic biogas potential of the samples in each batch

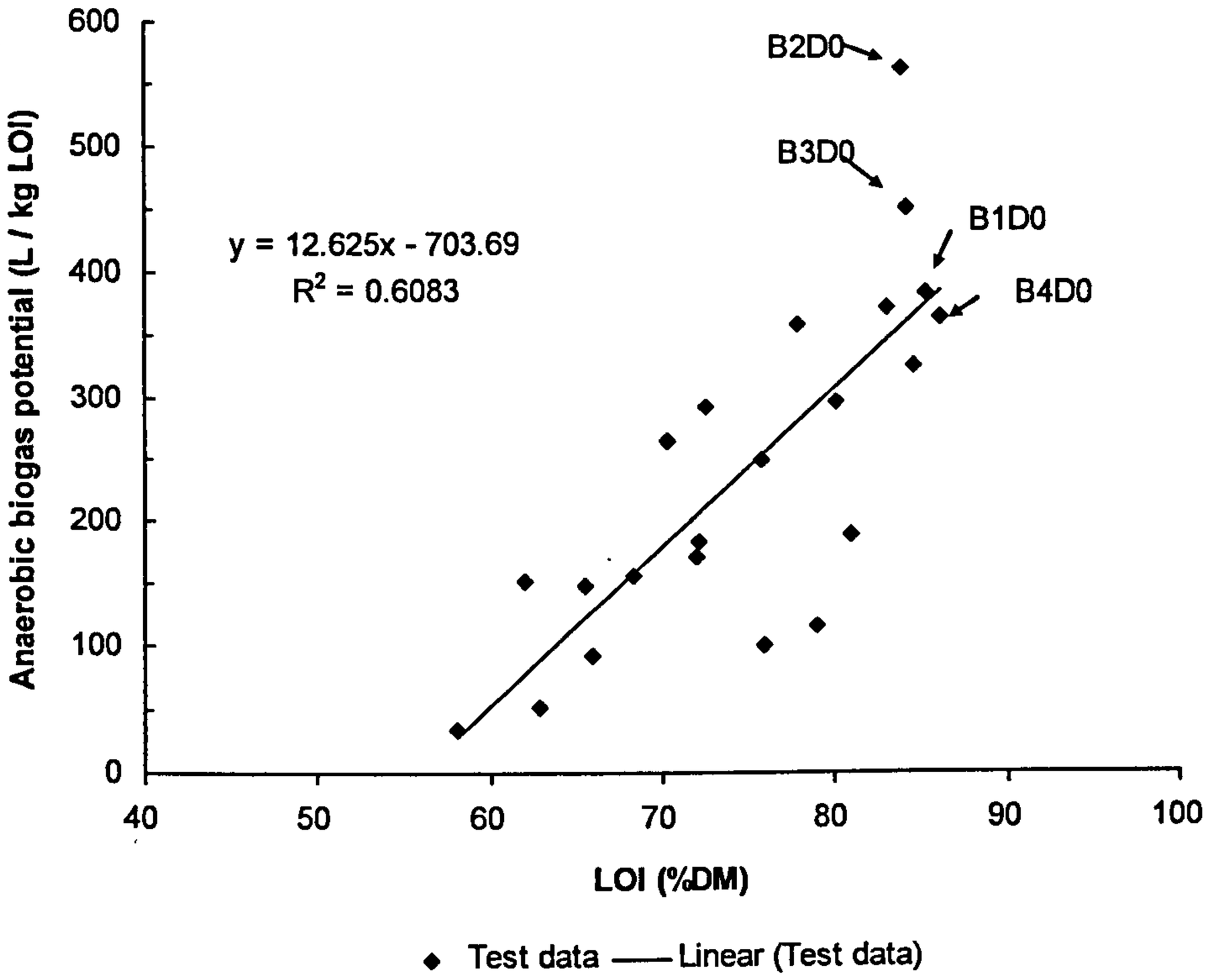


Figure 7.11b Correlation between LOI content and anaerobic biogas potential for untreated waste and treated waste (combination of the four batches)

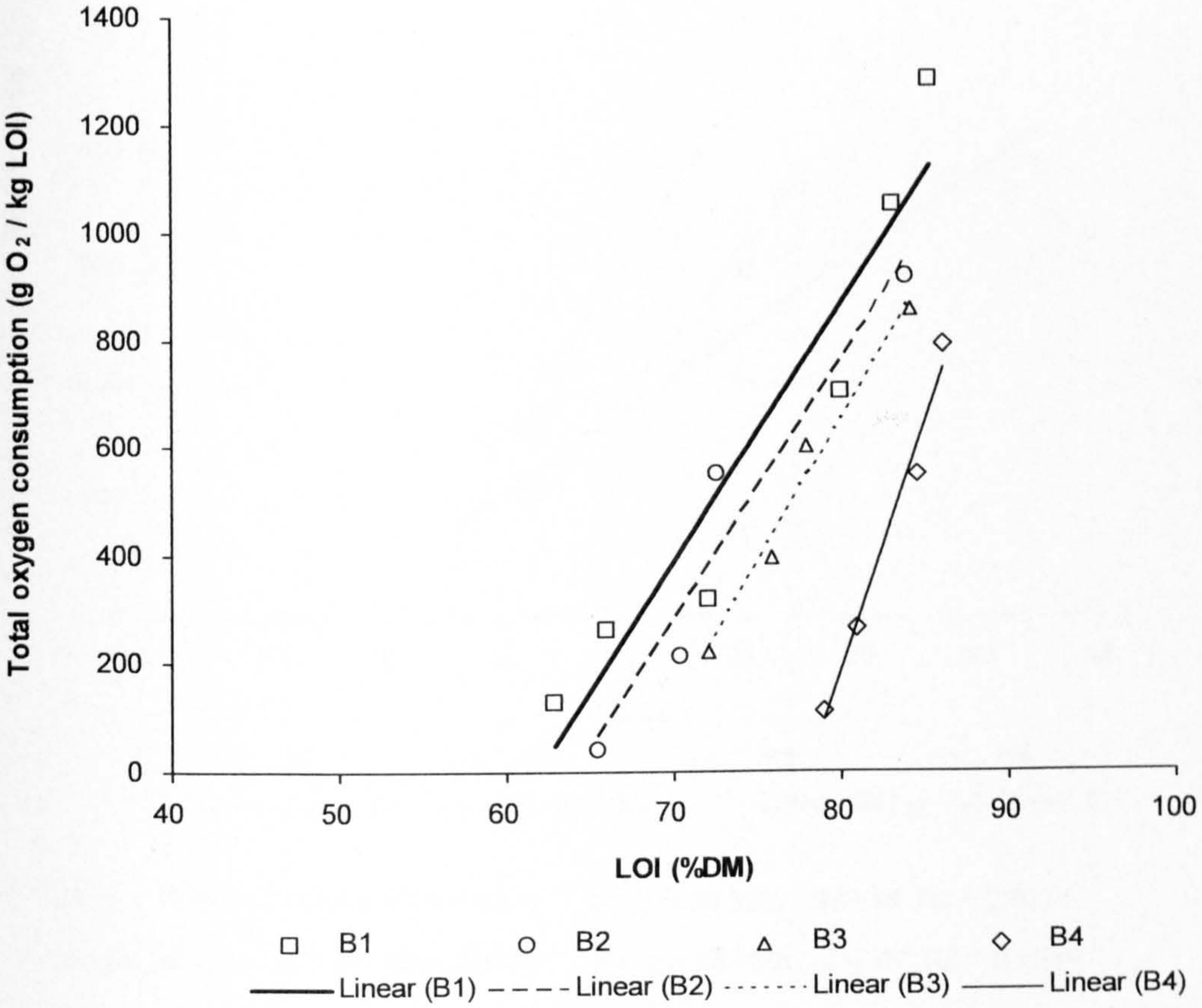
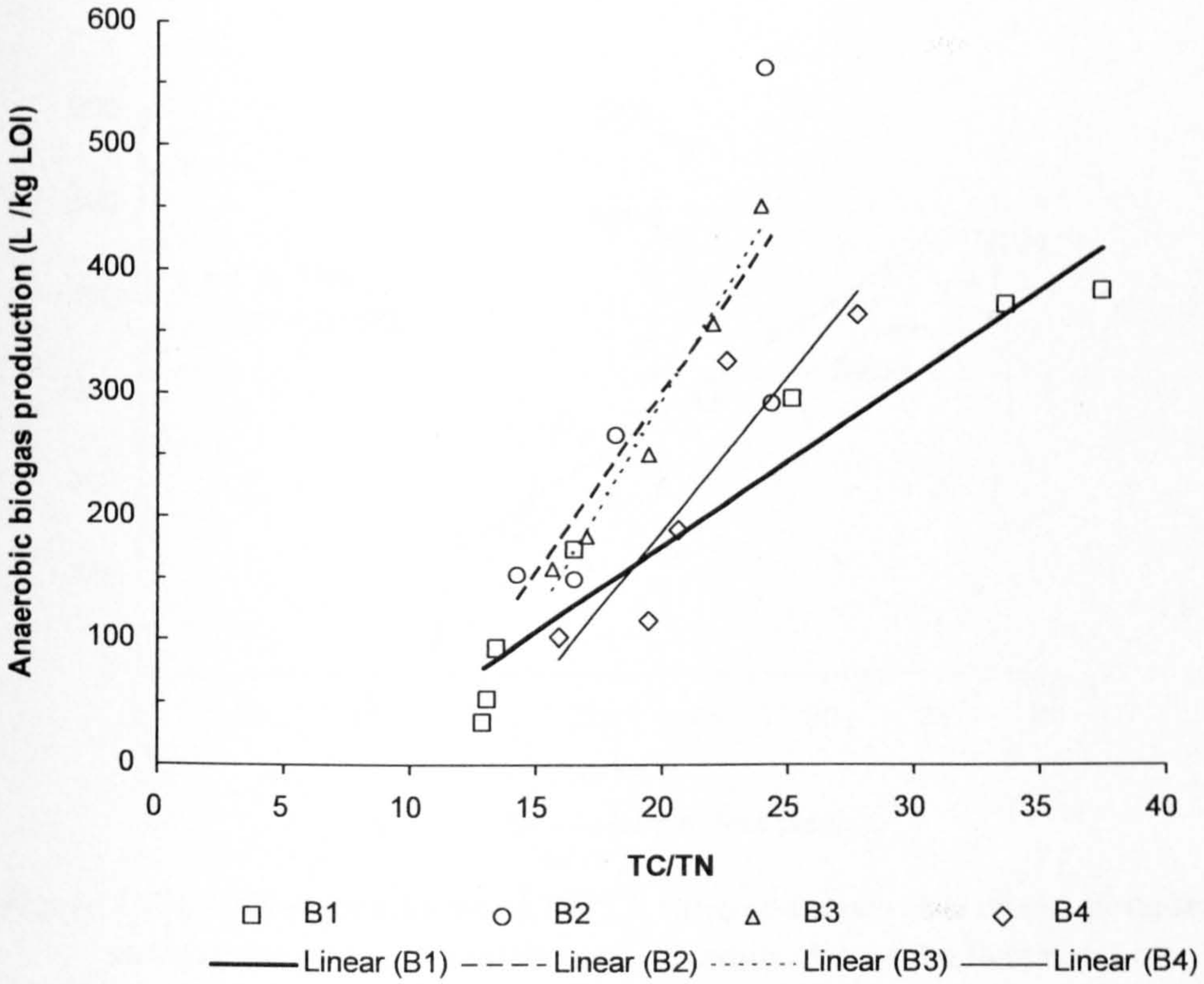


Figure 7.12. Correlation between LOI content and $DRI_{tot.}$ of the samples in each batch



B1: y = 13.848x - 101.41, R2 = 0.9378 B2: y = 29.346x - 288.18, R2 = 0.6276

B3: y = 35.459x - 415.61, R2 = 0.9774 B4: y = 25.189x - 317.05, R2 = 0.8289

Figure 7.13a Correlation between TC/TN ratio and anaerobic biogas potential of the samples in each batch

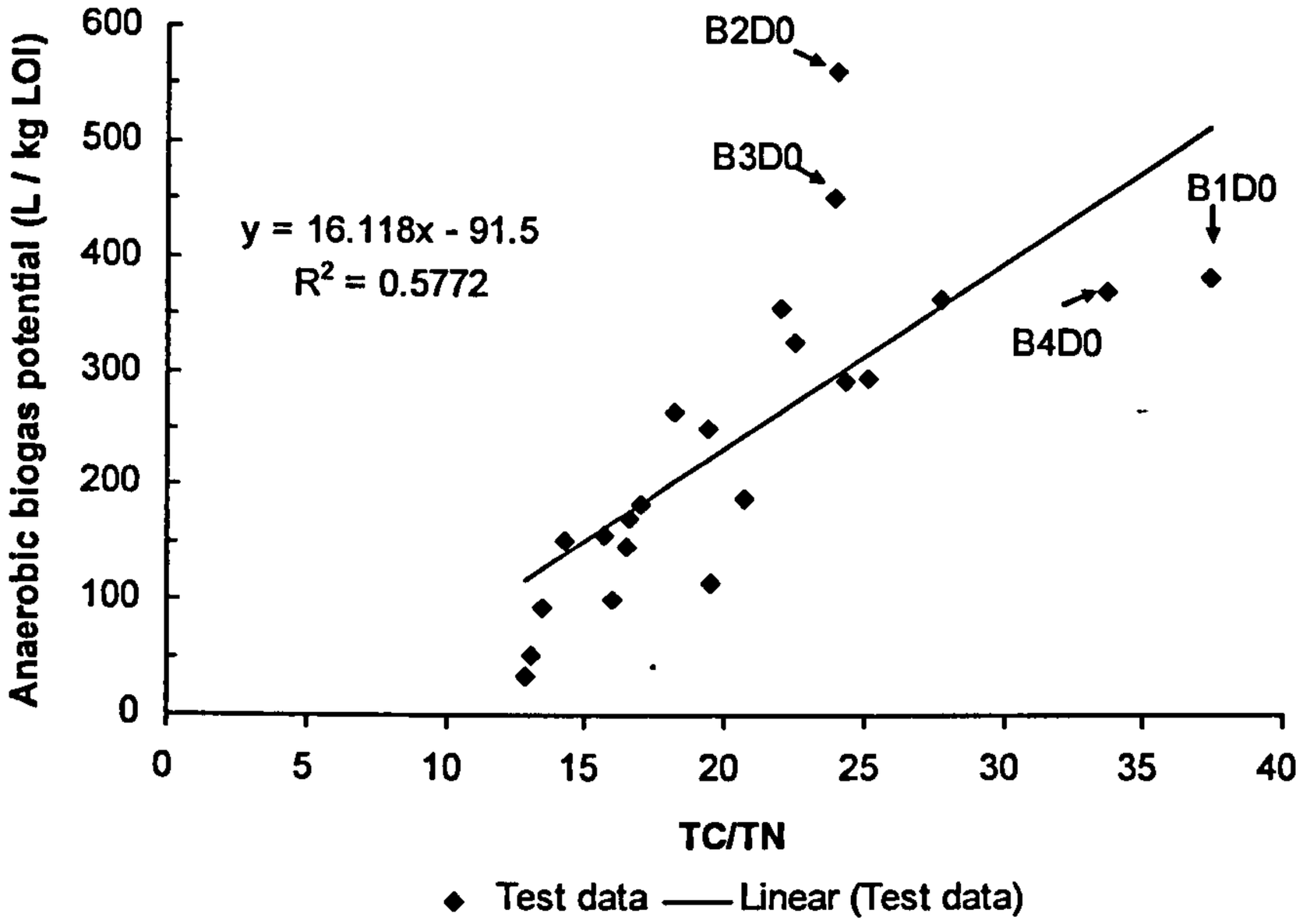


Figure 7.13b Correlation between TC/TN ratio and anaerobic biogas potential for untreated waste and treated waste (combination of the four batches)

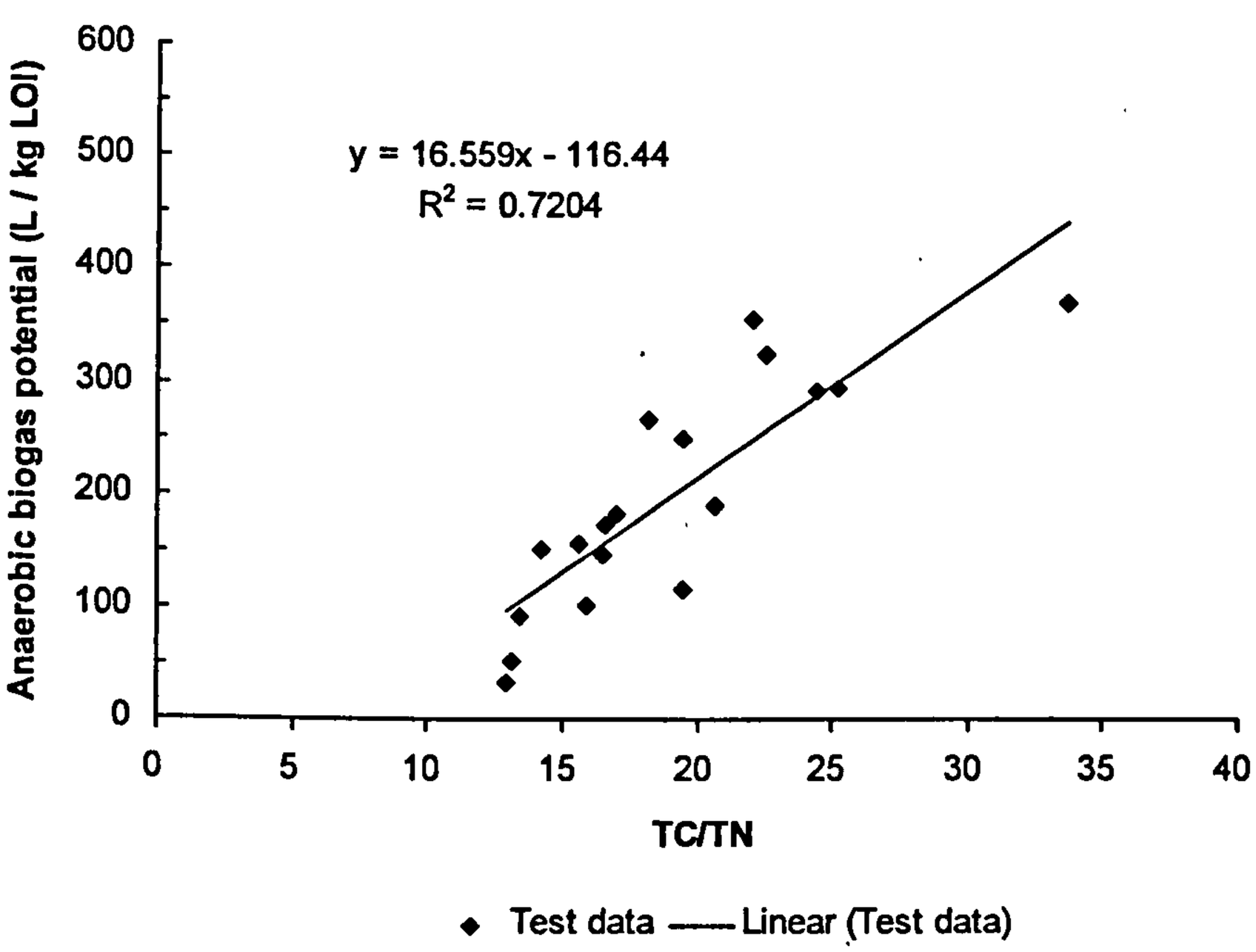


Figure 7.13c Correlation between TC/TN ratio and anaerobic biogas potential for treated waste (combination of the four batches)

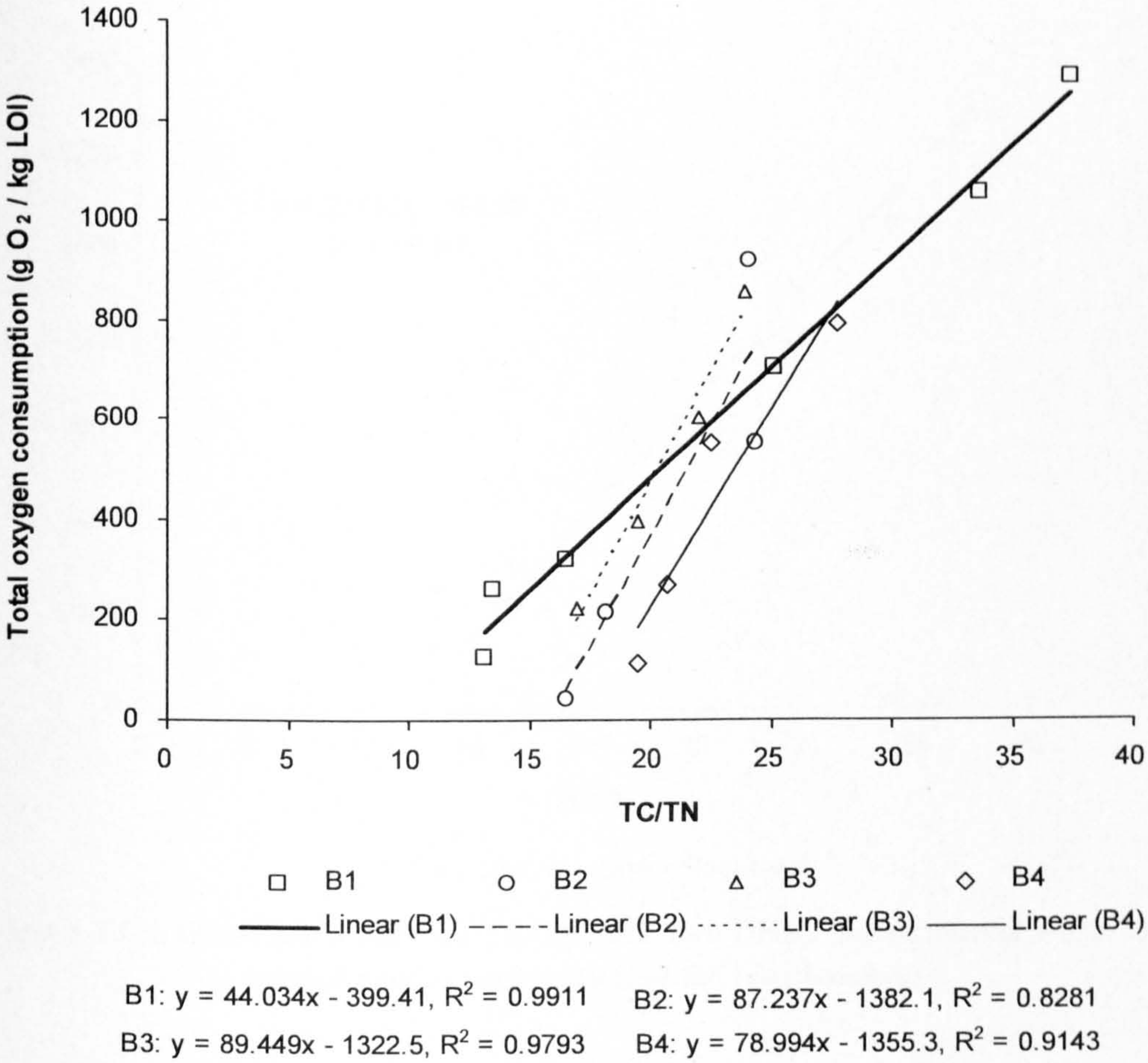


Figure 7.14a Correlation between TC/TN ratio and $DRI_{tot.}$ of the samples in each batch

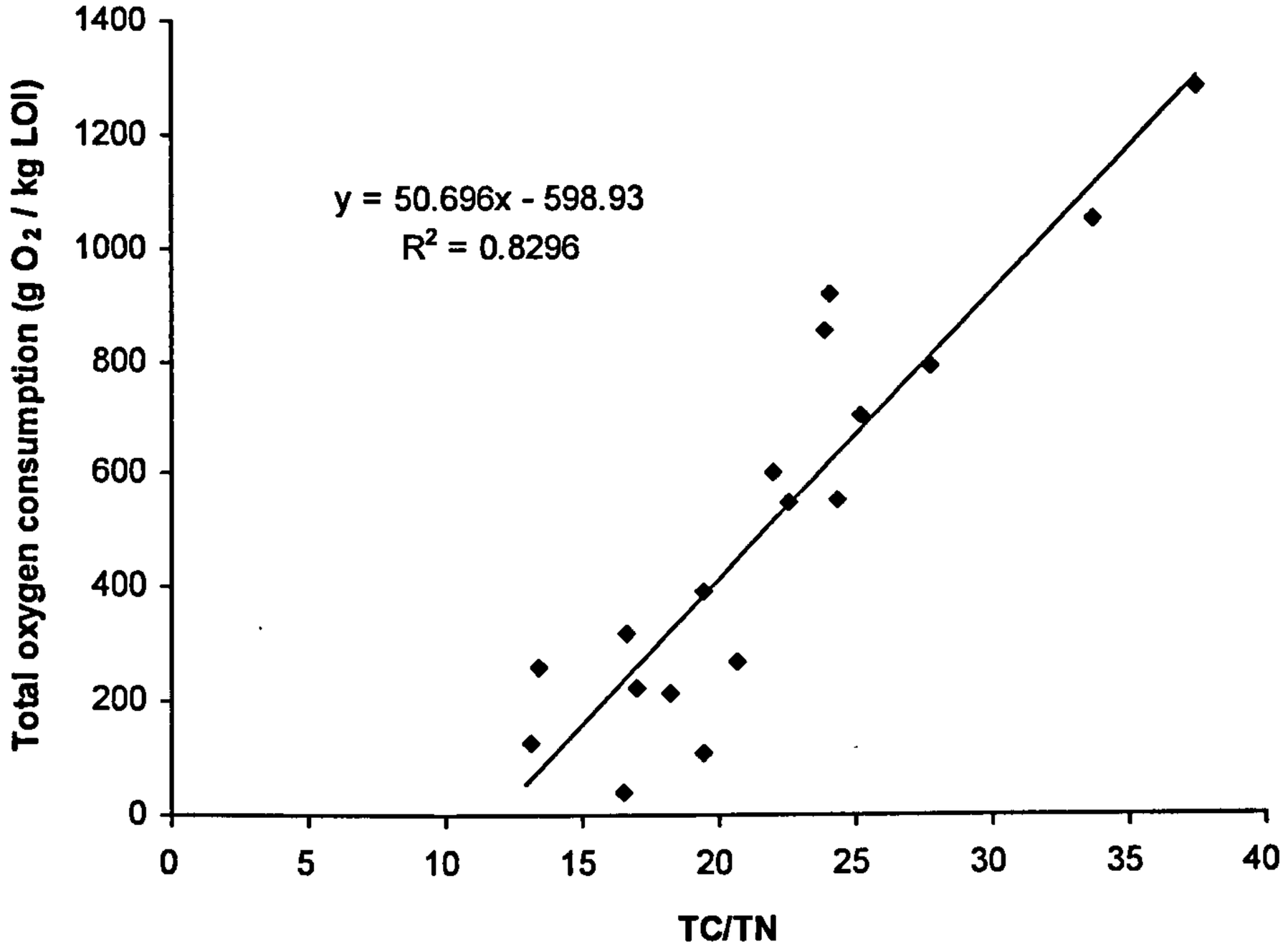


Figure 7.14b Correlation between TC/TN ratio and $DRI_{tot.}$ for untreated waste and treated waste (combination of the four batches)

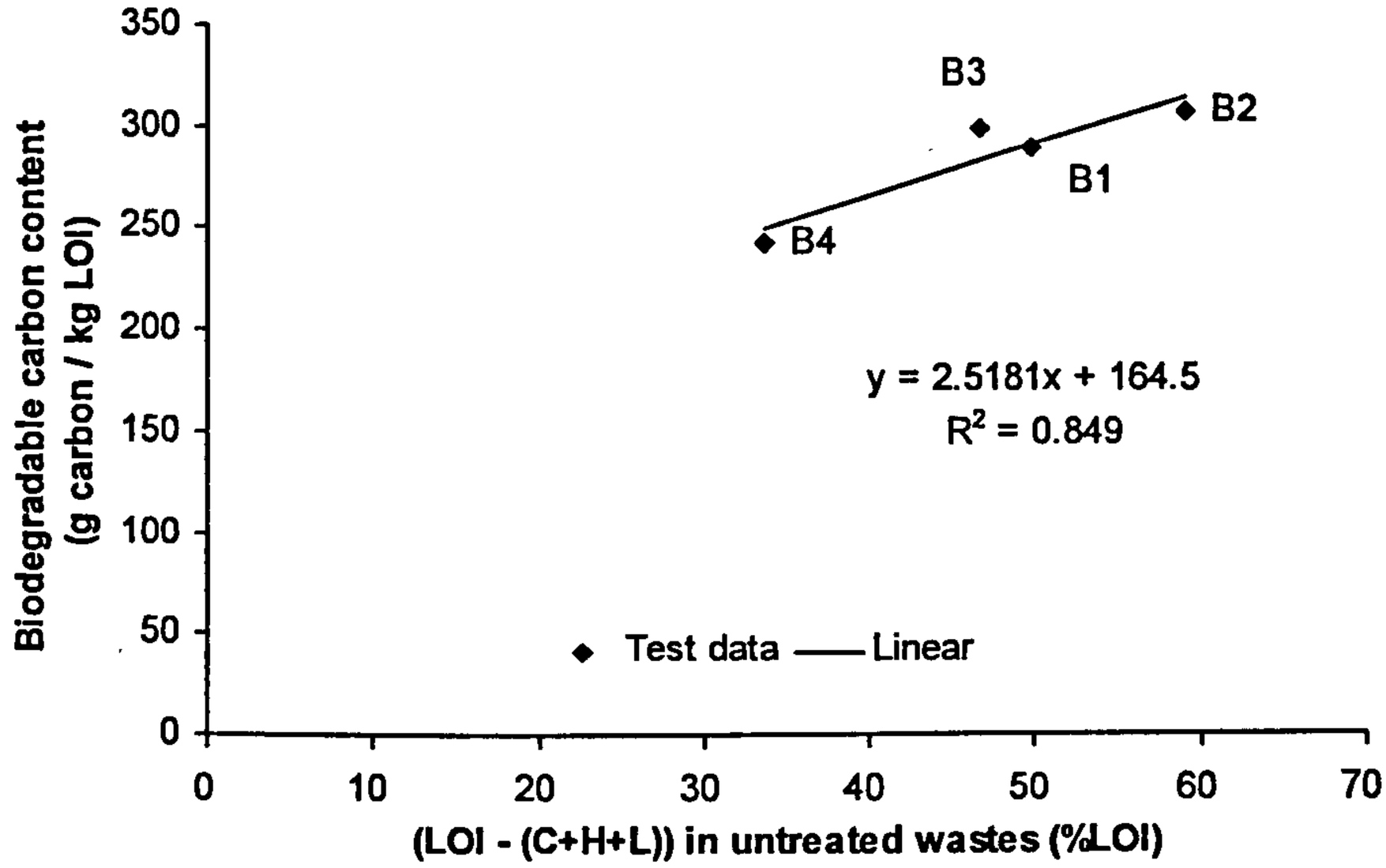


Figure 7.15. Correlation between cell soluble content and biodegradable carbon content for untreated wastes

(C: cellulose; H: hemicellulose; L: lignin)

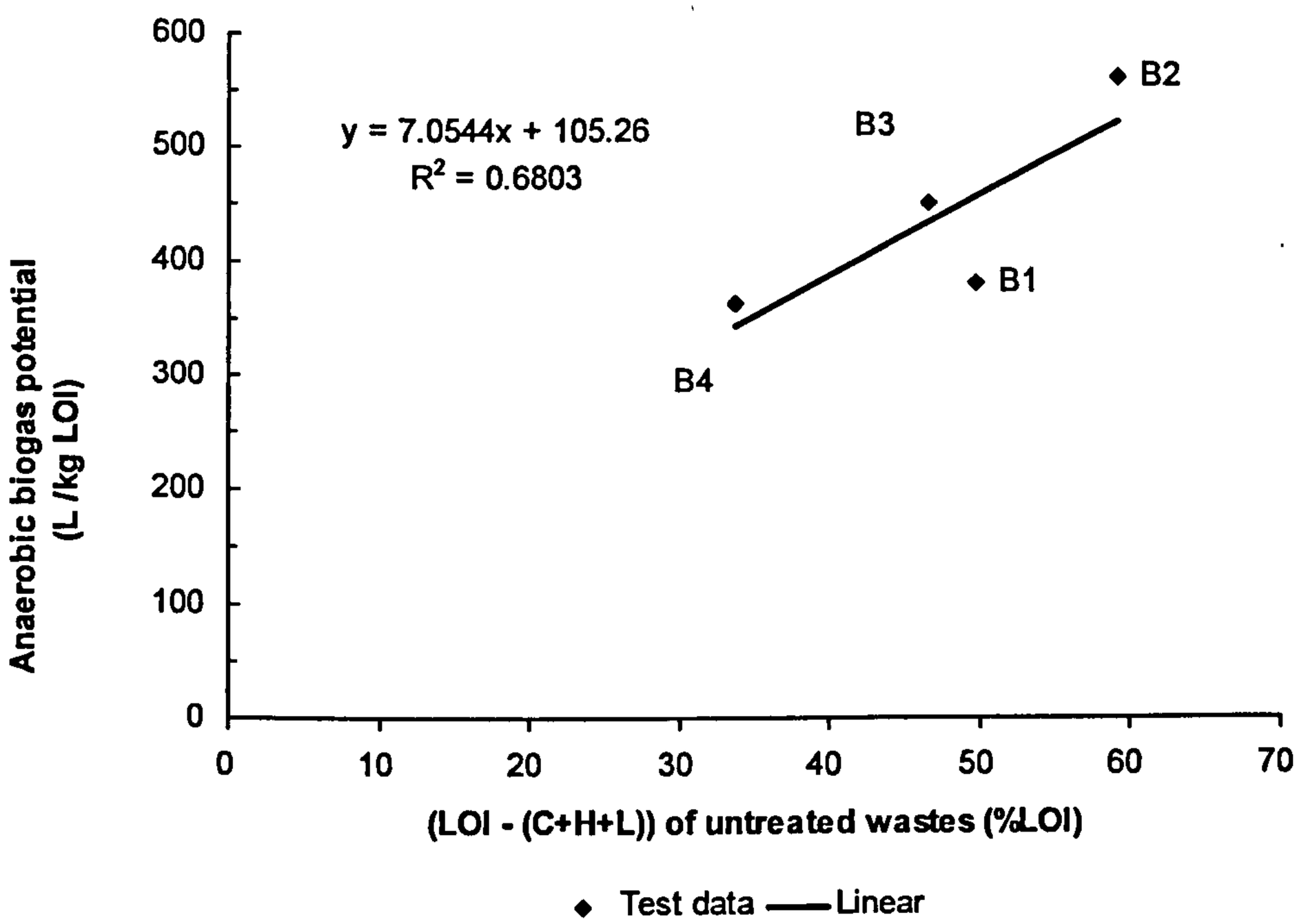


Figure 7.16. Correlation between cell soluble content and biodegradable carbon content for untreated wastes

(C: cellulose; H: hemicellulose; L: lignin)

CHAPTER 8 CONCLUSIONS AND SUGGESTIONS FOR FURTHER STUDY

8.1 Conclusions

The relative contents of cellulose, hemicellulose and lignin, determined by fibre analysis, have been used to assess the degree of decomposition in landfills. In order to identify if fibre analysis can provide a rapid surrogate measurement of biodegradability for the purpose of evaluating the performance of WDAs in meeting targets for diverting BMW from landfills by means of MBT, this chemical method was conducted and compared with the biological tests of BMP and DRI, as well as analytical measurement of TC/TN ratio and LOI content, using a variety of untreated and treated MSW or BMW. The conclusions drawn from this study are summarized.

8.1.1 Evaluation of the Test Methods

- The DRI4 test of measuring the cumulative amount of oxygen consumption per unit weight LOI in BMW over a four-day period was found to provide a useful indication of waste stability. In this study, some evaluation on stability status can be made for the waste samples according to the criteria proposed by Adani *et al.* (2004) for the division of the high and medium biologically stable material.

- The results of DRI4, as well as $DRI_{ave.}$ and $DRI_{max.}$, did not provide the predicted sequence of stability in the order of composting time in each batch. In all four batches of the composting runs, the biodegradation of BMW did

not always reach its maximum degree in 4 days. Lag phases over four days were observed to occur during the DRI tests, a phenomenon which was also reported by Binner (2002) and Godley *et al.* (2005). Additionally, no correlation was identified between DRI4 and $DRI_{tot.}$, which represented the relatively overall biodegradability in this study. It can be concluded that DRI4 indicates the biodegradation rate rather than the overall biodegradability of the test waste material and would underestimate the overall biodegradability in some cases.

- The BMP test has been shown to provide the most direct and reliable way to estimate the reduction in potential biogas production of MBT waste in landfills. The BMW diverted from landfills can be calculated by the percentage of the reduction in potential biogas production. Landfill biogas emissions can be significantly reduced when the BMW is biologically pretreated prior to landfilling. After 28 days of composting, the anaerobic biogas potential was found to be reduced by 70.1 ± 4.3% of that of the untreated waste.

- Fibre analysis using the FibreCap technique provides consistent and reliable measurements of fibre contents in BMW. During aerobic treatment, the results of the chemical tests and gravimetric test ((C+H)/L ratio or C/L ratio, TC/TN ratio and LOI content) were all found to correlate well with those of the biological tests (BMP and DRI) for each batch of waste samples analysed. The TC/TN ratio was found to be a good indicator of aerobic biodegradability as far as BMW was concerned. During anaerobic degradation, the (C+H)/L ratio was also found to correlate well with the anaerobic biogas potential.

- For the untreated wastes studied, it was not found that the higher the cellulose plus hemicellulose content, more biogas production potential.

Additionally, of the untreated wastes studied, the less lignified waste (higher (C+H)/L ratio) was not observed to give higher oxygen consumption or anaerobic biogas potentials. No correlations were established between the (C+H)/L ratios and biodegradability as far as the untreated BMWs were concerned. This is consistent with the findings of the research on the specific waste components (untreated waste) by Eleazer *et al.* (1997) and Godley *et al.* (2005).

- After an initial stabilization or adaptation stage in the aerobic treatment (such as after day 4, 5 or 6), improved correlations were found between the (C+H)/L ratios (or C/L ratios) and oxygen consumption or anaerobic biogas potential than when values of untreated waste samples were included. It can be concluded that for the BMW treated by MBT, the lower the (C+H)/L ratio, the more stable it is. Based on the correlation analysis, two linear models have been proposed to predict the biodegradability based upon the (C+H)/L ratio resulting from fibre analysis for the BMW during or after MBT processes. The limit value of (C+H)/L ratio predicted by the model is in the range of 0.4-0.6, which corresponds to the BMP of 20 L/kg DM and is acceptable for landfilling.

- The C/L ratio obtained by fibre analysis was also found to correlate well with the anaerobic biogas potential for all the treated waste. A linear model is also proposed to predict the biodegradability based upon the C/L ratio for the BMW during or after MBT processes. The limit value of C/L ratio predicted by the model (0.3-0.4) is quite close to that suggested by Komilis and Ham (2003) for indicating maturity for most of MSW type substrates (C/L ratio < 0.5).

8.1.2 Composting Reactors and Operations

In all the composting experiments performed, changes in temperature, oxygen consumption and carbon degradation demonstrated a high degree of consistency related to waste stabilization. The composting apparatus used in this study provided an effective way to determine DRI and monitor the waste degradation during composting.

The operation of sampling, moistening and mixing was found to disturb the composting process, but in cases when moisture content was not optimal or air was not evenly distributed, it accelerated microbial activity. Additionally, before the end of the active phase of composting the waste material continued to generate heat and the temperature climbed back to > 50 °C after each sampling, moistening and/or mixing event. After about 20 days, the active phase of composting processes was observed to end in all the batch experiments.

8.1.3 Composition and Degradation of BMW during Composting

The lignin content of the untreated BMW samples (D0) used in the preliminary study (P) and in the four batch composting experiments (B1-B4) ranged between 5.9% DM (B1 and B2) to 8.4 % DM (P) with an average content of 6.6% DM; cellulose contents displayed a wider range of 20.4 – 48.6 % DM with an average of 33.6% DM; hemicellulose contents were between 7.9-9.7% DM with an average of 8.5% DM.

At the end of the composting experiments, the contents of cellulose and hemicellulose were observed to decrease in terms of both DM and LOI, while the lignin content increased in comparison with wastes that had not undergone aerobic treatment. In all four batches, lignin degradation was also seen, although during limited periods of the composting processes. The production of measured 'lignin' was also detected, which was also observed by Komilis and Ham (2003) and could

be attributed to humic matter, a metabolic product during composting and co-measured in the 'lignin' fraction. The NDF contents of LOI in all the batches also increased after composting, which was concurrent with the increase of lignin contents.

8.2 Suggestions for Further Research

- Since the DRI4 could be misleading for indicating the biodegradability in the cases that lag phases or adaptation phases exceeding four days can occur during DRI determination, the test optimal conditions still needs further investigation. These conditions include factors critical to the aerobic degradation processes, which are moisture content, temperature and the presiding microbial population. At the same time, the test time needs to be extended to over 4 days if the oxygen consumption rate shows a trend of increasing in the last 24 hours of the 4-day test period.

- Although several biological tests had been proposed and applied to evaluate BMW diversion from landfill (such as DRI4, BM100 and GB21), no standard methods or indicators have been formally issued so far. Further investigation is needed to standardize these tests in order to monitor the performance of WDAs in diverting BMW from landfills. At the same time, there have been some critical values established on the stability of treated MSW disposed in landfills in terms of respiration activity and biogas generation, but they differ between studies (Binner, 2002; Soyez and Plickert, 2002; Adani *et al.*, 2004; Godley *et al.*, 2005). Therefore, criteria should be established for the acceptance of the treated waste in landfills in order to help local authorities meet the EU Landfill Directive targets.

- As cellulose, hemicellulose and lignin are less readily biodegradable compounds compared to other organic constituents found in BMW such as sugars, starch, the non-fibre parts (the cell soluble components) may play a significant role in waste biodegradability. Characterization of these more readily biodegradable constituents and understanding their contribution to biogas production may further assist the prediction of the biodegradability for untreated waste, thereby allowing the prediction of the reduction of BMW by MBT.

- As a chemical method, fibre analysis shows advantages in terms of speed and cost of analysis for evaluation of BMW diversion from landfills by measuring the change of lignocellulosic content in the target material during MBT. It is suggested that untreated BMW and treated BMW need to be investigated separately in further research to address the relationships between relative fibre content and biodegradability.

- Before the linear models established in this study can be more fully relied upon, a variety of treated waste samples from operational MBT plants needs to be investigated to validate them. Furthermore, for the purpose of predicting the remaining biogas potential of the waste after MBT, which will be disposed in landfills, the MBT wastes with lower (C+H)/L ratios or C/L ratios (e.g. less than 1) should be the focus of further investigation. This would assist in establishing criteria in terms of the (C+H)/L ratio or C/L ratio which WDAs need to achieve for disposal of MBT waste into landfills.

Appendix A: Calibration of Sensors

1. O_2 – Medical Sensor

The two M-09 O_2 – Medical sensors for detecting the O_2 concentration of the exhaust gas have the features of high signal stability and superior linearity over the entire range. The measurement range is 0 to 100 Vol. %. Response time t_{90} is less than 12 seconds. The drift is less than 1% volume O_2 /month at air. The operation temperature is 0 to 45 °C. The sensors gave mV output signal and the output signal at dry ambient air is 9.5 to 14 mV. They were calibrated at O2 free point and 100% O_2 point by using 100% nitrogen cylinder and 100% oxygen cylinder separately. After calibration, the two sensors gave linear output of 0.603 mV/ Vol. % and 0.598 mV/ Vol. % separately. The calibration results are presented in Table A.1 and Figure A.1.

Table A.1. Outputs of O_2 sensors at known gas concentration

No.	O_2 Vol.%	Output(mV)
1	0	0.05
	100	60.30
2	0	0.01
	100	59.83

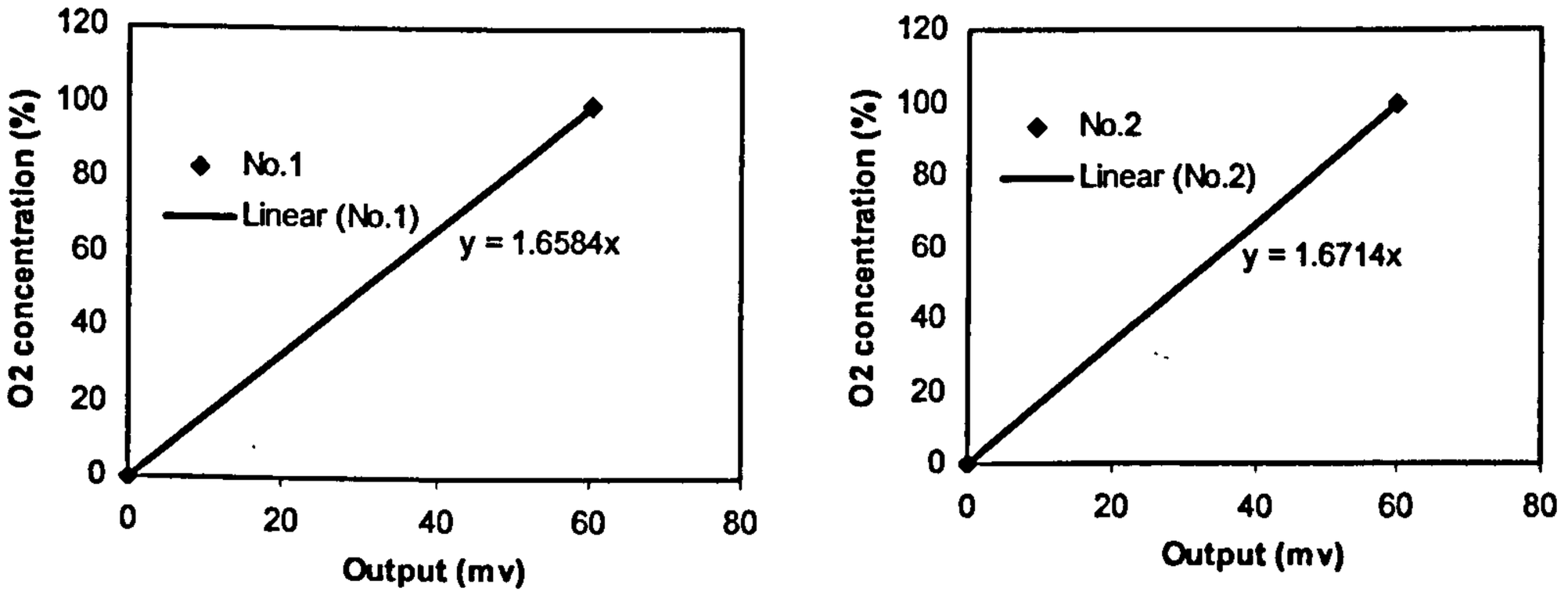


Figure A.1. Calibration lines of two O_2 sensors

2. CO_2 sensor

The two BCPCO_2 probe head CO_2 sensors were supplied by BlueSens Company in Germany. The working temperature range is 15-40 °C. The sensors gave mA output signal from 4mV to 20 mV corresponding to the concentration range of measurement 0-10 Vol. %. During calibration, the CO_2 concentration of ambient air (0.04 Vol. %) was assumed as 0 % to exclude the impact of CO_2 in ambient air so that during composting the amount of CO_2 monitored in the exhaust gas was totally produced by degradation. The sensors were calibrated by using gas with known CO_2 concentrations. The calibration results are presented in Table A. 2 and Figure A. 2. The sensors were 1-Point calibrated at ambient air (0.04 Vol. %) once a month or after each disconnecting of the probe head.

Table A. 2. Outputs of CO_2 sensors at known gas concentration

No.	CO_2 Vol.%	Output (mA)
1	0	4
	10	20
	7.41	14.56
	5.06	11.30
2	0	4
	10	20
	8.02	15.57
	5.06	11.23

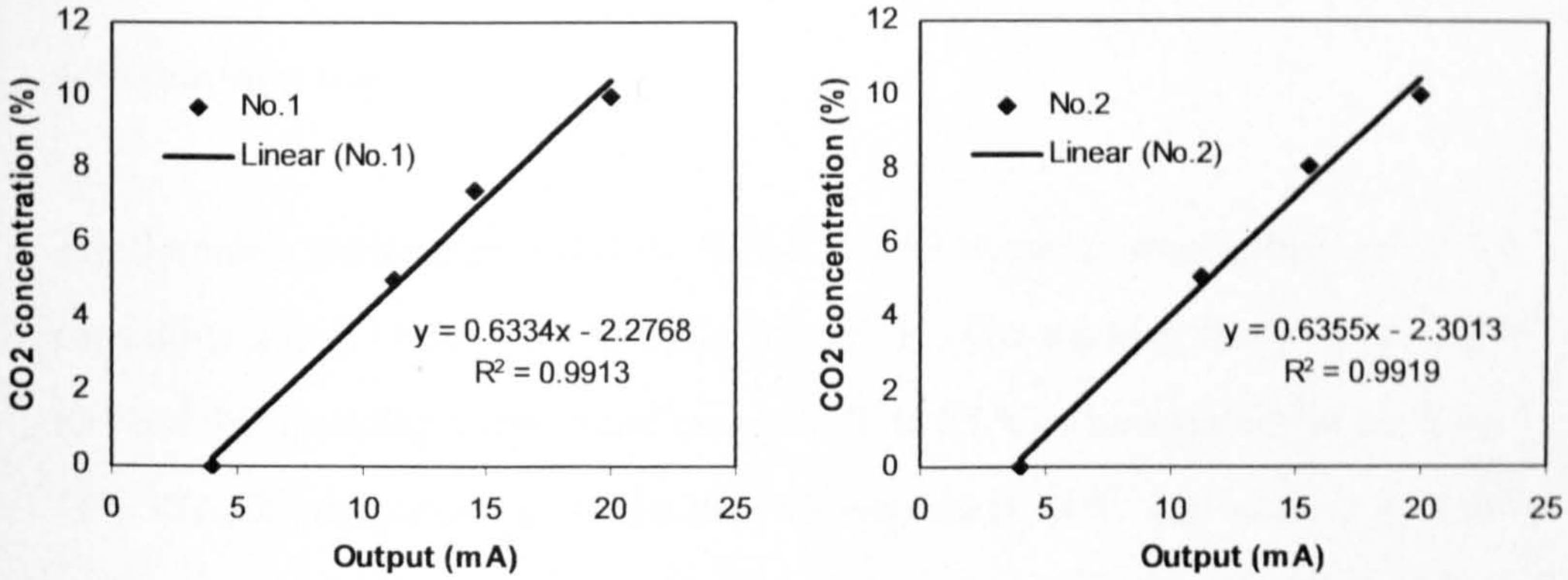


Figure A. 2 Calibration lines of two CO_2 sensors

3. Temperature probe

The Temperature probes used were thermistors, which are semiconductor devices that change their electrical resistance with temperature. They are sensitive but highly nonlinear. The resistance measured at 25 ℃ is 3000Ω. Six thermistors were calibrated in water bath at 12 different temperatures ranging from 24 ℃ to 80 ℃. The calibration relationship is presented in Figure A.3.

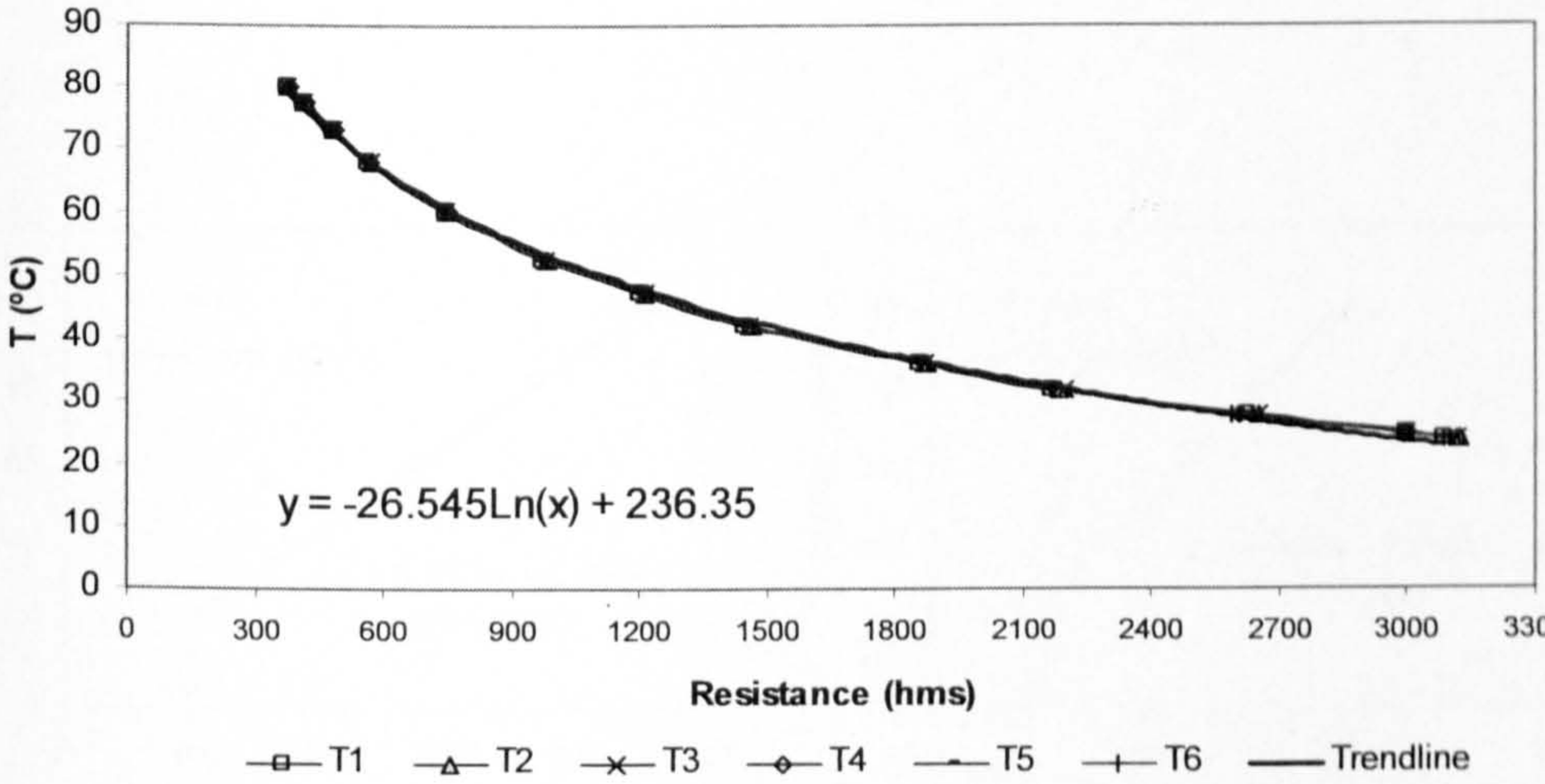


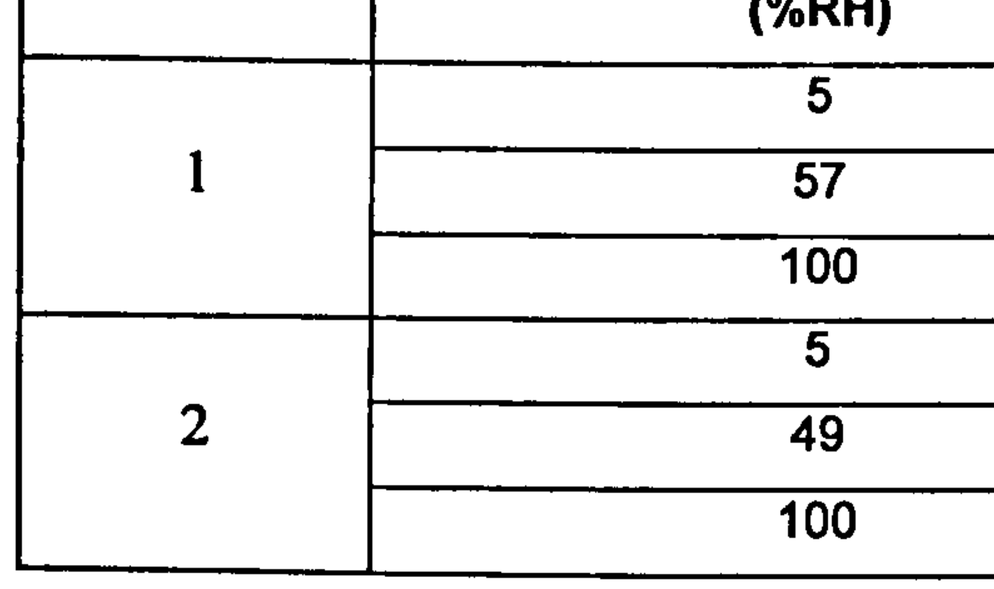
Figure A.3. Relationship of tempreture and resistance of the thermistors

4. Humidity transmitter

The humidity sensor element of the EE0-FTB4A3 humidity transmitter is a capacitive sensor of the HC 103 series from E+E. The working range is 0 to 100% RH and the operating temperature range is -40 to 85 °C. The accuracy at 25 °C is ±5% RH. The response time is less than 45 seconds at 25 °C. The sensors gave mV output signal which were linear with humidity output. A rough calibration of the sensors was made by using a HygroPalm Portable Humidity Temperature Indicator which can indicate humidity directly. The calibration results are presented in Table A.3 and Figure A.4.

Table A.3 Outputs of humidity sensors at known humidity

No.	Humidity (%RH)	Output (mV)
1	5	198
	57	312
	100	941
2	5	216
	49	297
	100	986



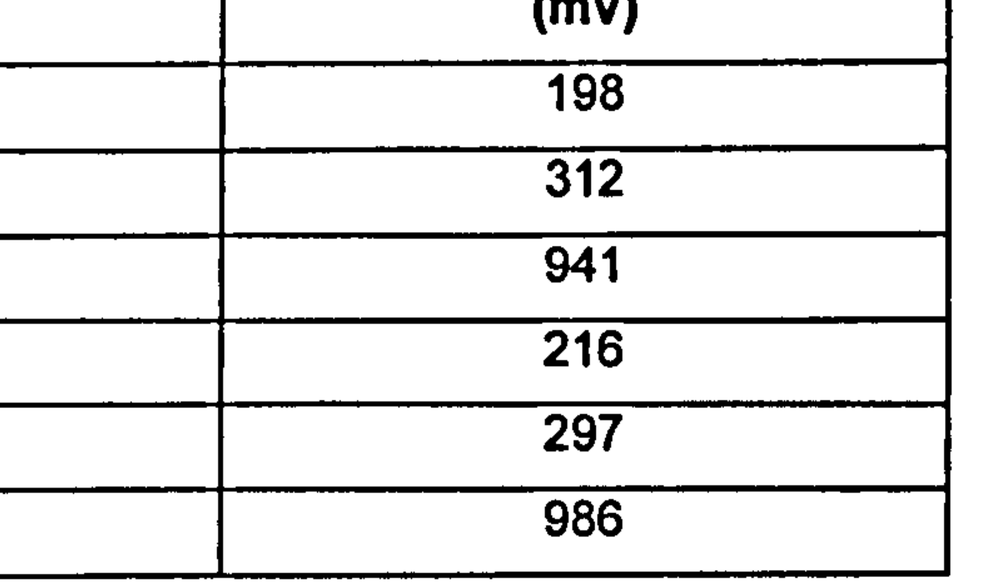


Figure A. 4 Calibration lines of two humidity sensors

Appendix B: Results of DRI Expressed in Four Different Ways

Sample		$DRI_{ave.}$ (mg O_2 kg^{-1} LOI h^{-1})	$DRI_{max.}$ (mg O_2 kg^{-1} LOI h^{-1})	DRI4 (mg O_2 kg^{-1} LOI 96 h^{-1})	$DRI_{tot.}$ (mg O_2 kg^{-1} LOI)
B1R1	D0	2460	3532	165260	1323861
	D5	3960	4740	289854	1099565
	D10	3746	5999	246978	768088
	D20	435	861	32910	352601
	D35	261	342	21656	279517
	D68	430	514	32380	146618
B1R2	D0	2516	3951	178580	1237375
	D5	4245	5014	316371	998415
	D10	3770	5677	255997	635105
	D20	469	775	27442	287740
	D35	279	305	20926	242575
	D68	385	460	26287	108255
B2R1	D0	3037	3664	224726	966421
	D6	4388	5528	303732	601950
	D12	2056	2641	157940	207814
	D20	204	296	6162	18997
B2R2	D0	3116	3805	225499	863419
	D6	3667	4007	225805	497785
	D12	1715	2336	120215	221883
	D20	380	437	29375	61147
B3R1	D0	2875	3788	225328	716907
	D4	2725	3235	153780	491579
	D8	958	1162	41920	337800
	D17	1480	1631	78547	195377
B3R2	D0	4031	4663	276533	990170
	D4	3275	4668	263634	713637
	D8	736	1699	74201	450004
	D17	1714	2151	97146	251876

Sample		$DRI_{ave.}$ (mg O_2 kg^{-1} LOI h^{-1})	$DRI_{max.}$ (mg O_2 kg^{-1} LOI h^{-1})	DRI4 (mg O_2 kg^{-1} LOI 96 h^{-1})	$DRI_{tot.}$ (mg O_2 kg^{-1} LOI)
B4R1	D0	2898	3301	228525	859504
	D4	3340	4262	294480	630979
	D8	1743	2244	86687	336499
	D16	967	1252	85872	172857
B4R2	D0	3272	3839	252001	719244
	D4	3682	4364	267857	467243
	D8	2413	2598	122375	199386
	D16	243	409	19732	50548

$DRI_{ave.}$- Average value of DRI_i taken during the 24 hours of the most intense biological activity in 4 days from the start (for D0 samples) or from the previous sampling event;

$DRI_{max.}$-Maximum value in 4 days from the start (for D0 samples) or from the previous sampling event;

DRI4- Cumulative value in 4 days from the start (for D0 samples) or from the previous sampling event;

$DRI_{tot.}$-Total respiration index, calculated as the sum of DRI_i from the start (for D0 samples) or from the previous sampling event to the very end of the composting experiment.

In B1, when assuming the composting lasted 28 days in order to be in agreement with the composting duration of the other three batches (B1-B3), the values of $DRI_{tot.}$ were different and calculated as following:

$$DRI_{tot.\ (D1\text{-}D28)} = DRI_{tot.\ (D1\text{-}D104)} - DRI_{tot.\ (D29\text{-}D35)} - DRI_{tot.\ (D36\text{-}D104)}$$

$$= \frac{1323.9+1237.4}{2} - 14.9 - \frac{279.5+242.6}{2} = 1004.7\, gO_2 kg^{-1} LOI$$

Appendix C: Changes in Waste Composition during the Composting

Table C.1. Batch 1

Time	Sample	Total w. (g)	Moisture (%)	Dry w. (g, 45°C)	pH	C (%)	N (%)	LOI (%DM)
D0	D0	6500.0	59.7	2965.3	6.9	40.03	1.07	85.3
R1								
Before	D5	5060.8	51.2	2525.3		39.78	1.23	83.9
D5	Sampling	348.2	51.2	173.7	5.5	39.78	1.23	83.9
After	D5	4759.6	51.7	-	-	-	-	83.9
Before	D10	3250.5	46.2	1885.3	-	37.95	1.55	80.5
D10	Sampling	235.0	46.2	136.3	6.3	37.95	1.55	80.5
After	D10	3297.5	50.8	-	-	-	-	80.5
Before	D20	1799.3	28.2	1349.4	-	34.81	2.12	69.3
D20	Sampling	167.3	28.2	125.5	7.8	34.81	2.12	69.3
After	D20	2132.0	45.0	-	-	-	-	69.3
Before	D35	1546.9	35.0	1105.7	-	33.37	2.43	67.2
D35	Sampling	180.2	35.0	128.8	8.4	33.37	2.43	67.2
After	D35	1666.7	46.7	-	-	-	-	67.2
Before	D68	1228.7	34.3	887.0	-	31.72	2.50	63.0
D68	Sampling	242.5	34.3	175.1	8.9	31.72	2.50	63.0
After	D68	1286.2	49.6	-	-	-	-	63.0
End	D104	1032.6	43.7	607.2	8.9	28.58	2.32	57.8
R2								
Before	D5	5459.6	54.5	2510.9	-	39.89	1.14	82.3
D5	Sampling	374.4	54.5	172.2	5.8	39.89	1.14	82.3
After	D5	5138.2	55.0	-	-	-	-	82.3
Before	D10	3171.3	44.3	1886.6	-	37.23	1.44	79.6
D10	Sampling	227.6	44.3	135.4	6.7	37.23	1.44	79.6
After	D10	3193.7	48.6	-	-	-	-	79.6
Before	D20	2467.0	42.5	1480.5	-	34.91	2.09	74.7
D20	Sampling	188.0	42.5	112.8	8.0	34.91	2.09	74.7
After	D20	2679.0	51.1	-	-	-	-	74.7
Before	D35	1956.4	44.0	1206.1	-	33.13	2.52	64.6
D35	Sampling	208.2	44.0	128.4	8.7	33.13	2.52	64.6
After	D35	1848.2	47.0	-	-	-	-	64.6

Time	Sample	Total w. (g)	Moisture (%)	Dry w. (g, 45°C)	pH	C (%)	N (%)	LOI (%DM)
Before	D68	1678.4	46.8	970.1	-	30.88	2.28	62.6
D68	Sampling	265.4	46.8	153.4	9.1	30.88	2.28	62.6
After	D68	1553.0	51.6	-	-	-	-	62.6
End	D104	1270.4	47.1	702.5	8.9	30.57	2.27	58.4

Before-Before sampling; After- After mixing, sampling and adding water; R1-Reactor 1; R2- Reactor 2; The moisture after sampling was calculated by the water added and total solids left.

Table C.2. Batch 2

Time	Sample	Total w. (g)	Moisture (%)	Dry w. (g, 45°C)	C (%)	N (%)	LOI (%DM)
D0	D0	6250.0	63.1	2515.6	38.97±2.17	1.62±0.09	83.9
R1							
Before	D6	4492.7	64.8	1705.0	36.23±0.38	1.51±0.03	73.0
D6	sampling	467.2	64.8	177.3	36.23	1.51	73.0
After	D6	4225.5	66.4	-	-	-	73.0
Before	D12	2863.4	60.1	1309.7	35.31±0.12	2.02±0.04	69.6
D12	Sampling	456.7	60.1	208.9	35.31	2.02	69.6
After	D12	2760.7	65.2	-	-	-	69.6
Before	D20	2302.6	59.3	1003.7	35.00±0.57	2.15±0.02	64.1
D20	Sampling	587.0	59.3	255.9	35	2.15	64.1
After	D20	1715.6	59.3	-	-	-	64.1
End	D28	1649.7	59.8	673.6	33.71±0.33	2.44±0.13	60.0
R2							
Before	D6	4509.3	64.2	1778.5	37.84±0.56	1.53±0.04	72.1
D6	sampling	470.2	64.2	185.5	37.84	1.53	72.1
After	D6	4239.1	65.8	-	-	-	72.1
Before	D12	3670.0	67.8	1499.2	35.43±0.98	1.88±0.13	71.1
D12	Sampling	505.5	67.8	206.5	35.43	1.88	71.1
After	D12	3264.5	68.8	-	-	-	71.1
Before	D20	2899.5	65.9	1060.6	34.62±1.59	2.07±0.09	66.8
D20	Sampling	648.6	65.9	237.3	34.62	2.07	66.8
After	D20	2250.9	65.9	-	-	-	66.8
End	D28	2126.4	64.3	794.4	32.53±0.39	2.21±0.06	63.9

Before-Before sampling; After- After mixing, sampling and adding water; R1-Reactor 1; R2- Reactor 2; The moisture after sampling was calculated by the water added and total solids left.

Table C.3. Batch 3

Time	Sample	Total w. (g)	Moisture (%)	Dry w. (g, 45°C)	C (%)	N (%)	LOI (%DM)
D0	D0	6200.0	70.1	1940.6	38.67±0.44	1.64±0.02	84.1
R1							
Before	D4	5204.4	69.0	1749.2	36.36±0.39	1.45±0.01	79.3
D4	sampling	506.6	69.0	170.3	36.36	1.45	79.3
After	D4	4697.8	69.0	-	-	-	79.3
Before	D8	3965.9	69.4	1240.1	37.50±0.48	1.87±0.07	75.8
D8	Sampling	504.9	69.4	157.9	37.50	1.87	75.8
After	D8	3461.0	69.4	-	-	-	
Before	D17	2981.7	68.2	1033.2	35.93±0.46	2.00±0.05	74.1
D17	Sampling	541.5	68.2	187.6	35.93	2.00	74.1
After	D17	2489.2	68.8	-	-	-	
End	D28	1910.8	64.0	758.4	36.16±0.30	2.20±0.05	70.5
R2							
Before	D4	5032.5	68.2	1688.9	35.57±0.78	1.88±0.07	76.5
D4	sampling	491.7	68.2	165.0	35.57	1.88	76.5
After	D4	4640.8	68.9	-	-	-	76.5
Before	D8	3557.9	67.2	1191.5	37.50±0.85	1.99±0.06	75.8
D8	Sampling	519.9	67.2	174.1	37.50	1.99	75.8
After	D8	3138.0	68.2	-	-	-	75.8
Before	D17	2555.3	63.8	936.5	35.22±0.31	2.20±0.05	70.0
D17	Sampling	514.6	63.8	188.6	35.22	2.20	70.0
After	D17	2091.7	64.7	-	-	-	70.0
End	D28	1699.7	62.1	657.3	34.67±0.55	2.34±0.11	66.2

Before-Before sampling; After- After mixing, sampling and adding water; R1-Reactor 1; R2- Reactor 2;
The moisture after sampling was calculated by the water added and total solids left.

Table C.4 Batch 4

Time	Sample	Total w. (g)	Moisture (%)	Dry w. (g, 45°C)	C (%)	N (%)	LOI (%DM)
D0	D0	6300.0	73.3	1793.6	41.02±0.65	1.48±0.07	86.1
R1							
Before	D4	5410.2	71.3	1740.5	39.91±0.30	1.85±0.08	83.8
D4	sampling	507.7	71.3	163.3	39.91	1.85±	83.8
After	D4	4902.5	71.3	-	-	-	83.8
Before	D8	3889.1	67.8	1304.4	39.11±0.88	2.18±0.01	80.7
D8	Sampling	583.0	67.8	195.5	39.11±	2.18	80.7
After	D8	3406.1	68.7	-	-	-	80.7
Before	D16	2852.5	68.2	911.1	37.13±0.3	2.02±0.08	77.3
D16	Sampling	514.5	68.2	164.3	37.13	2.02	77.3
After	D16	2438.0	69.5	-	-	-	77.3
End	D28	2261.2	72.1	696.9	35.6±1.07	2.18±0.08	75.2
R2							
Before	D4	5230.2	70.1	1746.9	40.81±0.73	1.74±0.04	85.3
D4	sampling	503.6	70.1	168.2	40.81	1.74	85.3
After	D4	4776.6	70.5	-	-	-	85.3
Before	D8	3604.8	66.8	1221.7	39.14±0.66	1.75±0.08	81.2
D8	Sampling	546.0	66.8	185.0	39.14	1.75	81.2
After	D8	3199.8	68.3	-	-	-	81.2
Before	D16	2305.4	65.9	832.0	38.61±0.45	1.87±0.09	80.8
D16	Sampling	511.4	65.9	184.6	38.61	1.87	80.8
After	D16	2094.0	70.8	-	-	-	80.8
End	D28	1998.5	72.0	622.7	35.8±0.35	2.33±0.06	76.6

Before-Before sampling; After- After mixing, sampling and adding water; R1-Reactor 1; R2- Reactor 2; The moisture after sampling was calculated by the water added and total solids left.

Calculation of biodegradable carbon – Example:

Biodegradable carbon content of B1R1D0 sample:

$$\sum_{1}^{104}\frac{C_{CO_2}}{LOI}=\sum_{1}^{5}\frac{C_{CO_2 1}}{LOI_1}+\sum_{6}^{10}\frac{C_{CO_2 6}}{LOI_6}+\sum_{11}^{20}\frac{C_{CO_2 11}}{LOI_{11}}+\sum_{21}^{35}\frac{C_{CO_2 21}}{LOI_{21}}+\sum_{36}^{68}\frac{C_{CO_2 36}}{LOI_{36}}+\sum_{69}^{104}\frac{C_{CO_2 69}}{LOI_{69}}$$

$$=\frac{152.4}{2.234}+\frac{187.7}{1.928}+\frac{160.2}{1.307}+\frac{10.4}{0.813}+\frac{2.9}{0.598}+\frac{2.0}{0.336}$$

=311.8 g carbon/kg LOI

Appendix D: Data of BMP Test

Table D.1. BMP test for B1

Mass of test samples used in BMP assays (g, wet weight):							
Sample	**D0**	**D5**	**D10**	**D20**	**D35**	**D68**	**D104**
R1	40.83	34.16	32.30	28.15	29.93	33.82	43.03
R2		37.38	36.86	33.33	39.36	42.06	45.28
Biogas production (ml/g LOI):							
Sample	**D0**	**D5**	**D10**	**D20**	**D35**	**D68**	**D104**
R1	381.3	392.8	312.6	178.8	86.6	49.6	32.2
R2		348.4	276.5	166.4	99.4	55.0	36.1

Table D.2. BMP test for B2-B4

Mass of test samples used in BMP assays (g, wet weight):						
B2	**Sample**	**D0**	**D6**	**D12**	**D20**	**D28**
	R1	48.12	54.43	50.37	53.73	58.02
	R2		51.19	61.08	61.37	61.38
B3	Sample	**D0**	**D4**	**D8**	**D17**	**D28**
	R1	55.64	57.02	60.38	59.43	55.19
	R2		57.51	56.30	55.29	55.91
B4	Sample	**D0**	**D4**	**D8**	**D17**	**D28**
	R1	61.46	58.15	53.81	56.75	66.64
	R2		54.96	51.92	44.45	65.19
Biogas production (ml/g LOI):						
B2	**Sample**	**D0**	**D6**	**D12**	**D20**	**D28**
	R1	561.1	291.2	252.3	126.4	153.2
	R2		290.3	277.2	168.8	150.5
B3	Sample	**D0**	**D4**	**D8**	**D17**	**D28**
	R1	450.0	366.9	221.7	197.5	156.4
	R2		345.1	277.1	168.8	157.3
B4	Sample	**D0**	**D4**	**D8**	**D17**	**D28**
	R1	362.7	337.5	221.4	111.4	92.8
	R2		312.5	155.8	120.0	109.1

Appendix E: Examples of Lignocelluloses Reduction

In B1R1, the lignin reduction during day 21 (20 days composted, after sampling) to day 35 (before sampling) was calculated as:

$$\text{Lignin rdc}_{(\text{D21-D35})}\ (\%) = \frac{Lignin_{D20} - Lignin_{D35}}{Lignin_{D20}} \times 100\%$$

$$= \frac{138.4 - 152.5}{138.4} \times 100\% = -10.2\%$$

The total lignin reduction from the start (day 0) to the end (day 104) of composting was calculated as:

$$\text{Lignin rdc}\ (\%) = \frac{Lignin_{D0} - \sum Sampled\ \ Lignin - Lignin_{D104}}{Lignin_{D0}} \times 100\%$$

$$= \frac{155.3 - 83.3 - 100.9}{155.3} \times 100\% = -18.6\%$$

Appendix F: Summary of Results

Sample		(C+H)/L	C/L	Biogas potential (L /kg LOI)	$DRI_{ave.}$ (mg O_2 kg^{-1} LOI h^{-1})	$DRI_{max.}$ (mg O_2 kg^{-1} LOI h^{-1})	DRI4 (g O_2 kg^{-1} LOI 96 h^{-1})	$DRI_{tot.}$ (g O_2 kg^{-1} LOI)	Biodegradable carbon content (g carbon / kg LOI)	LOI (% DM)	TC/TN
B1	D0	6.2	4.8	381.3	2488	3742	171.920	1280.618	288.4	85.3	37.4
	D5	5.3	4.4	370.6	4102	4877	303.113	1048.99	219.1	83.1	33.7
	D10	3.2	2.7	294.6	3758	5838	251.487	701.597	117.8	80.1	25.2
	D20	2.3	1.9	172.6	452	818	30.176	320.171	17.4	72.0	16.6
	D35	1.6	1.1	93.0	270	323	21.291	261.046	9.0	65.9	13.4
	D68	1.2	0.9	52.3	407	487	29.334	127.437	4.6	62.8	13.1
	D104	1.0	0.8	34.1	-	-	-	-	-	58.1	12.9
B2	D0	4.8	3.4	561.1	3077	3734	225.112	914.92	306.0	83.9	24.1
	D6	2.4	2.0	290.7	4027	4767	264.768	549.868	175.5	72.6	24.4
	D12	2.0	1.7	264.7	1885	2488	139.078	214.849	68.2	70.4	18.2
	D20	1.2	1.0	147.6	292	367	17.769	40.072	10.3	65.5	16.5
	D28	1.0	0.8	151.9	-	-	-	-	-	62.0	14.3
B3	D0	5.8	4.4	450.0	3453	4226	250.931	853.539	298.0	84.2	23.9
	D4	3.2	2.9	356.0	3000	3951	208.707	602.608	207.4	77.9	22.0
	D8	2.8	2.4	249.4	847	1430	58.06	393.902	129.2	75.8	19.4
	D17	2.0	1.7	183.1	1597	1891	87.846	223.627	70.1	72.1	17.0
	D28	1.5	1.2	156.8	-	-	-	-	-	68.3	15.6
B4	D0	8.1	6.6	362.7	3085	3570	240.263	789.374	241.9	86.1	27.7
	D4	3.6	2.8	325.0	3511	4313	281.169	549.111	156.9	84.6	22.5
	D8	2.6	2.1	188.6	2078	2421	104.531	267.943	65.0	81.0	20.7
	D16	2.1	1.6	115.7	605	831	52.802	111.703	25.6	79.0	19.5
	D28	1.7	1.4	100.9	-	-	-	-	-	75.9	15.9

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