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DOCTOR OF PHILOSOPHY

Evaluating the use of municipal solid waste as a feedstock for in-vessel composting

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Evaluating the use of Municipal Solid Waste as a feedstock for In-Vessel Composting

A thesis submitted to the University of Wales by Richard Furniss in candidature
for the degree of Philosophiae Doctor

School of the Environment, Natural Resources and Geography

Bangor University

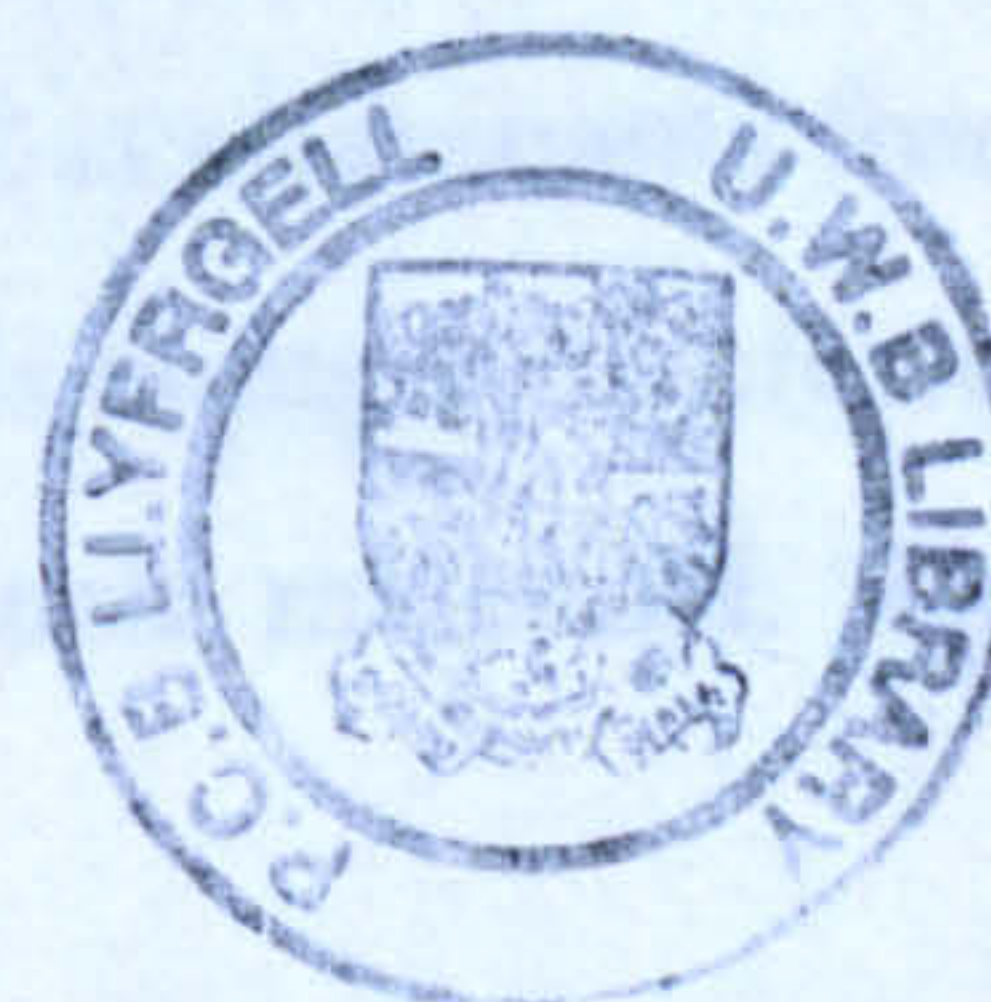
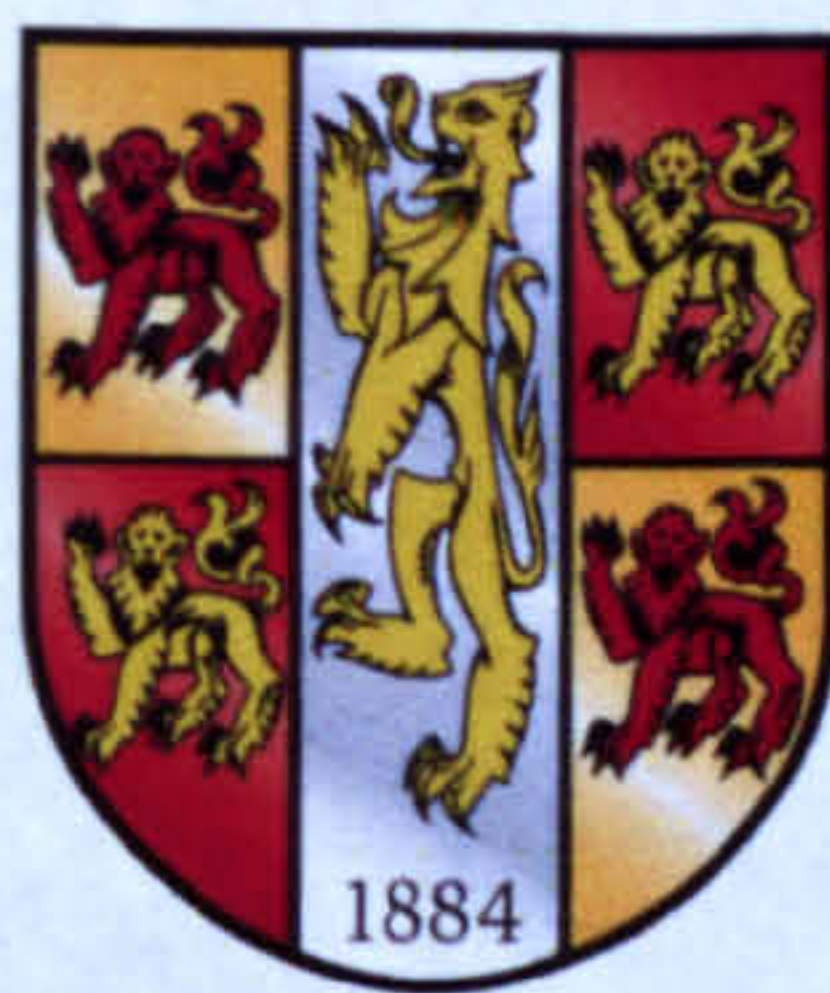
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Chapter 2. Maps. pp.84, 87

Chapter 3. Maps. pp. 150

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Summary

For centuries, the dominant form of waste disposal has been landfill, however, this is now deemed to be non-viable from a social, legislative, environmental and economic perspective. Therefore, more sustainable methods of waste disposal are needed globally to reduce the amount of waste going to landfill (e.g. composting). This thesis considered two different forms of in-vessel composting and their effectiveness in treating the biodegradable fraction of Municipal Solid Waste (MSW) that had been mechanically separated prior to composting in two areas of Wales. Further to this, the potential for using MSW-based composts as growth media was investigated alongside the physical composition of MSW-based composts.

This thesis found that primary-screened MSW contained a high amount (>50%) of biodegradable matter with varying amounts of glass (ca. 23%), plastic (ca. 10%) and other components present. Both in-vessel composting methods were successful at treating MSW and produced composting profiles similar to those for green waste (e.g. increased pH, reduced EC and increased NO_3^- content). Alongside this, both methods also reached temperatures high enough to cause destruction of human enteric pathogens making them compliant with the 2005 Animal By-Products Regulations. Inert matter (i.e. glass and plastic) remained unchanged during composting, but beneficially enhanced composting by improving aeration. In contrast, small bench-scale composting units failed to realistically mimic the performance of the larger scale MSW composting systems; the results obtained with them suggested that commercial composting additives did not enhance the rate of composting. The use of the MSW-based growth media showed that as the materials matured, yields of two common pasture land species increased significantly, and in some cases exceeded those of commercially available growth media used widely in horticulture. Combining agricultural soils with these MSW-based materials gave increased plant yields in comparison to soil alone, however, the benefits were dependent on the amount and type of MSW compost added. Further research is needed into the removal of physical and chemical contaminants from the parent materials for composting of MSW based materials along with clearer legislative definitions as to the possible end uses for such materials. The

presence of clear markets has the potential to drive improvements in the technology used for MSW composting. In conclusion, composting provides a viable method for the treatment of MSW-derived biodegradable waste and has the potential to form an important component of sustainable waste management in the UK.

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To everyone else who has helped in whatever way they have. Thank you for your help, you know who you are!

Ac yn olaf, fy nghariad i, Delyth. Diolch am bob peth Calon. Dw i'n edrych ymlaen at y dyfodol efo chdi, nawr bod hwn wedi gorffen. Dw i'n dy garu di am byth.

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Abbreviations

°C – Degrees Celsius

Al – Aluminium

C – Carbon

C₂H₄ – Ethane

Ca – Calcium

CaCO₃ – Calcium Carbonate

Cd – Cadmium

CH₄ – Methane

cm – Centimetre

cm³ – Cubic centimetre

CO₂ – Carbon Dioxide

Cr – Chromium

Cu – Copper

d – Days

DON – Dissolved Organic Nitrogen

DOC – Dissolved Organic Carbon

DN – Dissolved Nitrogen

EC – Electrical Conductivity

Fe – Iron

g – Gram

H – Hydrogen

h – Hour

ha – Hectare

K – Potassium

KCl – Potassium Chloride

kg – Kilogram

l – Litre

M – Molar

m – Metre

MBT – Mechanical Biological Treatment

min – Minutes
mg – Milligram
mM – Millimolar
mm – Millimetre
Mo – Molybdenum
Mn – Manganese
mS – Millisiemens
MSW – Municipal Solid Waste
N – Nitrogen
Na – Sodium
NaHCO₃ – Sodium Hydrogen Carbonate
NH₃ – Ammonia
NH₄⁺ - Ammonium
Ni – Nickel
NO₂⁻ - Nitrite
NO₃⁻ - Nitrate
O₂ – Oxygen
OM – Organic Matter
P – Phosphorous
Pb – Lead
PO₄³⁻ - Phosphate
Rev – Revolutions
t – Tonne
TOC – Total Organic Carbon
TON – Total Organic Nitrogen
w – Weight
wk – Weeks
v – Volume
Zn – Zinc
yr – Year
μS – Microsiemens

Introduction and Literature Review

1.1 Introduction

1.1.1 General introduction and need for research

As the rate of waste production has increased in line with increasing human populations, there has been a greater necessity to switch from unsustainable disposal methods such as landfill and move towards recycling waste (De Bertoldi *et al.*, 1983); especially as this reduces the potential for environmental pollution due to accumulation of waste in one place (Villar *et al.*, 1993; Koufodimos and Samaras, 2002). The shortage of “dumping grounds” for waste, related to the unsustainable nature of waste disposal, has further enhanced the need for alternative forms of disposal to be identified (Van Assche and Uyttenbroeck, 1982). Coupling this with stringent legislative controls on both landfill and incineration has lead many nations to look towards sustainable (and cost-effective) waste management schemes and policies (Bari and Koenig, 2001). Resource recovery is firmly fixed as an important part of waste management strategy and plays a significant role in the production of waste policy across the world (Koufodimos and Samaras, 2002). This can take the form of separating waste at source, e.g. placing different materials into different receptacles, to more complex systems involving multiple mechanical sorting units.

As public awareness of environmental issues has increased, there has been an increase in waste management legislation attempting to drive the move from the “throw-away” culture and increase the use of more sustainable methods of waste disposal. To successfully identify what type of processes can be used for managing waste, it is vital to know the quantity and type of material(s) being produced and how much of that waste is useful in terms of capturing/diversion from such endeavours as landfilling (Gidakos *et al.*, 2006).

Landfill is currently the dominant method of waste disposal in many nations, but this is seen as unsustainable and has the potential to cause harm to the environment; due to amount of CH₄ produced from the decomposition of biodegradable matter within landfill, the reduction of the amount of biodegradable materials being sent to landfill is seen as highly desirable. Within the UK, municipal waste contains a high amount of biodegradable material (approx 60%), but also contains much inert

matter (e.g. glass/plastic) that can be recycled. Such a high biodegradable content has meant that there has been increasing focus upon utilising technologies that concentrate on the decomposition of this material e.g. composting.

Commercial composts have value as plant growth media and with the amount of biodegradable matter, and hence potential nutrient content, within municipal wastes, there is scope for use of composted wastes as growth media. However, the presence of inert matter could hinder such an end use on a commercial scale.

As such, there is a need to understand and evaluate the chemical and physical composition of household waste pre and post composting to determine what the material can be used for.

1.1.2 Plan of thesis

This thesis undertakes the evaluation of two in-vessel composting procedures in terms of the chemical changes occurring during the composting process and the subsequent maturation period; this was expanded to include co-composting of the municipal waste material with green waste being used as a bulking agent. The second of these trials varied the amount of green waste within the compost feedstock to mimic times where the supply of green waste was reduced (e.g. winter months). Following this, the product of one composting trial is evaluated in terms of its potential for use as a growth medium both alone and when combined with agricultural soils using two common pasture land species.

Due to the variable nature of municipal waste composition, trials were carried out to determine the composition of the materials being produced from one of the sources for the compost feedstock mentioned above; a weight based method was compared to an area based visual method for assessing the composition. This trial also evaluated the changing composition of material as the screen size used was decreased.

To assess the impacts of screening material upon the heating potential of small scale composting units, a trial was conducted using insulated vessels on a benchtop scale. This trial evaluated the temperatures that could be reached using

this method and also the impact of adding solutions containing organic or inorganic nutrients to the material prior to heating.

1.1.3 Aims and Objectives of Thesis

The changing face of waste disposal is seen by many as an important topic both in the public and political sectors and as such, there is a need to evaluate many facets of processes such as composting so that more efficient methods of waste disposal can be implemented to shift towards sustainability. The aim of this PhD studentship was to evaluate the feasibility of using the biodegradable fraction of municipal solid waste as a feedstock for in-vessel composting. Further aspects of this were to investigate the potential for recovery of materials from the MSW Fines stream to increase the economic potential of the material both for composting and recycling. To achieve these aims, the thesis has the following specific aims, an overview of which is given in this introduction and are discussed in further detail in each subsequent chapter:

1. To determine the chemical composition of Municipal Solid Waste (MSW) and Green Waste materials from Local Authority collections in Gwynedd following mechanical treatment (Chapter 2).
2. To evaluate the performance of a Mechanical-Biological Treatment (MBT) process upon MSW based feedstock for in-vessel composting using the EcoPOD® composting method in Gwynedd (Chapter 3).
3. To determine the chemical composition of MSW and Green Waste taken from Local Authority collections delivered to a Materials Recovery Facility (MRF) in Ceredigion (Chapter 4).
4. To evaluate the performance of an MRF based process in producing an MSW-derived feedstock for in-vessel composting using a proven system for green waste compost in Ceredigion and Shropshire (Chapter 4).
5. To evaluate the suitability of MSW-based composted materials as growth media both alone and mixed with agricultural soils for two common pasture land species under greenhouse conditions (Chapter 5).

6. To investigate and identify the physical composition of MSW taken from an MRF using a weight based and an area based method and compare and contrast the two (Chapter 6).
7. To evaluate the potential for recovery of materials from the MRF by changing the size of the screen used (Chapter 6).
8. To investigate the potential for using screened and unscreened MSW fractions in small scale, bench top, heating trials along with the effect of addition of organic and inorganic nutrients upon the heating potential of these materials (Chapter 7).

Literature Review

1.2. Waste management legislation

Due to the worldwide problem of what to do with waste materials combined with a desire to shift towards a more sustainable society, a series of legislative drivers have been produced by the European Union to tackle the problem. These are often complimented by national policies, giving rise to differences in terms of the implementation of such regulations (Koufodimos and Samaras, 2002). Before the EU began producing such documents, member states without centralised administrations tended not to have strict processes for implementing legislation, mainly due to the lack of a need to do so (Koufodimos and Samaras, 2002). The main pieces of legislation that govern municipal waste collection/disposal strategy across the EU are:

- 1975 Waste Framework Directive (75/442/EEC)
- 2006 codified Waste Framework Directive (2006/12/EC) – this replaced 75/442/EEC due to the number of amendments made to the original Waste Framework Directive.
- Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives – this replaced the two previous Waste Framework Directives and implemented the notion

of utilising a form of Waste Hierarchy (see below) as a driver for changes in the way waste is managed.

- 1991 Hazardous Waste Directive (91/689/EEC)
- 1999 Landfill Directive (1999/31/EC)
- 2000 Incineration Directive (2000/76/EC)
- 1994 Packaging and Packaging Waste Directive (94/62/EC) – amended by 2005/20/EC to include the 2004 Accession Nations.

Waste management strategy has undergone vast changes within the last 30 years to keep up with the changing composition of municipal waste; the first piece of legislation to address waste issues on an EU-wide scale was the original EU Waste Framework Directive 75/442/EEC (EU, 1975). Within this piece of legislation, waste was defined as “...any substance or object...which the holder discards or intends or is required to discard...” This placed the onus upon the waste producer to dispose of their waste correctly without endangering public or environmental health in any way; this being one of the first major legislative acknowledgements that waste could pose problems to environmental health (previously human health had been the main focus). The recovery/recycling of waste materials was also encouraged to conserve natural resources. Since this directive was in place for over 30 years, during which the waste management landscape altered considerably, there was a great need for it to be updated due to the number of amendments made; the new EU Directive on Waste (2006/12/EC) was released on 5th April 2006 and featured the amendments made to 75/45/EEC incorporated into the main body of text. This has since been replaced by the Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives; this implements several changes including the notion of utilising a form of Waste Hierarchy (see below) as a driver for changes in the way waste is managed.

With the original Waste Directive (75/442/EEC) there was a large amount of freedom in terms of the interpretation of several definitions due to the clauses in Article 2 (2) excluding certain waste materials from being controlled by this

directive. This resulted in problems associated with disposal of more hazardous materials; therefore further legislation was needed to control the disposal of those substances which posed greater risk to human and environmental health (EU, 1975). To close the loophole regarding hazardous materials, the 1991 Directive on Hazardous Waste (91/689/EEC) was published. The EU Commission built in a clause which allowed any wastes to be added to the list of hazardous substances as and when this was needed; periodic reviews of this list were also implemented (EU, 1991). This directive provided a much needed update of waste management procedures, including the recording and separation of different waste streams; these updates reflected the changes in waste management strategies and operations during the 16 years since the original Waste Framework Directive was published.

The 1994 Packaging and Packaging Waste Directive (94/62/EC) is another piece of legislation that identifies a specific type of waste to deal with. This directive aims to control the production and disposal of any material which is used to package goods for consumer purchase or even at an industrial level by way of recycling in the short term and reduction/reuse as a longer term strategy. The reduction of the amounts of packaging used and development of reuse systems are actively encouraged in Member States and include several targets for reduction/reuse. These targets include:

- Minimum of 60% of packaging waste to be recovered or incinerated with energy recovery by 31st December 2008.
- Minimum of 60% by weight of glass, paper and board; 50% by weight of metals; 22.5% by weight of plastics and 15% by weight of wood.

Ireland, Greece and Portugal were allowed to extend the dates of compliance until 31st December 2011 due to factors such as the rural nature of the nation and low packaging consumption; the 10 expansion nations that joined in 2004 were given until 2012 to implement these targets (EU, 2005). Before the end of 2007, the European Parliament and Council were to set further targets for the period of 2009-

2014; incineration with energy recovery is seen as key to achieving both the short term and long term goals of this directive.

Aside from the original 1975 Waste Directive (and its updated forms from 2006 and 2008), possibly the most important piece of waste-related legislation produced by the EU is the 1999 Directive on the Landfill of Waste (1999/31/EC). This directive laid down the definitions for what constitutes Municipal Waste, Hazardous and Non-Hazardous Waste and Inert Waste along with identifying the different types of landfills for each of these materials; Municipal Waste disposal is allowed in non-hazardous waste landfills, subject to certain conditions. One of the major conditions was to place a requirement on member states that all waste which enters landfill must undergo some kind of pre-treatment process (Article 6(a)); this may take the form of complex thermal/chemical/biochemical processes or simply sorting of the material (Article 2(f)), as long as the characteristics of the waste are altered sufficiently in an attempt to enhance material recovery or volume reduction. Norbu *et al.* (2005) found that pre-treating MSW resulted in a material which was easier to handle, had reduced volume and that recyclables could be separated from with greater efficiency.

Targets were also set for the reduction of levels of biodegradable material (material which can undergo anaerobic or aerobic decomposition such as food waste; EU, 1999) being landfilled within EU member states (Article 5(2)); material must be reduced to certain percentages of 1995 disposal levels by agreed dates. These targets are as follows:

- 2006: 25% reduction in levels of biodegradable waste landfilled in 1995
- 2009: 50% reduction
- 2016: 65% reduction

For the UK, assuming a 3% annual growth rate and that 60% of MSW is biodegradable, the following figures were calculated to give the amount of biodegradable matter that the UK would be permitted send to landfill in each of the

years listed (figures in brackets represent the amount to be diverted from landfill each year):

- 2006 – 13.1 million t (11.0 million t)
- 2009 – 8.7 million t (17.6 million t)
- 2016 – 6.1 million t (26.2 million t).

(Welsh Assembly Government, 2002).

Under the terms of this legislation, each member state was free to set its own specific tonnage targets, above these minimums, for reduction by way of National Waste Strategies (see below); Coker and Murray (1981) state that the general public perceive local authorities to be more suitable for addressing issues relating to legislation in the local area and do so without the time-consuming administrative procedures that tend to occur with larger scale authorities. Therefore, that the EU allowed each member state to set its own targets can be seen as a positive step. Sweden is a good example of setting ambitious targets above those listed in the Landfill Directive, as shown in Svensson *et al.* (2004). This study states that in Sweden, from 2005, no organic waste was allowed to be deposited into landfill under any circumstances. This does raise the issue of the treatment processes to be used, with composting so far being the only suitable method identified due to the lack of experienced biogas plant operators.

One complicating factor in this process is that there is some difference in opinion across the EU regarding the percentage of Municipal Waste that can be called biodegradable. Other problems associated with the meeting of the targets from the EU Landfill Directive (1999/31/EC) include the growth of waste arisings year on year within nations, the lack of stable markets for recycled materials and poor funding for alternative disposal methods to landfill (Price, 2001).

As the limits on amounts of waste being permitted to landfill in the EU began to come into force, there was a need to regulate other disposal technologies such as incineration. The 2000 Directive on Incineration of Waste (2000/76/EC) was published to update the original documents covering new plants (89/369/EEC) and

hazardous waste incinerators (94/67/EC) (now both replaced by 2000/76/EC) and to extend legislative coverage to non-hazardous waste incinerators and to materials not regulated by previous directives. Emissions from incineration plants, minimum temperatures during the process and the solid waste products of incineration (fly and bottom ash) are strictly controlled by this directive; regular testing/monitoring of plants and residues are also required under these rules. An update on the implementation of this directive was due to be delivered to the EU Commission by the end of 2008 (EU, 2000); as of mid-2009, no published information on this has been identified.

One piece of key non-legislative advice given by the EU to member states is the Waste Management Hierarchy (Koufodimos and Samaras, 2002; Gidarakos *et al.*, 2006). The hierarchy is a framework based upon the principle that reduction and minimisation of waste is preferable to firstly, waste production, and secondly, landfill; the alternatives listed within the hierarchy have not been evaluated during its derivation (Gidarakos *et al.*, 2006). The diagram below shows a simplified version of the Waste Hierarchy system (the higher up the pyramid a management option is, the more desirable the option):

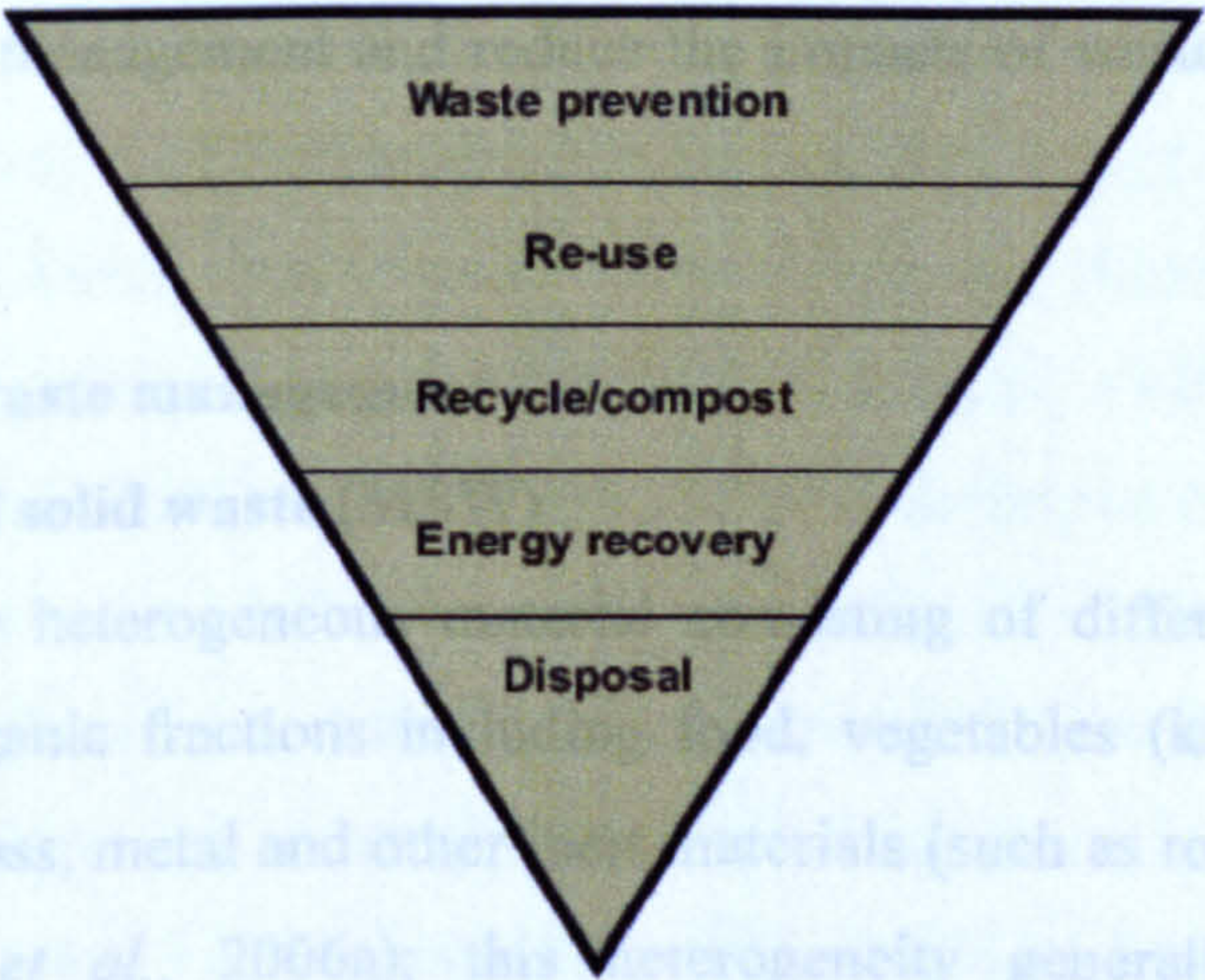


Figure 1.1: The Waste Hierarchy (DEFRA, 2007). With respect to the Waste Hierarchy, it is not actually a legislative instrument derived from evaluation of the alternatives it lists (Gidarakos *et al.*, 2006), but more of an advisory tool around which some legislation has been based and the public instructed how to go about changing their disposal habits. Therefore its use as a

tool for developing legislation is open to some degree of debate. Building upon this non-legislative basis, the 2008 EU Directive on Waste states that EU member states should take measures to implement waste treatment measures in line with the following hierarchy (EU, 2008):

- Prevention (prevention is defined as being measures taken before a substance, material or product has become waste);
- Preparing for reuse;
- Recycling (where recycling is defined as any recovery operation by which waste materials are reprocessed into products, materials or substances whether for the original or other purposes);
- Other recovery, notably energy recovery (recovery being any operation the principal result of which is waste serving a useful purpose);
- Disposal.

Once more, this is not a legislative tool as such; rather the waste hierarchy proposed by the EU is a framework tool around which legislation can be organized to improve waste management and reduce the impacts of waste production on the environment.

1.3. Current waste management

1.3.1. Municipal solid waste (MSW)

MSW is a highly heterogeneous material consisting of differing proportions of organic and inorganic fractions including food, vegetables (kitchen/food waste), paper, plastics, glass, metal and other inert materials (such as rocks) (Abu Qdais *et al.*, 1997; Mor *et al.*, 2006a); this heterogeneity generally results from a combination of factors including socio-economic status and the collection/sorting processes used by different areas. Variation of this nature also extends to the organic fraction, which in itself can alter the potential end uses of the material (Iglesias-Jiménez and Pérez García, 1989). It is a substance which should be seen

as a resource for potential materials recovery (Wong, 1985) rather than a waste product in need of immediate disposal.

Levels of MSW being produced year-on-year appear to be increasing in most nations, regardless of economic development, raising concerns about the viability (economic and environmental) of current disposal practices (Daskalopoulos *et al.*, 1998). Importantly, Price (2001) noted that MSW is not solely composed of household waste, but also contains some commercial waste from civic amenity sites and possibly waste from Local Authority collections of trade and street waste; this further complicates the production of a global definition of MSW due to the differences in such collections across the world. Knowledge of the composition of MSW is of great importance in planning the strategies for managing such waste materials (Bolaane and Ali, 2004).

A vast proportion of the materials contained within MSW can be recycled quite easily, a process which would then increase the lifespan of current landfill sites on the basis that less waste was entering them on a daily basis (Mor *et al.*, 2006a); one problem with this view is that many nations employ collection services that do not actively encourage the separation of the different fractions, allowing organic and inorganic fractions to mix together (Garcia *et al.*, 2005).

To adequately deal with MSW, it is important to recognise that it is a highly heterogeneous material that varies with seasonal, lifestyle, demographic, geographic and legislative impacts (Daskalopoulos *et al.*, 1998; Gidarakos *et al.*, 2006; Hansen *et al.*, 2007). Such a mixture of components and the variation it experiences makes the definition and measurement of waste composition exceedingly difficult and yet, at the same time, essential if the correct type of waste management programme is to be implemented (Prudent *et al.*, 1996; Gidarakos *et al.*, 2006). The average UK householder produced 324 kg per head of household waste in 2007/08, a decrease from 353 kg per head in 2006/07; this equates to a fall of around 80 g of waste per person per day less being produced (DEFRA, 2009). Whilst trends show this decrease, this is still higher than levels from other EU nations around 25 years ago. van Assche and Uyttebroeck (1982) recorded the average Belgian as producing around 0.6-0.7 kg d⁻¹, or around 220-260 kg yr⁻¹.

This shows the vast increase in waste production levels during the last 25 years when the 1982 study is compared to more modern data. Mor *et al.* (2006a) reference a study which found that the residents of Delhi, India, produced between 150 and 600 g of waste per day, depending on economic status; Bolaane and Ali (2004) recorded the average person in Botswana as producing 330 g of waste per day, with the majority (68%) of this being putrescible matter. Such figures highlight the differences between the developed and developing world in terms of waste production and go some way to show how wasteful the Western world has become.

The disparity between waste levels produced in developing and developed nations is partly due to the type of food consumed and the level of packaging produced. Mor *et al.* (2006a) and Abu Qdais *et al.* (1997) also concluded that increasing economic wealth and social status across developing nations such as India and Abu Dhabi has the potential to increase the total amount of MSW produced by up to 35% (Abu Qdais *et al.*, 1997). Das (1988) looked at the MSW composition in another Indian city, Kolkata (formerly Calcutta), and concluded that the presence of high levels of food/vegetable materials in the Kolkatan wastes differed from Western MSW. This was more likely to have larger proportions of plastics, glass and paper due to a reduced use of fresh food compared to India and would suggest the Indian MSW would make a better compost feedstock. Hansen *et al.* (2007) go one stage further, identifying the role of local authorities as changing the composition of collected materials and this often being dependent upon the type of systems (kerbside collection vs. industrial sorting etc) present and the level of access to information for the general public. Clear correlations exist between MSW composition and socio-economic status (Gidarakos *et al.*, 2006) and as consumer habits change over time, so does waste composition; this study found that the proportion of MSW made up of plastics doubled between the late 1990s and 2006, with paper exhibiting a similar trend. This is thought to be caused by the rise in food packaging used by most supermarket chains.

The type of system used to process MSW, and carry out source-sorting, can have a definite effect upon the quality of the material produced, especially the organic

fraction (Hansen *et al.*, 2007). Processing of MSW is mainly classified into one of three categories:

- (1) Density separation – based upon the relative weights of different components (e.g., plastic and paper are lighter than metals and wood);
- (2) Electric and magnetic field separation – e.g., removal of ferrous metals using magnetic separators;
- (3) Size separation – also known as “Screening” (Lau *et al.*, 2005). Screening employs the use of one or more surfaces, often with perforated with different size holes, to remove or allow through different size materials.

Trommel screens are widely used in the screening process; these consist of a large perforated rotating drum above a conveyor belt (if mobile) or collection hoppers (if fixed). Material is loaded at one end and moves through the drum as it rotates, with any particles smaller than the size of the perforations falling through into the collection hoppers or onto the belt. The larger material which passes out of the end of the drum is often then sorted further or sent straight for disposal depending on the position of the trommel screen within a facility (Lau *et al.*, 2005). Such a system reduces the manual input and is useful for processing of materials with high moisture contents and also for the removal of hazards such as glass fragments prior to hand picking lines (in terms of MSW processing). However, the highly variable nature of MSW can cause problems with the use of trommels for processing of domestic wastes such as clogging or issues with glass fragments (Wheeler *et al.*, 1989). Further investigation by Lau *et al.* (2005) discovered that a higher trommel rotation speed reduced the recovery efficiency of the trommel operation, whilst this was also reduced by increased particle size of waste – thought to be caused by blocking of the trommel perforations.

The efficient screening and sorting of MSW can be used to reduce the levels of potential contaminants, such as heavy metals contained within newsprint and plastics, and also physical contamination from metals and glass (Prudent *et al.*, 1996; Hansen *et al.*, 2007). Changing of the screen size of a trommel can lead to an

alteration to the composition of both the oversize and undersize materials produced; for example Wheeler *et al.* (1989) found that an aperture diameter of 200 mm produced an oversize material high in paper, textiles and plastic film but reducing this down to 50 mm changed the composition dramatically. This material consisted mainly of metals, dense plastics and putrescible matter.

The proportions of MSW which are classed as biodegradable material vary greatly throughout the relevant literature and often depend upon how each fraction is classified by the author. For example, Bari and Koenig (2001) quote a figure of 35% for the amount of MSW that is classed as organic in Hong Kong. Hu *et al.* (1998) also analysed Chinese MSW and found it to contain 50.7% organic matter, with a further 8.8% classed as paper; this highlights the differences present within a nation, let alone across the planet. Prudent *et al.* (1996) recorded over 56% of MSW as being biodegradable (paper, cardboard and organic materials), whilst according to Keeling *et al.* (1994) “domestic refuse” contains around 30% kitchen and vegetable waste which can be mechanically separated from the main MSW stream and used as compost feedstock. Gidakos *et al.* (2006) identified 39% of MSW as “Putrescible” (such as food wastes), 20% as paper, 17% as plastics; this amounts to 59% of MSW being classed as organic biodegradable matter with a further 5% consisting of textiles, wood and other more recalcitrant biodegradable materials such as leather. This study also found that the percentage of glass present rose during the summer months to a peak of 18%, a figure which was subsequently attributed to the increase in tourist activity in the local area. Another study based in Greece (Koufodimos and Samaras, 2002) recorded an organic proportion of 41% with a further 23% being classed as paper, giving a total of 64% as biodegradable matter. De Bertoldi *et al.* (1983) analysed MSW and found between 47 and 60% to be organic (biodegradable) with plastics, metals and glass all accounting for between 4 and 8% of the total composition. Lau *et al.* (2005) recorded a value of around 60% of MSW being paper, with a further 20-25% being classed as organic, meaning that over 80% of the waste being classed as biodegradable; this is slightly higher than the figures recorded by most studies for the percentage of MSW that is classed as biodegradable. In comparison to these studies, Kirchmann and Widén

(1994) characterised Swedish MSW and found that only 20-30% was classed as organic, a much lower figure than other studies. UK MSW contains around 50% biodegradable matter (paper/card (33.2%) & putrescibles (20.2%)) according to Sesay *et al.* (1998) whilst Bench *et al.* (2005) also quote two studies that give figures of between 17 and 22% for the putrescible content of UK MSW; no figures for the paper/card content are listed in the latter study.

Gidakos *et al.* (2006) found that virtually all of the fractions found within MSW vary with seasonality and that due to the different activities within the local area during the year even more variation was occurring. In fact, there was much variation between some of the sampling sites due to activities within the vicinity of each site such as greenhouse-based food production which gave rise to a large amount of plastic waste. Gidakos *et al.* (2006) goes on to conclude that there has been a large shift in waste composition during the 1990s and into the 2000s, resulting in great differences between the National Solid Waste plans in Greece and the actual situation “on the ground”; this is likely to have occurred in other EU states during the same period. It is obvious from these studies that there is a huge level of variation across the literature as to the percentage constituents for MSW, with seasonal variation playing a key role.

In microbiological terms, there are many different human and animal pathogens which are potentially present in MSW. These range from viruses to parasites including *Salmonella*, *Escherichia coli*, Enteroviruses and *Campylobacter* (Golueke, 1982; De Bertoldi *et al.*, 1983); the majority are present in food wastes and disposable nappies. The potential end uses of MSW necessitate that these organisms are destroyed, to reduce the potential impact upon human and animal health.

1.3.2. Landfill

Landfill is one of the most common methods of MSW disposal across both the developed and developing worlds and the UK disposes of around 85% of its MSW to landfill each year (Price, 2001; Mor *et al.*, 2006a); however, it has increasingly been under legislative pressure to become less polluting and this coupled with the

dwindling capacity in many nations (e.g., UK) has led to financial penalties being invoked for disposal (Sesay *et al.*, 1998). Lau *et al.* (2005) state that the amount of waste entering landfills in Hong Kong had been increasing by around 5% year-on-year since the mid-1980s; this would result in all of the landfills within the region being filled by 2015. Similar situations are occurring across most of the economically developed nations of the world.

Sites have improved in design and build quality over time, yet there is still potential for leachate to percolate through the outer areas of the landfill, potentially causing water pollution over the long term (Bari and Koenig, 2001; Mor *et al.*, 2006b). There is widespread belief that even sanitary landfill sites produce many bad odours (Schulze, 1962; Mor *et al.*, 2006a) and attract vermin such as flies/rats, furthering the potential for conflict between the operators and the population in the areas around such sites.

Leachate production and leakage from sites depend heavily upon the environmental conditions which prevail within landfills; the pH and redox potential of waste are two of the most important factors (Hoeks, 1983). The constituents of the leachate itself varies with the composition of the waste present within a landfill site, although high levels of organic compounds are to be expected, with Xenobiotic Organic Compounds (XOCs) being particularly common (e.g., phthalates and chlorinated aliphatics) (Baun *et al.*, 2004). There is also a possibility of the degradation products of XOCs producing even greater environmental damage than the parent compounds when released to the local environment.

Landfill gas is produced by the, often anaerobic, decomposition of biodegradable materials buried within landfill sites and typically consists of around 50-60% CH₄ and 30-40% CO₂, with the remainder being trace amounts of various compounds such as aromatics and chlorinated organics (Khalil, 1999); around 25 Tg yr⁻¹ of CH₄ is emitted from landfills according to this study. Prior to CH₄ generation, biodegradable matter degrades via hydrolysis to fatty acids, amino acids and other low molecular weight compounds; the dominant product here is regarded as being acetic acid (CH₃COOH). It is from these intermediate products that CH₄ is generated by methanogenic microorganisms (Kotze *et al.*, 1969;

Farquhar and Rovers, 1973; Hoeks, 1983; Bogner and Spokas, 1993; Bockreis and Steinberg, 2005). Due to the amount of CH₄ produced by landfill sites, they have been identified as being a prime source of anthropogenic CH₄ emissions. The high global warming potential of CH₄ (Crutzen, 1991; Mor *et al.*, 2006b) places a great need upon landfill operators to reduce emissions, yet the quantity of CH₄ being produced often relies upon the amount of waste in a particular site and its composition (Hoeks, 1983). Conditions for gas production include sufficient moisture content (over 25%) and a pH of between 5.5 and 7.0 (Farquhar and Rovers, 1973; Mor *et al.*, 2006a). Production rates increase through the thermophilic range up to around 55°C before reducing dramatically past this point, possibly due to microbial death (Kotze *et al.*, 1969). Whilst the production of CH₄ from landfill is very important with respect to the fight against greenhouse gas emissions and global warming, it also has localised effects upon the vegetation growing on, or around, landfill sites (Hoeks, 1983). Recovery of landfill gas would reduce emissions from landfill sites, helping to abate global warming to some extent (Mor *et al.*, 2006a).

Mor *et al.* (2006a) recommend that separation of the different fractions of MSW occurs, with the view that landfilling the different materials separately could reduce potential hazards and facilitate possible landfill mining activities in the future; this is especially significant if recycling is not implemented in some nations. However, this appears to be a quick fix and the use of sorting technologies would allow the processor to send material to recycling plants for reuse rather than individual landfill sites for disposal.

Covering materials are widely used at landfills to create layers between daily deposits of waste as a method for reducing pest infestation, movement of waste by wind and the reduction of odour emissions (Hurst *et al.*, 2005). There are many different types of material which are utilised for this process such as construction and demolition wastes (Hurst *et al.*, 2005) although this industry is now in decline in the UK and the amount of suitable material is reducing. As a consequence of changes such as this, MSW-based compost becoming a more popular widely available for use and is seen as more sustainable than most other

alternatives. Hurst *et al.* (2005) showed that a sample of MSW compost with a density of 590 kg m^{-3} removed around 69% of odorous compounds, whilst increasing this to 740 kg m^{-3} saw the removal efficiency rise to 97%. Current legislation does not mention the density of the material, but gives a minimum depth of 150 mm as a requirement. Hurst *et al.* (2005) state that increasing this appears to have no discernable effects as the majority of odorous compounds are removed within the first 100 mm of cover material. Reducing the depth of cover can cause its own problems, especially when revegetation of a site is undertaken. If MSW based composts or similar materials are to become widely used for daily cover, the problems of storage of the compost need to be addressed, with respect to where storage facilities would be cited. The major problem associated with utilising such cover is the volume within a landfill that it occupies, reducing that which is available for waste disposal, causing the landfill to fill sooner than expected (Hurst *et al.*, 2005).

Landfill is likely to remain as the dominant waste disposal route, for the short term at least, until more stringent action is taken towards changing the disposal habits of the public (Koufodimos and Samaras, 2002).

1.3.3. Alternatives to landfill

1.3.3.1. Incineration

Incineration is a costly process and is seen by many as a source of atmospheric pollution (Schulze, 1962; Lau *et al.*, 2005), somewhat unfairly due to the specifications of modern plants and the strict legislation they are governed by. Themelis *et al.* (2002) state that modern plants are far cleaner than older versions, with reduced emissions due to better management practices, better combustion and gas control systems and tighter control of the feedstock composition; improvements in the flue gas control mechanisms used have dramatically reduced the emission of toxic compounds and brought most modern incineration plants into compliance with legislation (Caserini *et al.*, 2004). Current methods of pollutant removal from exhaust gases include activated carbon adsorption, condensation and catalytic incineration, especially for the control of Volatile Organic Compounds (VOCs)

(Rene *et al.*, 2005). Pollution from ash leachate is also noted as being a major problem of incineration operations in some areas (Lau *et al.*, 2005). Treatment of the ash is widely used to reduce its potential to cause environmental harm; this can often be by way of solidification or chemical addition (Themelis *et al.*, 2002).

Many plants utilise a form of waste known as “Refuse-Derived Fuel” (RDF); RDF is produced by mechanically sorting/separating MSW either at source or prior to combustion. Such facilities improve the recovery efficiency of MSW treatment operations over mass-burn incinerators as the organic fraction is generally removed and treated elsewhere by composting or anaerobic digestion (Manninen *et al.*, 1997). Potential pollutants from incineration of MSW include heavy metals (especially Hg, Cd and Pb), organics (dioxins/furans), acidic gases (SO₂, HCl), dusts and grits and oxides of nitrogen (NO_x) (Caserini *et al.*, 2004; Lau *et al.*, 2005). One method of reducing the potential for heavy metal pollution resulting from incineration is considered by Lau *et al.* (2005); the removal of batteries at the waste production stage (in the household for MSW) prior to waste disposal. Chang *et al.* (1998) also mention the use of pre-sorting as a way to reduce the presence of potentially harmful substances such as heavy metals from the feedstock. There is some potential for energy generation from MSW incineration, either for domestic or industrial usage; this would help to reduce potential environmental impacts of the incineration process (Lau *et al.*, 2005). Incineration, either with or without energy recovery, is not considered as economically feasible when compared to other options due to the costs of operation and maintenance.

1.3.3.2. Recycling

Recycling of various components of MSW is a very simple process that can be conducted by the householder/waste producer prior to disposal; the majority of recycling on the household scale involves simply placing materials into different vessels to that material which is destined for landfill. There is a problem with this process, despite the simplicity, in that the householder often concentrates on removing waste from the site of production and disposing of it as quickly as possible (“Out of Sight, Out of Mind”) rather than thinking about the consequences

of such actions (Read, 1999; Price, 2001); there is also a large problem with public objections to sighting of waste management facilities within certain areas – the “NIMBY” (Not In My Back Yard) syndrome (Read, 1999).

To increase the chances of achieving the targets set out by the 1999 Landfill Directive, there is likely to be an increased onus on the householder to undertake separation of their waste and recyclables within the household, prior to collection and disposal. Such a practice would result in minor changes to lifestyle, when compared to other sustainable practices, yet there is a great feeling of apathy or reluctance to do so by many people who often feel that it is not their problem to deal with (Price, 2001). Weak markets for recycled materials along with the changeable composition of waste streams are key barriers to the progression of recycling past certain levels in the near future.

Recycling targets set by UK legislation in 2000 were highly ambitious and led to a massive increase in recycling facilities and in the information available to the public regarding the process. These targets continue for several more years, with the ultimate aim of recycling a minimum of 30% of household waste by 2010 (Mee *et al.*, 2004). Following a review of the various waste strategies within the UK, updated versions were produced for each nation in 2002 and contained different targets, often with higher recycling rates within the same time frame as the original 2000 document. The National Waste Strategy for Wales (*Wise About Waste*) sets out the following targets:

- 2003/04: Recycle/compost minimum of 15% of household waste with a minimum of 5% each.
- 2006/07: Recycle/compost minimum of 25% of household waste with a minimum of 10% each.
- 2009/10: Recycle/compost minimum of 40% of household waste with a minimum of 15% each.

(Welsh Assembly Government, 2002).

These are stringent targets, yet the 2003/04 one was met with the final figure being 16.5% of waste being recycled/composted; no data is yet available for the 2006/07 target, although the previous years (2005/06) figure was 21.9, indicating that it is likely that this will also be met (National Assembly for Wales, 2007).

The presence of many different types of plastics can hamper recycling efforts as well, this is because two types (e.g., PET (Polyethylene tetrathalate) and PE (Polyethylene)) have to be separated before recycling can occur as there tends to be no market for mixed plastics compared to the purer versions (Themelis *et al.*, 2002); this often adds to the costs of recycling facilities.

1.3.3.3. Waste minimisation

It can be said that the simplest method of reducing the amount of waste that ends up in landfill sites is to simply reduce the amount of waste being produced by householders, irrespective of recycling/composting in the home. Price (2001) cites the difficulty in getting people to perform this as coming from the lack of the “feel-good factor” that can be associated with utilising recycling facilities as the public in general like to have some reward for their actions and minimising waste does not give the same “reward”. The use of educational methods needs to become more widespread if minimisation is to succeed, yet Price (2001) highlights that the cost of this may not be met by the benefits and, as such, this method is not favoured by some Local Authorities within the UK.

1.3.3.4. Land spreading

Historically, there has been widespread use of plant and animal wastes as organic fertilisers on land used for cropping (Senesi, 1989), but this has been replaced by inorganic fertilisers as agricultural production became more intensive and desertification and erosion have become more widespread (Hernando *et al.*, 1989). In terms of disposal of organic materials, a variety of methods took over during this time including effluent release to water courses, incineration and ocean dumping; this resulted in many instances of pollution and constituted a loss of nutrients and energy (Sabey and Hart, 1975).

Interest in the disposal of wastes to land has been increasing for many years with a view to achieving sustainability within waste management and reduction of environmental problems associated with many disposal methods; legislative actions have also begun to place tighter restrictions on practices such as landfill and incineration. Sustainability is felt to be achievable through the recycling of nutrients and organic matter to soils, reducing overall losses from the system and as alternatives to incineration and other methods. The high energy costs associated with inorganic fertilizer production further highlight the need to switch to a more sustainable method of fertilization (Khaleel *et al.*, 1981; Chae and Tabatabai, 1986; Diaz-Raviña *et al.*, 1989; Levi-Minzi *et al.*, 1990; Prudent *et al.*, 1996). This is especially true in areas with high populations and a low amount of space suitable for other forms of waste disposal such as landfill (Schulze, 1962; Purves and Mackenzie, 1973; Wong *et al.*, 1983; Strom, 1985). The amount of organic matter contained in most non-industrial wastes makes them ideal candidates for land application to aid the replenishment/restoration of organic fertility in depleted soils and enhance it in those soils which have always been poor in terms of their organic content, especially as organic matter plays an important role in the humus balance of soil and, hence, soil structure (De Bertoldi *et al.*, 1983); it is also important to maintain organic matter levels in soils which are part of intensive cropping regimes (Senesi, 1989). Monoculture-based agriculture has also led to decreased soil organic matter levels and increasingly, crop rotation is encouraged to conserve this by way of residue recycling and different cultivation techniques, depending upon the crops grown (Herman *et al.*, 1977; Kanamori and Yasuda, 1979; Zielke and Christenson, 1986). The effectiveness of an organic material in terms of crop production is closely related to the nutrient content of the material, with mixes of different materials often reducing deficiencies of certain nutrients (Bunting, 1963). The main requirement of a composted organic waste to be used safely as a soil additive is the degree of stability or maturity it exhibits (Bernal *et al.*, 1998c); the absence of plant and animal pathogens and phytotoxic compounds are also highly important factors to be considered. Land spreading of waste meets two key objectives for sustainable waste management operations: (1) Waste Disposal, and

(2) Recycling of nutrients; although this can lead to pollution issues and as such, strict controls based upon the composition of the material in question and the type of soil to which this will be applied are needed to counteract this (Villar *et al.*, 1993).

It is important to note, however, that the addition of untreated wastes to soil is detrimental to the soil, and local, environment and that the vast majority of organic waste material which is added to land undergoes some form of treatment to minimise its environmental impact (De Bertoldi *et al.*, 1983; Senesi, 1989; Kostov *et al.*, 1991); if this is adequately carried out, there are many benefits to using organic wastes as soil amendments (García *et al.*, 1991). The treatment process (most commonly composting) is often matched to the best needs of the local area (Kostov *et al.*, 1991).

Zucconi *et al.* (1981b) and Van Assche and Uyttenbroeck (1982) found that the response of plants to the phytotoxic effects of MSW-compost applications is often species-specific and that the benefits of nutrient addition can be judged to outweigh the potential problems of toxicity.

In general terms, the N that is available to crops following organic material application to land is present as the mineralisable fraction of organic-N compounds and NH_3 (Donovan and Logan, 1983). It is important to note that any N present as NH_3 may be lost by volatilization, reducing the fertiliser value of the material.

The N-availability of a composted organic material to crops is also influenced by the timing of application to soil (Donovan and Logan, 1983; Sánchez *et al.*, 1997); spring crops require application around 2-3 months prior to sowing to allow microbial immobilization of N to be completed and re-mineralization to begin (Bernal *et al.*, 1998a). Following application, the rate of mineralization is often slow for an initial period and this can lead to N-deficiency in crops if delayed until sowing time; Bernal *et al.* (1998a) found that using a mature compost promoted the mineralization of N.

1.3.3.5. Benefits of compost addition to soil

Some of the properties exhibited by many composted organic wastes could be beneficial to soil and plants, especially the humus content which helps to preserve any beneficial effects within soils following application (Wong, 1985; Roletto *et al.*, 1985; Kostov *et al.*, 1994). Organic matter also contains large amounts of available plant nutrients; further benefiting soils (Rasmussen *et al.*, 1980). The positive effects of organic material being added to soils are often not achievable using inorganic fertilizers (such as improved humic structure), further highlighting the benefits of using organic-based matter (Mays *et al.*, 1973). Beloso *et al.* (1993) state that the land-spreading of composts is the most suitable method of recycling such organic materials whilst Perucci (1992) identifies it as a cost-effective method of disposal that can enhance the physical properties of the soil (Gallardo-Lara and Nogales, 1987).

It is particularly advantageous to add refuse-derived composts to soils with a view to improve the water-holding capacity as the organic content of the wastes increases during composting. It has been noted that the presence of extra quantities of humic organic matter form cationic bridges which help to increase the stability of the soil structure, allowing reduced pore space sizes (Mays *et al.*, 1973; Wong, 1985; Hernando *et al.*, 1989). According to Purves and Mackenzie (1973), the absorbent cellulosic fibres contained in the paper waste found in MSW helps to increase the water holding capacity, especially in drier weather. Other benefits include improved soil texture, enhanced permeability to water and air, retention of easily leachable minerals to humic colloids and a slower rate of nutrient release. The plant root system becomes stimulated by the presence of additional organic matter as a result of the improved mechanical characteristics of the soil and the extra nutrients present; this is often expressed by way of greater rooting density and more widespread root systems (De Bertoldi *et al.*, 1983). Avnimelech *et al.* (1990) however, found that this was only a temporary effect as the organic matter decays with time (this rate is greater during the summer months), giving rise to a need for repeated applications. Suppression of diseases/pathogen activity is highlighted as a

potential benefit of mature compost addition to soils by several studies (Hoitink and Boehm, 1999; Pascual *et al.*, 2000; Tilston *et al.*, 2002).

Bernal *et al.* (1998b) state that most organic wastes are high in N and fresh organic matter which help improve the agricultural value of soils and stimulate microbial activity. The presence of potentially available nutrients and organic carbon is mentioned by Wu and Ma (2001) as occurring in all but the most mature composts.

Several different studies have observed increased yields of various crops and other plants which correlate closely to the age (and maturity/stability) of refuse-derived compost. Two studies recorded this increase in *Brassica parachinensis* Bailey (False Pak Choi) (Wong *et al.*, 1983; Wong, 1985), whilst Avnimelech *et al.* (1990) recorded an increase in wheat (*Triticum* sp.) yields following compost application. This increased yield effect has been dubbed the “organic matter” effect by Senesi (1989) who also found that the addition of good quality compost increased the productivity of soils above that normally associated with the addition of an equivalent amount of inorganic fertiliser. However, this is not a nutrient concentration effect; it is more closely related to more efficient interactions in terms of nutrient supply and uptake whilst also reducing the possibility of yield decrease following over-adding of fertiliser, something which is commonly associated with inorganic fertilisers. Keeling *et al.* (1994) found that the addition of composted MSW (albeit immature compost) caused increased yield and attributed this to the improved concentrations and availability of nutrients. A gap of between 1 and 3 wk following application is given by studies cited in De Bertoldi *et al.* (1983) as the time before added organic matter degrades sufficiently to be beneficial to crops.

1.3.3.6. Potential problems with land spreading

1.3.3.6.1. Phytotoxicity of composted materials

Early work by Garner (1966) identified that the addition of refuse-based composts increased yields, but to a lesser degree than manure derived materials; it was concluded that there was something contained within the refuse that inhibited or

reduced plant growth, although at this stage definitive answers were not obtained. Phytotoxicity of materials is often associated with the possible accumulation of various substances within soil following land application and limits the scope of this form of waste management (Prudent *et al.*, 1996). The addition of immature compost (not stable with regards to mineralization and humification) (Iglesias-Jiménez and Pérez García, 1989; Serra-Wittling *et al.*, 1995) is one of the most frequent causes of decreased crop yields in areas where compost use in agriculture is common.

During composting, the general pattern of phytotoxicity is one where there is a rise during the intermediate stages of composting, followed by a steady decline towards the end of the process as maturation proceeds (Zucconi *et al.*, 1981 a,b). Frasinetti *et al.* (1990) identified the link between the maintenance of phytotoxicity and anaerobic conditions within the composting mass, stating that there is even the possibility for an increase under the wrong conditions. One of the main ways that a superficially added MSW Compost can affect the soil profile is through leaching following rainfall. Avnimelech *et al.* (1990) observed that there was an increase in Electrical Conductivity (EC) up to 60 cm below the surface following compost application although further rainfall appeared to result in the leaching of the soluble salts further into the soil profile and out of the root zone; this study also found that NH_4^+ enrichment also occurred at similar depth (60 cm) along with a drop in soil pH.

It is a concern that with the addition of fresh composts to land that phytotoxic effects such as reduced crop yield may be induced within the soil system, due to the presence of toxic substances within the composted material produced during the decomposition of organic matter (Sabey and Hart, 1975; Zucconi *et al.*, 1981a,b;). Substances such as ammonia (NH_3), ethylene-oxide ($\text{C}_2\text{H}_4\text{O}$), volatile fatty acids, alcohols, phenols, other organic compounds and various heavy metals (e.g., Cu, Zn, Mn, Pb and Fe) are widely acknowledged as being amongst those factors responsible for phytotoxicity that can be associated with decomposing organic material (Wong *et al.*, 1983; Wong, 1985; Gosz and White, 1986; Tam and Tiquia, 1994; Bernal *et al.*, 1998b; García-Gómez *et al.*,

2003). Several studies (Lynch, 1977; Lynch, 1978; DeVleeschauwer *et al.*, 1981; Tousson *et al.*, 1968; Manios *et al.*, 1989) suggest that the presence of organic acids such as acetic acid contribute to the phytotoxic nature of immature composts; propionic, butyric, benzoic and phenylacetic acids are often also present, albeit in lower concentrations. Acetic acid is a precursor of CH₄ and, as such can be used as an identifier of the onset of anaerobic conditions (Lynch, 1977).

Gallardo-Lara and Nogales (1987) and Senesi (1989) concur with this and these studies also mention a drop in available O₂ and increased soil temperature as other possible effects. If material maintains a high state of biological activity as it is added to land, it can continue degrading within the soil and utilise oxygen (O₂) from the rhizosphere to mineralize organic-C within the compost (García-Gómez *et al.*, 2003), often resulting in anaerobic conditions within soil and a break down of indigenous soil organic matter (Bernal *et al.*, 1998b).

Biological blockage of soil-available N is another effect that is responsible for a decreased crop yield by way of N-deficiency (Hortensine and Rothwell, 1973; Terman *et al.*, 1973). It is also possible that certain micronutrients may be raised to toxic concentrations by addition of immature material. Anaerobiosis was identified by Avnimelech *et al.* (1990) as being responsible for reduced crop yields at high compost application rates; it was felt that the extra material utilised more O₂ and delayed crop seed germination.

The use of plant and seed based assays to determine if compost has phytotoxic tendencies is common due to the ease and reproducibility of such tests (Wang and Keturi, 1990). Wollan *et al.* (1978) recorded a lag in germination of *Lolium perenne* L. seeds in soils treated with sewage sludge. However, further experimentation found that this was not due to the presence of heavy metals in high concentrations as these were largely present in unavailable forms; the cause was thought to be germination inhibitors (such as NH₃ and C₂H₄) released by microbial activity under anaerobic conditions. Wong *et al.* (1981) concur with Wollan *et al.* (1978) that the presence of such substances causes a lag in germination times; however, Wong *et al.* (1981) do question the role of heavy metals in causing this lag, alongside volatile compounds.

Reduced plant metabolism is associated with the presence of an inhibitory environment, in turn this allows resistivity to phytotoxic conditions to build up within the plant; Zucconi *et al.* (1981a) concluded that increased metabolic rates (due to temperature or other conditions) could therefore lead to greater phytotoxic effects. The presence of complex organic molecules (such as poly-phenols) is known to cause phytotoxic effects, but the mechanism by which this occurs is either by direct inhibition of microbial action or that such molecules cannot be used as energy sources by heterotrophic bacteria (Gosz and White, 1986).

It is not only a reduction in yields which can result from the addition of immature, phytotoxic composts to soils, there is also the possibility of crops absorbing harmful concentrations of various substances (Purves and Mackenzie, 1973); this in turn can have an affect on the health of the consumers, be they animal or human.

Phytotoxicity could result from high soluble salt content (high Electrical Conductivity) or the presence of high levels of available heavy metals (Van Assche and Uyttenbroeck, 1982; Wu *et al.*, 2000); this limits which tests can be conducted to determine the maturity of a compost. Villar *et al.* (1993) recorded values of between 7000 and 12000 $\mu\text{S cm}^{-1}$ for finished compost and stated that these were considered to be high. The levels of salts (e.g., chlorides) (Van Assche and Uyttenbroeck, 1982) and heavy metals present within the feedstock, along with the maturity of the resulting compost need to be controlled to reduce the detrimental effects on plant life following addition to soil (Villar *et al.*, 1993).

It has been recorded that a drop in rhizospheric O_2 concentration following immature compost addition to soil can lead to problems with root respiration and, subsequently, the formation of Hydrogen Sulphide (H_2S) or nitrites (NO_2^-) by way of anaerobic transformation of sulphur compounds and incomplete nitrification of NH_3 (García-Gómez *et al.*, 2003). The drop in O_2 levels, when combined with organic matter decomposition can also lead to accumulation of short-chain fatty acids and phenols in the rhizosphere, further inhibiting root development and growth (García-Gómez *et al.*, 2003). This study also identifies the stimulation of microbial activity occurring after the addition of uncomposted wastes as being

induced by the high levels of Total Organic Carbon (TOC) present within this material; Lynch (1983) states that around 50% of the microbes present in soils are dormant under normal conditions. Several studies (Reinertsen *et al.*, 1984; Cogle *et al.*, 1989; Marstorp, 1996; Bernal *et al.*, 1998b) identified that water soluble components (e.g., carbohydrates and amino acids) of various plant materials tend to degrade swiftly, often releasing a flush of CO₂ causing low O₂ availability and strong reducing conditions within the soil; this can also be phytotoxic to certain species of soil fungi (Alexander, 1977). Iglesias-Jiménez and Pérez García (1992) highlight the use of O₂ by decomposing material, which in turn can lead to the development of a strongly reducing environment, allowing heavy metals to become more available and also inhibit seed germination. High CO₂ levels caused by increased respiration of readily available C-sources can result in the inhibition of nitrification due to the negative effects of elevated CO₂ upon nitrifying bacteria (Keeney *et al.*, 1985). Soil NO₃⁻ content has been found to decrease following compost spreading in all sections of the soil profile (Avnimelech *et al.*, 1990). This was felt to be related to anaerobic conditions and the onset of denitrification caused by organic matter addition. The study by Avnimelech *et al.* (1990) also noted that addition of MSW Compost during the summer resulted in a lower pH reduction within the soil; previous work by the author had shown that the decomposition of organic matter in warm temperatures resulted in the production of humic compounds whereas in lower (e.g., winter) temperatures, organic acids tended to form. Bernal and Kirchmann (1992) recorded that the decomposition of organic matter is an alkaline process, causing an increased pH; nitrification is classed as an acidifying process and tends to cause a drop in pH unless balanced by other factors.

Amending immature compost with other, stable, materials such as Peat-Based composts has been found to reduce phytotoxicity to a certain degree. The effect of adding in the other material may have caused a dilution effect upon the concentrations of phytotoxic substances or may have promoted the degradation or immobilisation of these (Keeling *et al.*, 1994). García *et al.* (1991) found that sewage sludge composts produced an initial drop in yields following application to soil, although this negative effect eventually disappeared and yields improved with

time; the breakdown of phytotoxic compounds within the soil was cited as the reason behind this.

Wong (1985) states that the decrease in phytotoxicity with time suggests that degradable organic compounds can have major impacts upon plant growth/yields. It is a well documented fact that immature, or partially decomposed, materials cause more adverse effects than beneficial ones following addition to land, whilst those which have undergone a longer period of decomposition, are humified, more “stable” and have a tendency to exert beneficial influences upon the soils to which they are added (Senesi, 1989). One way to prevent such problems is to chemically analyse the materials in question before they are added to land. De Bertoldi *et al.* (1983) state that there is a need for phytotoxicity tests to be carried out on composted materials prior to application to land irrespective of whether or not the composition of the feedstock or its history is known.

1.3.3.6.2. Heavy metals

If heavy metals are found to be present within compost applied to land, they can present a pollution risk and have the potential to interfere with the normal physical and chemical processes in soils and crops (De Bertoldi *et al.*, 1983). Heavy metals (such as Cu, Zn, Mn, Fe and Pb) tend to be present within MSW due to a number of sources such as food products and consumer goods (newspapers etc) (Van Assche and Uyttenbroeck, 1982) and as such, tend to be present varying concentrations within domestic waste. These concentrations are often quite high by comparison to other organic wastes, with the exception of sewage sludge. Heavy metals are all toxic above certain threshold concentrations, although even low concentrations of certain elements can cause chlorosis, inhibition of root growth or delay seed germination (Adriano *et al.*, 1973; Wollan *et al.*, 1978; Van Assche and Jansen, 1979; Wong and Bradshaw, 1982; Wong, 1985). Van Assche and Uyttenbroeck (1982) and Wong (1985) also state that the effects of the individual heavy metals would be different to that shown in their study if MSW-based compost was used due to the interactions between metals. Such an example is that Selenium (Se) has been shown to reduce Cadmium (Cd) accumulation in pig livers (Elfving *et al.*,

1981). The presence of “antagonistic ions” such as Ca^{2+} also helps to lower the level of toxicity of certain heavy metal ions (Van Assche and Uyttenbroeck, 1982; Wong, 1985). These studies also state that an older compost has a lower toxic effect resulting from heavy metals despite their concentrations changing only slightly with time, this suggests that other factors such as the relationship between pH and metal availability change with composting; availability of metals to soil is highly important in determining potential phytotoxicity. Villar *et al.* (1993) state that when compost is added to soils at too high an application rate, heavy metals could become available to plants, resulting in decrease yields; the metals which are most likely to become available in this situation are Mn, Zn and Cu, yet Pb and Cd tend to be present in very high concentrations as well.

High water solubility of heavy metals in composts can also be an indicator of the potential for those metals to leach into soils following application. Wong and Bradshaw (1982) and Van Assche and Jansen (1979) counter this to a certain degree by concluding that the level of root development inhibition caused by heavy metals is often species-specific, often relating to the presence of substances within the plants which can complex free metals, decreasing their toxicity. This study specifically looked at the effects of metals upon root development in *Lolium perenne* and discovered that Mn was unusually toxic to ryegrass, although this is thought to be linked to its inability to grow well in low pH soils (where Mn tends to be readily available). Van Assche and Uyttenbroeck (1982) discovered that whilst Lettuce (*Lactuca sativa*) experienced a drop in yields after MSW-compost application, Celery (*Apium graveolens*) was found to produce greater yields; this further highlights the species-specific nature of MSW-compost phytotoxicity. This study also stated that it is not just the concentration of a particular metal ion within soil which is responsible for the adsorption potential of a plant, there is an effect attributed to the pH and the metal availability due to this which can increase/decrease adsorption. Solubility of heavy metals tends to reduce as organic matter stabilization increases (Zucconi *et al.*, 1981a,b; Ciavatta *et al.*, 1993).

1.3.4. Composting

Composting is increasingly seen as a suitable, sustainable, method for treating organic wastes across the world due to its ability to treat different amounts of waste whilst remaining, in the basic form, a cheap and robust process (Kuhlman, 1990; De Guardia *et al.*, 2002). The resurgence in the popularity of composting links back to ancient times as the process has been used for thousands of years for treating and reclaiming organic wastes from agriculture (De Bertoldi *et al.*, 1988).

Decomposing organic wastes break down at differing rates depending on several factors including C/N ratio and C content (Khaleel *et al.*, 1981; Wu *et al.*, 2000) during both initial composting and the curing/maturation phase. This highlights the importance of the composition of the parent material, with any variation between batches of this contributing significantly to the properties of the end products; this is especially true for highly heterogeneous feedstock materials such as Municipal Solid Waste (MSW). Alternative sources of organic matter for both horticulture and agriculture are needed as the peat-based materials which are currently widely used are exhaustible and there is such a vast need for these at the current time (Van Assche and Uyttenbroeck, 1982).

Increased paper content of feedstock can result in slower decomposition rates due to the presence of recalcitrant cellulose fibres. This is often coupled with the fact that cellulose in paper is rarely pure and consists of many different length fibres and some lignin, causing breakdown to be a series of enzymatic reactions (Golueke, 1981).

Composting is an accelerated version of the natural aerobic biological process whereby heterogeneous labile organic materials are transformed into homogeneous end products containing humic-like substances by saprophytic microbes whose metabolism occurs in the aqueous phase (bacteria, fungi and actinomycetes) over a period of time often under controlled conditions; other substances produced include carbon dioxide (CO₂), ammonia (NH₃), water vapour (H₂O) and inorganic nutrients such as nitrate (NO₃⁻) (De Bertoldi *et al.*, 1983; Wong *et al.*, 1983; Wong, 1985; Viel *et al.*, 1987; Senesi, 1989; Lau *et al.*, 1992; Bernal *et al.*, 1998bc; Bari and Koenig, 2001; Barrington *et al.*, 2003); high

temperatures are achieved during the process by way of high metabolic activity of microbes and various exothermic processes occurring within the composting material, often resulting in the elimination of pathogenic organisms (De Bertoldi *et al.*, 1983; Kuhlman, 1990). The end product is stabilized through humification and partial mineralization of the organic compounds present within the feedstock material; actinomycete activity is key to the humification of organic matter during the latter stages of composting (De Bertoldi *et al.*, 1983; Viel *et al.*, 1987; Bari and Koenig, 2001). To be properly labelled “Compost” the parent material needs to have degraded to a humus-like end product which is easy to handle and store or has benefits when added to soil; this is linked to the stability of the organic components within a material (Zucconi *et al.*, 1981b).

The composting process provides an economic, hygienic, ecological and sustainable alternative route for waste management to incineration or landfill and can be used to process mixed waste streams, removing the need for separation facilities (Hortensine and Rothwell, 1972; De Bertoldi *et al.*, 1982; Keeling *et al.*, 1994; Bari *et al.*, 2001; Bari and Koenig, 2001); this process has the potential to dramatically reduce the size of the MSW stream destined for landfill.

Composting, under optimum conditions, can be completed within 1 month, although these conditions are not always achieved. It is important to note that this is considered the minimum time frame for successful composting, producing a stable end material (De Bertoldi *et al.*, 1983). The use of aerobic methods, as opposed to anaerobic digestion or composting (composting without proper aeration), is more widely used due to this potential for accelerating the process, reducing the time frame for treatment (Mbuligwe *et al.*, 2002). There are four main stages which occur during composting according to Schulze (1962):

- (1) Fermentation;
- (2) Acid Formation;
- (3) Thermophilic Activity; and
- (4) Temperature Decline;

Some more modern studies combine the first two stages and refer to this as the “mesophilic” stage (25-45°C) with the third being commonly known as the “thermophilic” stage (45-75°C) (Viel *et al.*, 1987; Senesi, 1989). Several different terms exist for the fourth and final stage of composting including “temperature decline stage” (Schulze, 1962), “maturation phase” and “stabilization phase” (Senesi, 1989). Bernal *et al.*, (1998b) define a different four stages;

- (1) Initial Stage; the raw material has not undergone any kind of decomposition yet;
- (2) Thermophilic Stage; the stage at which maximum temperature is reached (>40°C) and the rate of decomposition is fastest;
- (3) End of the Bio-oxidative Phase; temperatures begin to decrease back to ambient levels (7-12 wk after starting composting depending on feedstock (Bernal *et al.*, 1998c) and;
- (4) Maturation Phase; this is a “lengthy period” during which the material stabilises, humification increases and stabilises.

As research has increased, the names of each stage have changed according to new findings, with the study by Bernal *et al.* (1998b) quoting those names which are widely accepted at the current time.

The mesophilic stage is characterised by the presence and activity of yeasts and bacteria causing the breakdown of most easily degraded components of the feedstock, the labile C compounds (Bernal *et al.*, 1998a); a marked increase in the specific respiration of the material was recorded at this stage by Kostov *et al.* (1994) and was thought to indicate that CO₂ production was exceeding the energetic needs of the microbial population. O₂ consumption is highest during this early phase, whilst temperatures are generally maintained between 30 and 55°C (De Bertoldi *et al.*, 1983).

Thermophilic bacteria are the dominant organisms in the following stage (as the name suggests) and these break down cellulosic matter whilst the biooxidative heat produced by this process causes the destruction of many animal and plant

pathogens (e.g., *Salmonella sp.*) (Christensen *et al.*, 2002); this is also the stage at which most phytotoxic substances are released from the decomposing material (Senesi, 1989). Several different species of bacteria are present at thermophilic temperatures including streptococci and lactobacilli, although the most common species are spore-forming rods which resemble the *Bacillus* genus (Edwards and Rettger, 1937).

The destruction of plant and animal pathogens occurs when temperatures of over 65°C are reached for several hours or over 70°C for 30 min according to De Bertoldi *et al.* (1982) and De Bertoldi *et al.* (1983). These figures have since been revised by several legislative acts, including Schedule I: Part II (1) of the 2005 Animal By-Products Regulations which state that the temperatures reached must be a minimum of 60°C for 48 h or 70°C for 1 h and that this must be achieved twice during the composting process, with a turning event between the two runs (DEFRA, 2005). Pathogenic destruction is mainly caused by the heating, although it can also result from increased competition for nutrients with the indigenous microflora of the composting system whereby pathogens are out-competed for nutrients (De Bertoldi *et al.*, 1982).

During the transition between these two stages there is often a marked reduction in O₂ consumption and the rate of temperature increase slows for a short period (Viel *et al.*, 1987; Bari *et al.*, 2000; Bari and Koenig, 2001); this drop in O₂ use is likely to result from a decrease in organic matter mineralization which has been found to occur after around 3 wk of composting (García-Gómez *et al.*, 2003).

For the remainder of the thermophilic stage, temperatures can rise to levels over 70°C, levels which help to eliminate thermosensitive pathogens (Golueke, 1982), and oxygen consumption decreases as microbial action becomes inhibited by the increased temperature; this is related to denaturation of enzymes within the microbes (Bari *et al.*, 2000). Nitrification of NH₄⁺-N is also inhibited by the high temperatures experienced during the thermophilic stage, explaining why increases in NO₃⁻ tend to occur in the temperature decline stage (Bernal *et al.*, 1998c).

Collectively, the mesophilic and thermophilic stages is sometimes referred to as the “stabilization phase” (Viel *et al.*, 1987), although the majority of authors

(Senesi, 1989) use this term to describe the phase of composting which occurs as temperatures decline (after the end of the thermophilic stage) and characterise it by way of a decreased decomposition rate. The dominant microbes during the temperature decline stage (maturation/curing phase), are actinomycetes and fungi, often the same species that were present during the original mesophilic phase (Schulze, 1962; Chanter and Spencer, 1974; De Bertoldi *et al.*, 1983; Senesi, 1989). A longer maturation phase (temperature decline stage) results in increased fertilizer value of the compost, often resulting in increased NO_3^- levels and a more stable material which will mineralize N at a more suitable rate following addition to soil (Bernal *et al.*, 1998a).

Proteins and carbohydrates are transformed into humic acids during the early stages of humification, whilst the more recalcitrant components such as lignin (recalcitrant due to the cross linkages between the polymeric chains) tend to undergo this process in the latter stages of composting by way of cellulolytic eumycete activity (De Bertoldi *et al.*, 1982; De Bertoldi *et al.*, 1983; Roletto *et al.*, 1985; De Bertoldi *et al.*, 1988; Bernal *et al.*, 1996; Bernal *et al.*, 1998a). Lignin decomposition is limited to fungal activity, specifically basidiomycete activity (De Bertoldi *et al.*, 1982); these organisms reach peak activity levels after around 1 or 2 months of composting (De Bertoldi *et al.*, 1983). It is thought that static pile composting using forced aeration favours basidiomycete action due to the lack of turning cycles which could possibly disturb the fungal hyphae within the material (De Bertoldi *et al.*, 1982). Increased turning rates can also lead to large scale N losses, reducing the value of the material as a fertiliser (Bernal *et al.*, 1998a). During the latter stages of composting, De Bertoldi *et al.* (1982) recorded a decrease in the population of cellulolytic bacteria and an increase in cellulolytic fungi, the latter benefiting from the reduced temperature.

N content of composting material usually decreases with time due to losses by processes such as NH_3 volatilization, although if the dry weight values are considered, the N content actually increases; this is manifested in the decreasing C/N ratios associated with progressing composting (De Bertoldi *et al.*, 1982; De Bertoldi *et al.*, 1983). The presence of N-fixing bacteria also helps to increase the N

content of compost during the latter stages of the process; high temperatures inhibit the activity of these microorganisms during the earlier phases of composting (De Bertoldi *et al.*, 1982; De Bertoldi *et al.*, 1983). A period of 20 d has been recorded before autotrophic nitrification was detected in composting, felt to be caused by a drop in temperatures to levels below the thermophilic range ($<40^{\circ}\text{C}$) (De Bertoldi *et al.*, 1982; De Bertoldi *et al.*, 1983); it is thought that any nitrification that does occur during the early stages of composting is conducted by heterotrophic nitrifiers (Eylar and Schmidt, 1959; Hirsch *et al.*, 1961; Marshall and Alexander, 1962). The smaller molecular weight compounds (such as sugars) undergo further degradation as composting proceeds, with the organic carbon content often used as energy sources for the microbial population, resulting in an increase in inorganic substances such as NO_3^- (Dell'Agnola and Ferrari, 1979).

Virtually any organic waste material can be treated by composting (Senesi, 1989) including sewage sludges, food-processing waste and waste from the timber industry (bark wastes) (Witter and Lopez-Real, 1988). Composting is seen as a highly effective method of producing a material which works better with natural soil processes than the feedstocks used, especially in terms of its humic content (Senesi, 1989). It is also recommended to be used to help minimise the spread of plant pathogens and weed propagules through the high temperatures achieved by metabolic heat production (Lau *et al.*, 1992), although the soluble C and N contents and microbial activity do decrease with time (Robertson and Morgan, 1995).

Schulze (1962) states that the temperature that can be maintained within a composting heap depends upon the size of the pile, with larger piles being able to reach greater temperatures and sustain the thermophilic period for longer. Temperatures which can be reached during composting are generally between 60°C and 70°C ; however some larger compost piles can reach temperatures in excess of 80°C (Suler and Finstein, 1977) which can result in less than optimum decomposition rates. Rothbaum (1961) gave 76°C as the temperature at which thermophilic bacteria cease producing heat, identifying that this is around the maximum temperature that these organisms can withstand during compost.

Kuhman (1990) states that materials with C/N ratios of up to 50 can be composted

1.3.4.1. Optimisation of the composting process

Several key factors need to be optimised during composting to provide a good quality and aesthetic end product including particle size, porosity, moisture content, pH and C/N ratio and temperature (Viel *et al.*, 1987; Senesi, 1989; Lau *et al.*, 1992). The optimisation of porosity and moisture can be performed by using bulking agents which help to absorb excess moisture whilst opening out the structure, increasing aeration (Viel *et al.*, 1987). Schulze (1962) states that C/N ratios of over 50 would necessitate the addition of N-sources to allow composting to proceed; Melillo *et al.* (1982) and Bernal *et al.* (1998c) also state that a feedstock material with a high C/N ratio would undergo a reduced rate of mineralization and slower decomposition due to N deficiency. The use of C/N ratio as a predictor of decomposition rate is discounted by Levi-Minzi *et al.* (1990) who found no correlation between the two factors; this was backed up by the work of Reinertsen *et al.* (1984) who found that wheat straw decomposition did not follow this pattern. Despite having a faster rate of decomposition than several other organic feedstocks, Bernal *et al.* (1998b) discovered that a mixture that included maize straw did not reach the same level of maturity and stability during the same timeframe. They concluded that the particle size of the mixture meant that some of the waste did not come into contact with the microorganisms present; this may well link to the studies of Levi-Minzi *et al.* (1990) and Reinertsen *et al.* (1984) and explain why their results did not show correlation between C/N ratio and decomposition rate.

De Bertoldi *et al.* (1983) found that a correlation between C/N ratio and decomposition rate did, in fact, exist. They state that starting ratios over 35 will lead to microorganisms having to proceed through various life cycles to oxidize the excess C until the C/N ratio favours their usual metabolic processes, so resulting in decreased decomposition rates. However, lower starting values, e.g., around 10, although favourable to microorganism metabolism, can result in N losses by NH₃ volatilization, especially if pH and temperatures are high. The optimum C/N ratio for a compost feedstock is around 25, with most organic waste possessing values which can be corrected to these if needed (De Bertoldi *et al.*, 1983), although Kuhlman (1990) states that materials with C/N ratios of up to 50 can be composted

successfully; this is much higher a value than is found in the vast majority of studies.

The optimum pH for composting is between 5.5 and 8, although a range of between 3 and 11 can, theoretically, be processed; bacteria flourish in neutral pHs, whilst fungi prefer acidic environments (De Bertoldi *et al.*, 1983). The pH of composting material can vary due to the original pH of the feedstock, yet Schulze (1962) found that despite variations in the pH of material being added to the composting pile, the final pH was unaffected; this may be a result of a potential stabilising effect caused by composting. An initial drop in pH is often recorded as organic matter is broken down to acidic intermediate forms (De Bertoldi *et al.*, 1983). Following this, there is often an increase in pH as NH_3 is released from the mineralization of organic N sources (Atchley and Clark, 1979; McKinley and Vestal, 1985). Whilst the pH was below 7 during decomposition, Schulze (1962) found that a “sour odour” was emanating from the material.

O_2 availability is critical for composting to proceed as it is used by many microorganisms as the primary electron acceptor during aerobic respiration along with the oxidation of organic matter. De Bertoldi *et al.* (1983) give a figure of 18% for the minimum O_2 concentration in composting material; a later study by the author (De Bertoldi *et al.*, 1988) found that maintaining O_2 between 15 and 20% allowed composting to progress without excess temperatures and dehydration occurring. These figures are backed up by Finstein *et al.* (1987) who recommend 19% O_2 as being “biologically favourable”. Temperature of the composting mass can affect the solubility of O_2 within water, specifically the biofilm which itself can then affect the rate of microbial action (Finstein *et al.*, 1987). De Bertoldi, *et al.* (1982) recommend periodic turning of the composting mass to maintain the O_2 level; such practices can increase the running costs of composting operations and disturb the growth of fungal hyphae, therefore the need to turn the material has to be balanced against minimising disturbance. The onset of anaerobic conditions from reduced O_2 availability can result in the production of low molecular weight compounds including NH_4^+ , volatile fatty acids and Hydrogen Sulphide (H_2S) which in turn can cause bad odours to be emitted from the material (Cooper and

Cornforth, 1978; Spoelstra, 1979; Guenzi and Beard, 1981; Finstein *et al.*, 1987; Mathur *et al.*, 1990).

Moisture content of a composting material depends upon the physical state and particle size of the material; Tiquia *et al.* (1996) highlight the importance of controlling moisture levels due to the influence this exerts over gaseous exchange, itself having influences on O₂ availability within the composting pile and the maintenance of aerobic conditions. To modify the moisture content once composting has begun is difficult and often expensive, highlighting the need to achieve optimum values prior to commencement of the process (De Bertoldi *et al.*, 1983). Low initial moisture contents often cause the material to become dehydrated early in the process, reducing biological activity; high moisture levels result in the saturation of pore spaces, reducing aeration (Finstein *et al.*, 1987). Optimal moisture content for composting feedstock is said to be between 50 and 60% (Schulze, 1962; Suler and Finstein, 1977; McKinley and Vestal, 1985; Kuhlman, 1990); Iglesias-Jiménez and Pérez García (1992) give the range 40-60% as optimum whilst Viel *et al.* (1987) state a figure of 60-66% for this. Suler and Finstein (1977) state that moisture contents in excess of 70% are too high for optimal composting to proceed; Sesay *et al.* (1998) found that 21% was too low for microbial action and recorded a decrease in pile temperature when this level was reached. Again these studies highlight the differences of opinion which are present within the literature.

Moisture is produced during various metabolic processes during composting and this extra water can lead to problems with reduced O₂ content in the pore spaces of the feedstock and unless it is removed from the composting material either by gravity or in the normal flow of air (evaporative cooling; Finstein *et al.*, 1987) this can cause a reduction in available O₂ for microbes present in the feedstock. Schulze (1962) found that the moisture content of compost feedstock and the final product were relatively similar, showing that metabolic moisture had been removed in the exhaust gases. Iglesias-Jiménez and Pérez García (1992) found that after 10 weeks of composting, the moisture content of another MSW-based compost had decreased by around 17%, although it is possible that some of

this had been lost to the atmosphere during turning; these figures are echoed by Bari *et al.* (2000) who recorded a decrease of between 10 and 20% following a composting period of around 50 d (7 wk).

Variations in temperature of the composting mass are one critical facet of composting processes which can vary with aeration method (Hogan *et al.*, 1989; Bari *et al.*, 2000). Incorrect aeration levels can lead to an accumulation of heat if the rate of air input to the composting system is below requirements, which in turn can lead to a reduction of microbial action; if heat loss from the system is greater than the rate of metabolic heat production, the system cools before composting is completed (Lau *et al.*, 1992). The use of forced pressure ventilation can induce evaporative cooling in the most insulated regions of the composting pile, allowing heat to be carried to the outer layers (De Bertoldi *et al.*, 1983). Use of this in conjunction with automatic blow time regulation, controlled by temperature sensors placed into the pile, allows the optimisation of temperature within a pile and the composting rate to be more easily predicted (De Bertoldi *et al.*, 1982).

Industrial scale composting processing often requires some sort of initial conditioning to take place prior to commencing; this is utilised to increase the surface area/mass ratio by reducing the particle size, allowing greater microbial activity to occur (Golueke, 1981). However, the structural strength of a material reduces with smaller particle size, causing reduced pore space area, itself leading to a drop in aeration.

Pre-treatment of organic wastes can either be mechanical or biological-mechanical. The former often employs shredding and screening to remove inert substances, whilst the latter utilises short term biological degradation (1-3 d) followed by mechanical separation as before. Using biological degradation results in easier separation of the inert fraction from the degradable (De Bertoldi *et al.*, 1983).

1.3.4.2. Types of composting

Composting can take one of many different forms depending upon the aeration method and often the requirements placed on the end product can also influence the

type of composting process used (Barrington *et al.*, 2003). Aeration methods are commonly one of three different types: natural or static pile, passive and active or forced aeration. Natural aeration, as the name suggests, is a process which relies upon the diffusion of air through the composting material by convection and relies upon the properties of exposed surfaces. Passive aeration uses ducts installed underneath the compost pile to enhance the movement of air into the material. Active (forced) aeration uses the same technique as the passive form, but increases the amount of air entering the pile by way of fans forcing it into the ducts (Golueke, 1982; Barrington *et al.*, 2003); this often requires monitoring of the air flow rate to prevent cooling (excessive flow) or temperature accumulation (inadequate flow). Despite passive aeration being a cheaper alternative to the active variation, the latter reaches higher temperatures during composting; this is favourable as it increases the chances of destruction of pathogenic organisms (Barrington *et al.*, 2003). De Bertoldi *et al.* (1982) discovered variations in the length of the thermophilic stage of composting were linked to the type of aeration used. A temperature-feedback system using forced aeration had a thermophilic stage of around 3 wk, compared to 5 wk for a suction system and 6 wk for material that had undergone mechanical turning. This shows that the type of aeration used can speed up the early stages of the composting process by several weeks. The use of a variable rate aeration system is favoured by Leton and Stentiford (1990) as this study discovered that fixed rate systems were unable to cope with changing conditions within the composting material. Several studies (Suler and Finstein, 1977; Finstein, 1980; MacGregor *et al.*, 1981; De Bertoldi *et al.*, 1982) concur that using forced aeration (blowing) maintains greater control over temperature and moisture than vacuum-based aeration during composting.

Aside from the type of aeration used, there are two main forms of composting which are undertaken on commercial and industrial scales; In-Vessel Composting (IVC) and Open Windrow Composting, each with its own particular advantages and disadvantages.

In-Vessel systems generally consist of large containers connected to a fan arrangement allowing forced/active aeration of the composting material. Many IVC

systems utilise metering of input and exhaust gases to reduce variability of composting progression and odour/dust problems respectively (Haug and Davis, 1981). This basic form has been adapted and altered into many different systems often depending upon the legislative requirements of the region and the type of material which is being composted. These tend to operate with a shorter running time, mainly due to a reduction in the length of the mesophilic and thermophilic stages (Viel *et al.*, 1987), and increased temperatures when compared to open-windrows; this results in higher rates of pathogen destruction within the material being composted when compared to windrow composting (Golueke, 1982). In-Vessel composting allows a greater degree of control over the key factors which effect the rate of organic matter degradation, especially O₂ supply and temperature regulation (De Bertoldi *et al.*, 1983); the latter is important as temperatures over 65°C can cause problems with the rate of decomposition due to microbial death. Resistance to climatic changes is improved by using IVC systems, compared to windrowing, along with a greater reliability of pathogenic destruction (related to temperature control) (Haug and Davis, 1981). The level of microbial activity within a composting vessel is far more homogeneous than in windrow or static pile methods, which tend to suffer from gradients of activity (Viel *et al.*, 1987).

In-Vessel composting (IVC) can suffer from reduced aeration of some areas of the composting material, especially those away from the air source. With IVC systems which employ unidirectional aeration, the problem of poor aeration is often exacerbated as the temperature within the material is not kept constant across the entire pile (Bari *et al.*, 2000); this is also true for forced aeration windrow systems. Such a temperature distribution necessitates the mixing of material at intervals to improve the quality of the end product from systems using the unidirectional aeration approach. IVC systems also suffer from increased costs associated with installing and running such facilities, compared to the lower-tech versions such as Windrowing and Aerated Static Pile composting (De Bertoldi *et al.*, 1983). These costs often result from the fact that composting is not a simple process, it involves many species of microbes and cannot be completed within a few days; rather it takes weeks if not months to be completed successfully (De Bertoldi *et al.*, 1982).

Windrowing of materials is cheaper than IVC systems and is more versatile, providing more predictable results (Coker and Murray, 1981; Haug and Davis, 1981; Higgins, 1982; Jongejan, 1982; Kresse, 1982). Aeration of the composting mass is one of the most important factors affecting the progression of composting. In windrow composting, this is accomplished by either mechanical turning (Conventional Windrowing) or forced aeration (Golueke, 1982; De Bertoldi *et al.*, 1982). Windrows are often placed on concrete bases to allow ease of access for turning practices, allowing good aeration of the material; coverage using a fixed roof reduces the impact of climate (especially rainfall or excess heating/cooling) upon windrow composting, providing improved process control (Haug and Davis, 1981). One variation of this process is the Aerated Static Pile (ASP) method of composting, although this is not used as widely as conventional windrowing (Leton and Stentiford, 1990).

One disadvantage of Windrow composting is that the outer material does not reach the same temperatures as the core of the windrow; studies cited by Schulze (1962) state that the greatest temperatures within a windrow occur around 10-15 cm below the surface depending on moisture content of the material and ambient weather conditions. As the temperatures do not reach the same levels as those experienced within the pile, it is common for mesophilic organisms to survive for the duration of composting unless the pile is turned and mixed (Chanter and Spencer, 1974); ideally this mixing would place the outer material into the centre of the windrow pile for the next stage of composting, yet this can lead to recontamination of pathogen-free material (Golueke, 1982). This highlights the need for the two barrier system detailed in the 2005 Animal By-Products Regulations to ensure that all the material being composted has undergone at least one bout of heating to recommended temperatures and that the destruction of pathogenic organisms has occurred.

Compared to In-Vessel Composting, there is far less control over the conditions within the composting pile (Schulze, 1962) and these are often subject to changes in ambient weather conditions, especially if the process is undertaken outdoors; the smaller the windrow, the greater its susceptibility to change caused by

environmental conditions, although excessively large windrows can suffer from decreased aeration (Kuhlman, 1990). Schulze (1962) cites a minimum time period of between 4 and 8 months to produce finished compost due to the slower rate of decomposition associated with windrowing.

Across the profile of a windrow, there are different bands of microbial populations with the surface being colonised by mesophilic bacteria and fungi, the next layer favours the growth of thermophilic microbes (bacteria, actinomycetes and fungi with the core being populated by anaerobic spore-forming bacteria (Chanter and Spencer, 1974), especially in poorly aerated materials. Mechanical turning of the compost piles/windrows allows the mixing of these layers, reducing variations in temperature and oxygen levels and helping to lessen the possibility of phytotoxic compounds forming; Sherlock and Goh (1983) identified that reduced aeration due to increased CO₂ levels can result in the formation of anaerobic microsites and N₂O production; He *et al.* (2000) also mention the presence of such areas in aerobic systems where aeration is not as high as needed. Manios *et al.* (2007) found that windrows are environments where both aerobic and anaerobic conditions can exist, especially with increased pile size; this study suggests the use of higher proportions of bulking agents and increased turning events as methods that can combat the onset of anaerobic decomposition.

A study cited by Iglesias-Jiménez and Pérez García (1992) set a figure of 6-8 turning events during a 8-10 wk long composting period (bio-oxidative phase); the heaps in this study were left unturned for 12 wk during the maturation phase. García-Gómez *et al.* (2003) also quote a figure of 12 wk for maturing following the end of the active phase of composting.

For a decomposed organic waste to be classified as compost, it must meet several different requirements placed down in legislation. There is also a legislative need for the material to be considered as “mature” or “stable” if it is to be applied to land successfully (Wu *et al.*, 2000). Senesi (1989) defines compost as being a “stabilized and sanitized end-product of composting...beneficial to plant growth”; Keeling *et al.* (1994) also highlight the importance of the end product being sanitised and that a lack of this can lead to problems with the range of potential end uses available.

Use of immature or unstable materials as soil additives can result in phytotoxic effects being brought about due to incomplete degradation; Wong (1985) suggests a minimum time period of 4 months must pass before MSW-based compost can be considered safe to use as a “soil conditioner”. DeVleeschauwer *et al.* (1981) also identified 4 months as being sufficient to reduce any phytotoxic effects associated with immature composts. Senesi (1989) suggests that material which has not undergone the thermophilic stage of composting should not be used for agriculture; Zucconi *et al.* (1981b) found that strong toxicity did reduce following the first 3-4 wk of composting (end of the thermophilic phase), yet residual toxicity was still present after two months.

Most other authors state clearly that some weeks, if not months, of “curing” must occur before material can be safely used, although the study by Senesi (1989) freely admits that further degradation would occur within the soil if the pre-curing material was added; other studies have shown this to be detrimental to soil properties and favour curing before addition to land. Zucconi and De Bertoldi (1987) identify a link between the type of feedstock being used and the minimum amount of time which must pass prior to the material being classified as compost due to the great variation which occurs between feedstock types. Iglesias-Jiménez and Pérez García (1992) found that most of the various measures of maturity were optimal after 50-60 d of composting, a time which coincided with the end of the bio-oxidative phase. This study also found, however, that the level of optimisation was increased if the material was left to mature for a further time period (here this was 12 wk).

1.3.4.3. Maturity/stability of compost

Compost stability/maturity in terms of its physical, chemical and biological nature plays a key role in determining the end use of such material, and so is a facet of composting which has to be closely monitored (De Bertoldi *et al.*, 1983; Lasaridi and Stentiford, 1998). The use of unstable/immature materials can lead to the inducing of phytotoxic effects upon the recipient soils or begin the onset of

anaerobic conditions which are not favoured by native soil bacteria (Benito *et al.*, 2003).

One major complication associated with the composting process is the lack of a truly global definition of compost stability or maturity that covers both the commercial and research communities (Wu *et al.*, 2000; Wu and Ma, 2001); often leading to poor quality unstable materials being sold as compost (De Bertoldi *et al.*, 1983). To compound this, there is no single standardised test to evaluate the stability/maturity of any compost which is regarded as being accepted across the spectrum of compost producers and researchers; the provision of adequate specifications to all those involved in the composting industry appears to be a key factor in improving the quality of the end product, no matter what the feedstock (Senesi, 1989). Monitoring and determination of maturity/stability can be divided into three categories depending on the type of analysis used; physical (temperature/odour/colour), chemical (C/N ratio/volatile solids/humic substances) or biological (O_2 consumption/plant growth/ CO_2 production) (Lasaridi and Stentiford, 1998). To help produce a standard set of characteristics to identify compost maturity, the UK has adopted the PAS 100 Compost Specifications.

Wu *et al.* (2000) state that “Compost stability” refers to the degree or rate of decomposition of organic matter and therefore is closely related to microbial activity and, subsequently, the rates of oxygen (O_2) uptake and carbon dioxide (CO_2) production; changes in heat release can also be used to identify any changes in microbial activity (Wu *et al.*, 2000; Bernal *et al.*, 1998c; Wu and Ma, 2001). The respiration activity of a material is inversely related to the degree of stabilisation of a material undergoing composting (Bernal *et al.*, 1998c; García-Gómez *et al.*, 2003); this again refers to microbial activity as a method of assessing stability.

“Compost maturity” refers more to the degree of breakdown of phytotoxic compounds (especially organic compounds) and is generally tested for using plant and seed based assays (Wu *et al.*, 2000; Wu and Ma, 2001). Phytotoxic substances may be present either from the parent material, as intermediate products of microbial metabolism or from the humification of organic matter (De Bertoldi *et al.*, 1983; Tang and Miller, 1993; Bernal *et al.*, 1998c) and as such, they require a

lengthy period to degrade, highlighting the need for a long maturation period. Some studies (Zuconni *et al.*, 1981 a,b) identify the presence of phytotoxic substances as a transient stage of composting and as such it is a phase all composting processes go through prior to full stabilization.

Another study which uses microbial activity as an indicator of the degree of stability a compost possesses is Zuconni and De Bertoldi (1987). This states that when activity reaches a low and stable level, compost can be considered as mature; this study also places the requirement that organic matter is present in a predominantly humified state. Bernal *et al.* (1998c) refer to maturity being associated with the phytotoxicity and plant growth potential of compost (Ianotti *et al.*, 1993). Importantly, De Bertoldi *et al.* (1983) conclude that the absence of phytotoxic substances alone does not mean that compost is of “good” quality and that other indicators of this are needed. Wu and Ma (2001) appear to favour the use of microbial indicators of stability/maturity because composting is a microbial process and the stability/maturity of a material results from microbial action. One important point to note is raised by Bernal *et al.* (1998c); this study states that unstable composts often contain microorganisms which are producing phytotoxic compounds.

Senesi (1989) refers to “mature compost” as being stabilised organic material which has undergone a degree of “curing” but then goes on to mention highly stabilised material as “cured compost”; no minimum time scales are quoted for these materials, further compounding the problem of the lack of a common definition of each composting stage. This use of “mature compost” appears to blur the lines between the definitions from Wu *et al.* (2000) as this is described as “stable”; once more this highlights the differences within the literature. Regarding timescales for achieving maturity/stability, Bari and Koenig (2001) found that recirculating air inside the composting vessel, along with reusing the spent exhaust air, maintained a more homogeneous temperature and allowed a stable end product to be created in around 50 d (7 w); it is worth noting however that this is at maximum efficiency and is unlikely to be replicated 100% of the time on an industrial scale.

Several authors have mentioned the use of C/N ratio as an indicator of compost maturity but there are often great differences between the ideal figures quoted; Zucconi and De Bertoldi (1987) stating a value of 22, with two other studies cited by Villar *et al.* (1993) quoting upper limits of 20 and 15. Roletto *et al.* (1985) agreed with these studies to a certain extent with C/N ratios of between 20 and 15 reflecting humified organic material at the end of composting. Iglesias-Jiménez and Pérez García (1992) found composted MSW to have a C/N ratio of 12 following a 12 wk maturation period with Bernal *et al.* (1998c) and Das (1988) also finding this to occur in a variety of feedstocks after around 8 wk maturation. Das (1988) recorded a linear decrease of C/N ratio over the first 12 wk of composting, followed by a much slower rate of decline during maturation. García-Gómez *et al.* (2003) found that C/N decreased from 33.8 to 16.7 after a similar 12 wk curing period; the time periods recorded in these studies are far less than that quoted by Keeling *et al.* (1994) who suggest that stabilization of composted materials takes at least 6 months to achieve. Senesi (1989) concurs with these studies, giving a figure of <20 for a finished product, yet De Bertoldi *et al.* (1983) state that values of this magnitude would result in competition between soil microorganisms and plant roots for N sources. However, materials with low C/N ratios (<10) at the end of composting are often high in N and lose some of this during curing, causing the C/N value to rise (Senesi, 1989). Wong (1985) suggests that a C/N ratio in the vicinity of 10-14 gives the material in question a higher fertilising potential than those with higher values.

However, Iglesias-Jiménez and Pérez-García (1992) and Senesi (1989) also mention that C/N ratio should not be used as an “absolute indicator of maturity” because of the range of lignin and cellulose contents that could be present (and the effect this has on biodegradability) amongst other factors; increased paper contents tend to reduce biodegradability due to the cellulose molecules being more recalcitrant (Golueke, 1981). Zucconi and De Bertoldi (1987), Villar *et al.* (1993), Senesi (1989) and Iglesias-Jiménez and Perez-Garcia (1989, 1992) all state that C/N ratio should only be used as a “stabilization index” rather than an actual measure of compost maturity and that a C/N ratio of <20 is a necessary condition to

establish maturity, but it is not sufficient on its own. As the C/N ratio taken in the solid phase depends upon the N-richness of a material (Iglesias-Jiménez and Pérez García (1992) and because the majority of plant root/microorganism interaction occur within the aqueous phase, Senesi (1989) and Chanyasak *et al.* (1981) recommend the use of measurements of the ratio of organic-C and organic-N content in water extracts to give a better overall indication of C/N ratio with a value of around 5-6 indicating mature compost. However, this has been shown to be difficult to measure (Bernal *et al.*, 1998c) due to the low organic-N content in water extracts from mature composts; Hue and Liu (1995) favoured using the ratio of water soluble organic-C to total organic-N (TON) and give a value of <0.70 as indicating a stable compost; Parades *et al.* (2000) found that co-composting various organic wastes for around 15 wk resulted in the majority of mixtures achieving this ratio. Work by Bernal *et al.* (1998c) found that the value of 0.70 was reached by some immature samples at the end of the active phase or during the thermophilic stage, leading to the conclusion that a lower limit is needed; this study proposed an upper value of <0.55 for this ratio. Wu and Ma (2001) state that a material with a high level of water soluble organic-C is often considered as being immature and unstable; such materials also exhibit high rates of CO_2 evolution due to the increased microbial respiration. CO_2 production is also mentioned by Kostov *et al.* (1991) as a method of assessing compost stability.

These studies indicated that using this as a measure of maturity would help reduce any variation between parent materials, irrespective of geographical origin (Iglesias-Jiménez and Pérez García, 1992); something which is hugely important when considering composts produced from MSW. Iglesias-Jiménez and Pérez García (1992) experimented with composting MSW and found a figure of 6.0 for the C/N ratio in water extracts for mature materials.

Nitrification is an important measure of maturity as well, with the presence of NO_3^- coupled with a decrease in NH_4^+ suggesting a material is ready to use for land application (Bernal *et al.*, 1998c). Another N-based ratio which can be used to assess maturity is the $\text{NH}_4^+/\text{NO}_3^-$ ratio as long as the value is less than 0.16; it is thought that the longer a material is left to mature the more likely it is to produce a

figure of this magnitude, a 2 month period is mentioned as the minimum to ensure nitrification (Bernal *et al.*, 1998c). Work by Parades *et al.*, (2000) recorded co-composted organic wastes as having $\text{NH}_4^+/\text{NO}_3^-$ ratio values ranging between 0.10 and 3.37, concluding that only 2 out of the 4 mixtures in this study could be considered mature using this method.

If the material is to be considered sufficiently “mature” or “stable” to be used in agriculture, the characterisation of its humic content is felt to be a suitable indicator for the organic fertiliser value of compost (Senesi, 1989). Iglesias-Jiménez and Pérez García (1992) and Bernal *et al.* (1998c) also highlight the importance of the level of humification and the humic content of compost in assessing maturity. However, the first of these two studies goes on to dispel the idea of using the humification ratio of a material due to the method used to assess this being subject to extracting both humic and non-humic compounds at the same time. De Bertoldi *et al.* (1983) also reject the use of humification as a measure of maturity on the grounds that this does not always change during composting as existing humus may be mineralized whilst new humus is being produced. A development of the C/N ratio in its many guises that utilises the change in humic content to assess maturity/stability is the use of the Cellulose/Lignin ratio, which tends to decrease with composting, by Komilis and Ham (2003); this study gives a value of 0.5 for a mature compost.

Other factors which have been quoted as important in establishing maturity/stability of composted matter include cation exchange capacity (Roig *et al.*, 1988; Iglesias-Jiménez and Pérez García, 1992), latent metabolism and the presence of certain “indicator” organisms (Senesi, 1989), although most studies tend to use C/N ratio as values are widely quoted in the literature. The idea of indicator organisms is developed by De Bertoldi *et al.* (1982), but the changes in the populations of these species is favoured rather than an absolute value.

A well-stabilised material often has a good nutrient balance and allows the slow release of nutrients to soil although this may be caused by a lack of readily degradable C and N (Robertson and Morgan, 1995), whereas those fresher materials often release nutrients in such great quantities that microbial action is

inhibited until a suitable N source can be found (if C is in abundance and N is scarce). This can lead to unwanted competition between microbes and plants for N and subsequent N deficiency in some vegetation (Senesi, 1989) due to incorporation of inorganic-N into microbial tissues (Bernal *et al.*, 1998b). Iglesias-Jiménez and Pérez-García (1992) cite the biological blockage of available N as one of the major problems associated with the use of immature composts. García-Gómez *et al.* (2003) take this further and identify the cause of decreased inorganic-N content of the immature material (with a high C/N ratio), following addition to soil, as being microbial immobilization of N, especially as there was negligible N loss by NH_3 volatilization with aerobically decomposed material (Donovan and Logan, 1983); this process further decreases the amount of N available to plants. N immobilization tends to result from the addition of large concentrations of easily degradable C, such as those present in unstabilised composts (Flowers and Arnold, 1983; Beloso *et al.*, 1993; Bernal *et al.*, 1998a). Despite the positive effects of uncomposted materials (Keeling *et al.*, 1994; Robertson and Morgan, 1995) such as more readily available nutrients, there are also the associated problems of phytotoxic substances being present within the material prior to application, which can have detrimental effects upon crop yields (Zucconi and De Bertoldi, 1987).

Several studies (Chanyasak *et al.*, 1980; Hirai *et al.*, 1986; Hue and Liu, 1995; Mathur *et al.*, 1993) have cited using the levels of Dissolved Organic Carbon (DOC) as a possible method for assessment of compost maturity and stability. The Dissolved Organic fraction is felt to be most representative of level of transformation of the organic matter fraction as it is the part most subject to change during composting (Chefetz *et al.*, 1998). Lower DOC levels correspond with a higher degree of stability/maturity. Chefetz *et al.* (1998) identified three stages of change that DOC level go through; a primary phase of rapid decrease (approx 33 d) followed by a slowing rate of decrease (day 33-105) before a plateau is reached (day 105-end of maturation phase). However, the use of DOC as an indicator is affected by feedstock type, bulking agents and curing time (Wu *et al.*, 2000).

These differences can result in *stable* materials containing phytotoxic substances and *mature* composts with a high respiration rate; the conclusion of Wu

et al., (2000) is that if the feedstock was truly homogeneous, the use of DOC as an indicator could work, yet this study also recommends using pH and EC to monitor the progression of composting and maturation (these link to microbial activity and heavy metal availability). Senesi (1989) recommends that any organic amendment (compost etc.) must be classified on the basis of “composition, properties and behaviour in soil as close as possible to that of native soil organic matter, particularly humus”, if the negative effects of amendment are to be overcome. García-Gómez *et al.* (2003) identify mineralisable-C levels as possible indicators of maturity due to the inverse relationship between CO₂-C production and composting time; measuring mineralization rate of C in composts is one method calculating the rate of decomposition and extracting the best from compost following addition to soils (Bernal *et al.*, 1998b). This study also noted that mineralization rates levelled off as material degraded further with the labile organic compounds present breaking down during the first few hours/days of composting.

One of the final conclusions of the study by Iglesias-Jiménez and Pérez García (1992) was that a plant growth trial was needed as part of further work to establish the parameters for measuring maturity as “definitively the compost maturity is determined by plant yield response”; this is backed up by Das (1988) and García-Gómez *et al.* (2003) who also mention the use of plant-based assays to test for the present of phytotoxic substances and the suitability of composted organic matter upon field-scale trials.

Wong *et al.* (1981) and Zucconi *et al.* (1981 a,b) both agree that the use of biological-based testing methods (such a plant growth trials) is highly accurate for determining the maturity/stability of composted materials. García-Gómez *et al.* (2003) tested the effect of increased maturity upon yields of Perennial Ryegrass (*Lolium perenne* L.) and found that there was a distinct increase in yields with greater composting/maturing time.

Overall, it can be concluded that a greater time period that passes from the end of the composting phase before the material is used (the maturation phase) allows the various chemical properties of compost to become stabilised and produce a material with reduced toxicity and improved nutrient content.

1.3.4.4. Other maturity indicators

Microbial activity reducing and stabilising at a low rate is a better indicator of maturity according to Robertson and Morgan (1995); this study also reiterates the theory that the parent material plays a highly significant role in determining the finished product and the length of time required for a sufficiently mature compost to be produced.

Pathogenic organisms present within feedstock are generally destroyed by composting, with higher temperatures resulting in more sanitized materials. Studies cited by Schulze (1962) stated that most pathogenic organisms were destroyed by temperatures exceeding 55°C, although it was recommended that material reached 60°C to ensure sanitization; these studies also mention that time is an important factor in achieving destruction of pathogenic organisms during composting. The stability of compost determines the potential for pathogenic regrowth following composting (Finstain *et al.*, 1987), often during the maturation phase.

Rendering compost “inhospitable” to recolonisation by pathogens is a key condition of declaring a material mature/stable; this is usually the case when there is a lack of “readily metabolizable substrate” present. In this situation, reinoculation from outside sources (birds etc.) does not result in regrowth due to a lack of suitable material for microbial activity (Hussong *et al.*, 1985; Burge *et al.*, 1987). Finstein *et al.*, (1987) recommend that, whilst higher temperatures result in microbial death, the approach of stabilising the substrate will ultimately be more successful in reducing pathogen infection, a view backed up by Christensen *et al.* (2002) who cite several studies where high temperature alone was not enough to produce a completely sanitized end product. Russ and Yanko (1981) and Yeager and Ward (1981) found that given the correct conditions of temperature, moisture (highly important) and nutrients, *Salmonella* sp. repopulated composted, sterilized, biosolids after a duration of 1 yr; the bacteria were said to have survived in a desiccated state during this time. Sidhu *et al.* (2001) studied similar materials, with the results showing that increased storage time was a major factor in determining the level to which *Salmonella* sp. repopulate composted biosolids.

1.3.4.5. MSW composting

Composting of MSW has the potential to produce a more environmentally acceptable end product and a more aesthetic method of dealing with waste when compared to landfill and incineration by way of a process which consumes low amounts of energy (De Bertoldi *et al.*, 1983; Wong, 1985); this is mainly a result of the composting process transforming the heterogeneous starting material into a humified, homogeneous end product by way of microbial action (De Bertoldi *et al.*, 1983). The final product of this is often a useful organic amendment that can be used to improve soil properties and nutrient status (Gallardo-Lara and Nogales, 1987). There are several issues, however, which can narrow the range of end uses that a refuse-based compost could be used for and because MSW is such a highly heterogeneous feedstock material (Iglesias-Jiménez and Pérez García, 1992; Sesay *et al.*, 1998), the matter is only complicated more (Perucci, 1990; Levy and Taylor, 2003).

One such problem to be faced is the potential for issues to arise with inert materials being present in the finished compost, such as plastics and glass (Senesi, 1989; Perucci, 1990; Perucci, 1992; Ciavatta *et al.*, 1993; Villar *et al.*, 1993). The presence of these materials is the subject of some differences within the literature with some studies concluding that they offer increased porosity (Costa *et al.*, 1991) whilst others are of the opinion that they dilute the nutrient content of compost (Zucconi and De Bertoldi, 1987). Such contaminants often preclude the use of the end product for agricultural application, although it may be deemed suitable for post-industrial land reclamation or other areas with reduced impact on both environmental and human health (Senesi, 1989; Perucci, 1990; Perucci, 1992). Sorting of the material prior to treatment by composting can reduce the potential for toxicity and contamination from MSW-based composts following addition to soil (Paré *et al.*, 1999).

It is not only inert materials within MSW which can affect the quality, and subsequent uses, of the end product of composting. There are many different substances present within MSW which can place limits on the end uses including heavy metals, biodegradable toxic organics (e.g., detergents, surfactants and

pharmaceuticals) which can either pose a direct health risk or do so by way of their degradation products (Senesi, 1989).

The end quality of MSW-based compost depends on many different factors including the presence of phytotoxic elements, potential plant or animal pathogens and inert elements such as plastic/glass; the presence of these increase the C/N ratio of MSW compost feedstock, a factor which would then reduce the decomposition rate of this as well as reducing the quality and number of possible end uses (Van Assche and Uyttenbroeck, 1982; Das, 1988). It is not only potential contaminants that affect the quality of the product, there is often a need to keep transportation costs low (to maximise profit) and as such any inert matter used as bulking agents or excess moisture is seen to be unfavourable (Van Assche and Uyttenbroeck, 1982).

1.3.4.6. Possible end uses of composted MSW

Compost, generally speaking, has three main forms of land application (if this is the chosen end use): 1) Cropland application; 2) Disturbed land (e.g., mine spoils or roadsides); 3) Recreational land use (e.g., parks, golf courses and landscaping) (Kuhlman, 1990). Due to the presence of various phytotoxic elements (e.g., heavy metals) within MSW Compost, it is highly unlikely, at the current time, that this could be used in agriculture on any kind of scale.

There is the potential for MSW-based compost to be used as landfill cover material, especially as several studies have concluded that even relatively thin layers of this material (28-55 cm) can aid in the reduction of odour problems; Mor *et al.* (2006a,b) also cite this material as having the potential to oxidize CH₄ emissions, transforming them to CO₂, but it not clear how mature or stable this compost is.

Composting MSW prior to disposal in landfill is one method of disposal and also helps to reduce the volume of waste going into such sites (by way of degradation) (Bari *et al.*, 2000). The use of composted MSW as a biofiltration method is touted by Rene *et al.* (2005). Biofilters are often cost-competitive to chemical methods of filtration of exhaust gases from incinerators and other

methods of waste disposal. The presence of microbes within the material helps to degrade noxious chemicals with the rate of breakdown often depending upon factors such as the solubility of the substance within the liquid layer or the biofilm. Fluctuations in the process' conditions were found to cause problems with the efficiency of the biofilter (Rene *et al.*, 2005).

1.3.4.7. Nitrogen in composts

The majority of N present within compost feedstocks, especially refuse-derived materials, is present as insoluble protein which breaks down to form ammonia (NH_3), fatty acids and proteins. NH_3 forms when microbes are unable to utilise all of the N released by decomposition of these proteins (and other N sources) (Wong, 1985); excess NH_3 and NH_4^+ can lead to problems with normal plant growth such as inhibition of seed germination (Megie *et al.*, 1967). A high amount of easily degradable N (compared to C – giving a low C/N ratio) results in the accumulation of NH_3 -N followed by losses as composting progresses (Witter and Lopez-Real, 1988). To counteract this, several studies have tested the addition of easily degradable C-sources to composting material; this allows microbes to immobilise any NH_3 formed into their biomass but due to the costs involved, this strategy is often ignored at industrial composting plants.

High temperature associated with composting, along with the alkaline pH values and forced aeration of compost piles can cause rapid volatilization of NH_3 (Martin and Chapman, 1961; Witter and Lopez-Real, 1988; Bernal and Kirchmann, 1992); losses of N as NH_3 reduces the value of composted material as a fertiliser and also cause a certain degree of environmental pollution. This is highly relevant when sewage sludge is used as the feedstock for composting. (Witter and Lopez-Real, 1988). NH_3 changes from an ionized state at lower pH values, to the gaseous form as pH increases (Golueke, 1981).

Organic-N is mineralized to NH_4^+ during the earlier stages of composting, with this then being oxidised to NO_3^- by the end of the maturation phase; the NO_3^- concentration during the first few weeks of composting is often negligible (García *et al.*, 1991).

Total N (TN) has been observed to increase during composting, mainly due to a concentration effect caused by the degradation of labile organic-C containing compounds (Bernal *et al.*, 1998c). The rise in TN observed in this study indicates that the rate of organic matter breakdown was greater than that of N loss by NH_3 volatilisation. Increased NH_4^+ levels occur during the thermophilic stage of some composting materials due to the mineralization of organic-N compounds (Bernal *et al.*, 1998c), whilst decreased levels are often caused by NH_3 volatilisation in materials with low C/N ratios and high initial NH_4^+ concentrations.

The presence of NH_4^+ in composted materials, even in small quantities, was shown by (Van der Eerden, 1982) to cause root toxicity and inhibit normal growth and development of plants following application to soil. The mechanisms by which NH_4^+ affects plants include saturation of the lipid molecules within the cell membranes causing increased permeability of the cell (necrosis) and decreased flexibility (Van der Eerden, 1982); in turn this can cause an increase in the plants sensitivity to frost. Detoxification of NH_4^+ by plants can occur, as long as there is a high enough rate of metabolic activity and carbohydrates are sufficiently available; both of these are decreased at lower temperatures and carbohydrate abundance drops during darker hours, increasing sensitivity.

Increased NH_4^+ often results from the inhibition of nitrification by ambient conditions such as high CO_2 (Keeney *et al.*, 1985) or the presence of substances such as water-soluble tannins (Olsen and Reiners, 1983; Baldwin *et al.*, 1983). Reduced breakdown of NH_4^+ is also possibly due to the inability of heterotrophic nitrifiers to utilise it as a source of energy, especially when other substances (such as P or Mo) are limiting (Purchase, 1974).

An abundance of NH_4^+ can indicate a compost is not mature or stable enough to be applied to land as it has not fully decomposed and unstable substances are present (Villar *et al.*, 1993); this is often due to a stabilisation period which is too short. Senesi (1989) quoted data proposed by Zucconi and De Bertoldi (1987) that hypothesised that if a material was found to have a low C/N ratio and still have an abundance of NH_4^+ compared to NO_3^- ; it was likely that this could be classed as unstabilised. The use of NH_4^+ as an N-source by microbes is often favoured over

the use of NO_3^- due to the reduced energy input needed to assimilate the former into amino acids compared to NO_3^- which must be reduced and added to a carbon-skeleton (Eviner and Chapin, 1997).

1.4. The future of waste production and disposal

Whilst the amount of waste being produced across the world continues to rise, especially with increasing economic status of many nations in the “Developing” world, there will need to be a shift from unsustainable waste management practices (e.g., landfill) to the more sustainable methods such as composting of biodegradable matter and incineration with energy production. It is likely that the cost of waste disposal is to be passed onto the producer, no matter what level they are at, be they large scale industrial concerns, or the householder who will have to pay for the amount of waste they throw away each week. In the UK, this is currently accounted for in the council tax system with any changes to accommodate “Pay As You Throw” schemes requiring alteration to existing environmental legislation. Such practice could see the cost of landfill increase further, with other methods of disposal becoming more economically favourable as a result (Price, 2001). Price (2001) also states that there must be substantial changes to the waste management infrastructure in the UK (for example) to allow the switch to sustainable practices to be achieved with greater ease.

Overall, there is a great need for the general public to switch their waste disposal habits away from the throw-away culture that currently exists in some nations to a more sustainable practice. It is also important that environmentally friendly disposal options are employed in the near future to prevent any further environmental, social or economic problems.

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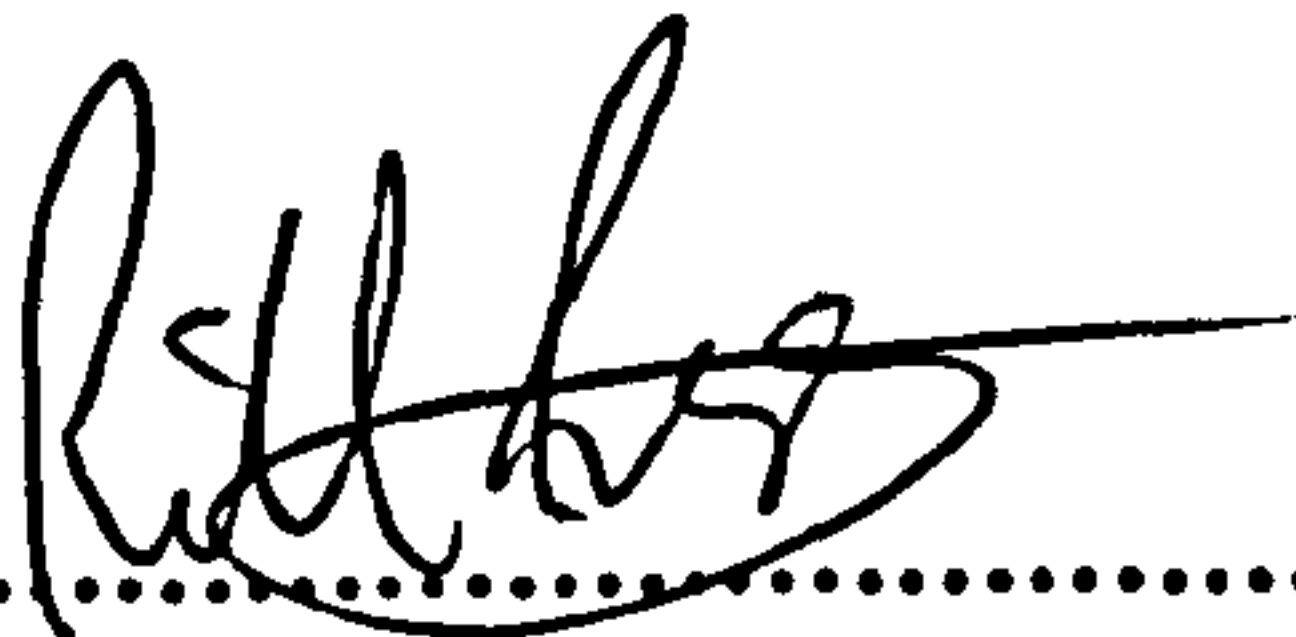
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**Evaluation of an in-vessel EcoPOD composting
system for the treatment of MSW**

Richard W. Furniss, Mark Farrell and David L. Jones

I declare that the experimental portions of this section, and any results derived from this, were carried out in co-operation with Dr. Mark Farrell; the data analysis, discussion and conclusions were independently produced by Mr. Richard W. Furniss and have not been submitted in candidature for any other degree.

Signed... 

Mr. Richard W. Furniss

Evaluation of an in-vessel EcoPOD composting system for the treatment of MSW

2.1 Abstract

As waste management techniques become more advanced and sustainable, there is a need to trial various alternative methods and materials for use in treating Municipal Solid Waste and divert it from landfill. MSW contains a large amount of biodegradable material and as such, has the potential to be used as a feedstock for processes such as composting and anaerobic digestion. The study aimed firstly to characterise two materials with high biodegradable contents prior to composting. MSW Fines and Green Waste were produced from local authority questions. Secondly, a trial was undertaken using in-vessel composting to evaluate the suitability of the two materials as composting feedstocks when MSW Fines were used alone and when combined with Green Waste in equal proportions. It was noted that both materials reached temperatures above various legislative requirements ($>60^{\circ}\text{C}$) (e.g. ABPR 2005) at the core of the vessel; however, it was also noted that the surface layers of the composting vessel did not reach these temperatures. As such, in terms of the sterilisation effect of composting, neither material was completely devoid of pathogenic life following composting; again, this is highlighted by other data. The use of Green Waste as a bulking agent in the mixed section appeared to speed up degradation of the feedstock when compared to the MSW Fines alone; this is highlighted by the CO_2 emissions data amongst others. Heavy metal content of the mixed section also complied with the PAS 100 Composting Specification, whereas the Fines alone did not. Overall, it was concluded that MSW Fines could be composted using in-vessel methods. However, the results also showed that the use of Green Waste or other bulking agent had more beneficial effects upon the chemical quality of the compost when compared to MSW Fines being composted without such an addition.

2.2. Introduction

The need to reduce levels of waste being sent to landfill across the world has been the subject of increasing focus at many levels in recent years, from national governments to local communities. Municipal Solid Waste (MSW) being produced across the UK, and the EU, has increased over the past two decades and with there being a finite capacity for landfill, a reduction in both the levels of waste produced and the amount ending up in landfill needs to occur.

Public participation in recycling schemes has increased over the last decade with greater provision of facilities for such practices. Increased numbers of kerbside collection schemes along with more Civic Amenity (CA) sites can result in larger amounts of materials being recovered; materials which would otherwise end up in landfill (Slater and Frederickson, 2001). The impact of this is two-fold; firstly, it can reduce the need for virgin raw materials (reducing environmental and economic impact), and secondly, extend the life span of a landfill site. From surveys of UK Local Authorities, Slater and Frederickson (2001) predicted that the number of kerbside recycling collections will increase sharply. Based upon Italian waste statistics, it was hypothesised that anything up to 375 kg of organic waste could be recovered from the average UK household each year if collections operated at maximum potential.

Despite the inclusion of green and kitchen waste collections at CA sites, and even the implementation of kerbside collections, there is still a significant amount of biodegradable material in the waste stream destined for landfill. Tonglet *et al.* (2004) noted a tendency for waste research to focus on developing methods to deal with waste after it had been produced rather than minimising its production. Minimisation is placed higher on the waste hierarchy as it has a far lower environmental impact than recycling or disposal (see section 1.1.1 of Chapter 1).

There is potential for biodegradable material to continue degrading rapidly whilst inside a landfill site, often resulting in increased landfill gas emissions (Norbu *et al.*, 2005). Ground subsidence has also been identified as a potential hazard of building on recently closed landfill sites as waste degrades and settles

which in turn can cause damage to any structures constructed on the site (Hirshfeld *et al.*, 1992; Lisk, 1991).

To reduce the levels of biodegradable matter being sent to landfill, various pieces of legislation have been introduced at a national, local and international level. The EU created the main legislative driver for change, the 1999 Landfill Directive (99/31/EC) (EU, 1999). This states that each EU member state must reduce the levels of biodegradable matter being placed in landfill. National waste strategies were implemented within individual nations stating targets for the reduction of biodegradable waste entering landfill. They also identified methods to allow utilisation of biodegradable materials and produce useful end-products such as compost or energy.

Composting is a viable method of controlled microbial degradation to break down large molecular weight polymeric molecules into smaller monomeric units that can be taken up and used more easily by plants and microbes for nutrition. MSW has been treated using composting techniques for many decades in several countries, with the final product often being applied to land to increase soil organic matter content; this is common practice in regions where soils are depleted in organic matter such as the Mediterranean (García-Gil *et al.*, 2004) and parts of Africa (Soumaré *et al.*, 2002). MSW composting in these areas is seen as a cost-effective method for helping to improve soil quality due to the relatively high costs of inorganic fertilizers or scarcity of other organic materials.

Within the UK, composting is used primarily by local authority to treat park and garden waste (green waste). Increasingly, there has been a move to collect kitchen waste (i.e. food waste or catering waste) from CA sites or kerbside collections to further reduce landfill and to improve the quality of the composting feedstock. There have also been some trials on the composting of unsorted MSW. However, unsorted MSW can contain large amounts of recalcitrant materials (e.g. glass, plastic) which don't biodegrade and can pose a potential physical hazard within the end product, limiting its potential end-use.

Gwynedd in North Wales utilises farm-based green waste composting at several sites, however, there is only one landfill site open which is located at Ffridd

Rasmus near Harlech in the south of the county. Other waste produced in Gwynedd is now sent to the Llanddulas landfill site in the neighbouring county of Conwy after the closure of the Cilgwyn landfill site in northern Gwynedd. The planning approval for new landfill sites in the county is very unlikely (Peter Evans, Gwynedd Council, pers. comm.). Consequently, space limitations plus legislative drivers mean that new approaches are needed for treating MSW. These approaches should be cost-effective, publically acceptable and environmentally friendly. The Ffridd Rasmus landfill site possesses a DANO® drum waste pre-treatment system. This system employs a large trommel screen to separate waste into two fractions; one with a particle size of >40 mm in diameter (Oversize Material) and one with a particle size of <40 mm (MSW Fines). This chapter investigates the feasibility of using a combination of the DANO® drum with an in-vessel composting system to treat the MSW Fines. This would simultaneously reduce the volume of waste and its greenhouse gas producing potential. In addition, the material produced (compost) may be useful for providing landfill cover or other uses.

The objectives of this work were as follows:

- Determine the composition of MSW Fines and Green Waste over a 6 month period;
- Evaluate the changes in physical and chemical composition of different materials during in-vessel composting and subsequent maturation.

2.3. Experimental

2.3.1. Site Description

This project was conducted at the Ffridd Rasmus Landfill Site, near Harlech in southern Gwynedd. Ffridd Rasmus is a municipal waste landfill taking waste from households in southern Gwynedd (Fig. 2.1). The DANO® drum (Keppel-Seghers UK Ltd., Wolverhampton, UK) is a mechanical waste pre-treatment facility which is around 30 m in length with MSW loaded into one end which moves progressively along the length of the drum. As the drum rotates, steel teeth within the drum shred the refuse bags, spilling the contents into the drum with material

smaller than the 40 mm in diameter (the size of the apertures in the sides of the drum) passing through to a conveyor belt below (Fig. 2.2).

Material that passes through the 40 mm screen is known as the “undersize fraction”, or, for the basis of this project, “MSW Fines”. Any residual material that does not pass through the mesh is moved to the end of the drum, unloaded and sent directly to landfill. The MSW Fines were used as daily cover material on the landfill site at Ffridd Rasus, providing a barrier between layers of waste deposited each day on the landfill in line with UK legislation.

Figure 2.1: Location of Ffridd Rasus Landfill Site, near Harlech, Gwynedd.

2.3.2. Composting at Ffridd Rasus

Waste managers at Gwynedd County Council identified three waste types for evaluation:

- (1) MSW Fines
- (2) MSW Fines mixed with Green Waste (50/50 volume/volume)
- (3) Untreated MSW (MSW which had not passed through the DANO® drum).

Prior to the onset of the composting trial, the MSW Fines and Green Waste were characterised over a 6 month period to establish the temporal variability in chemical composition.

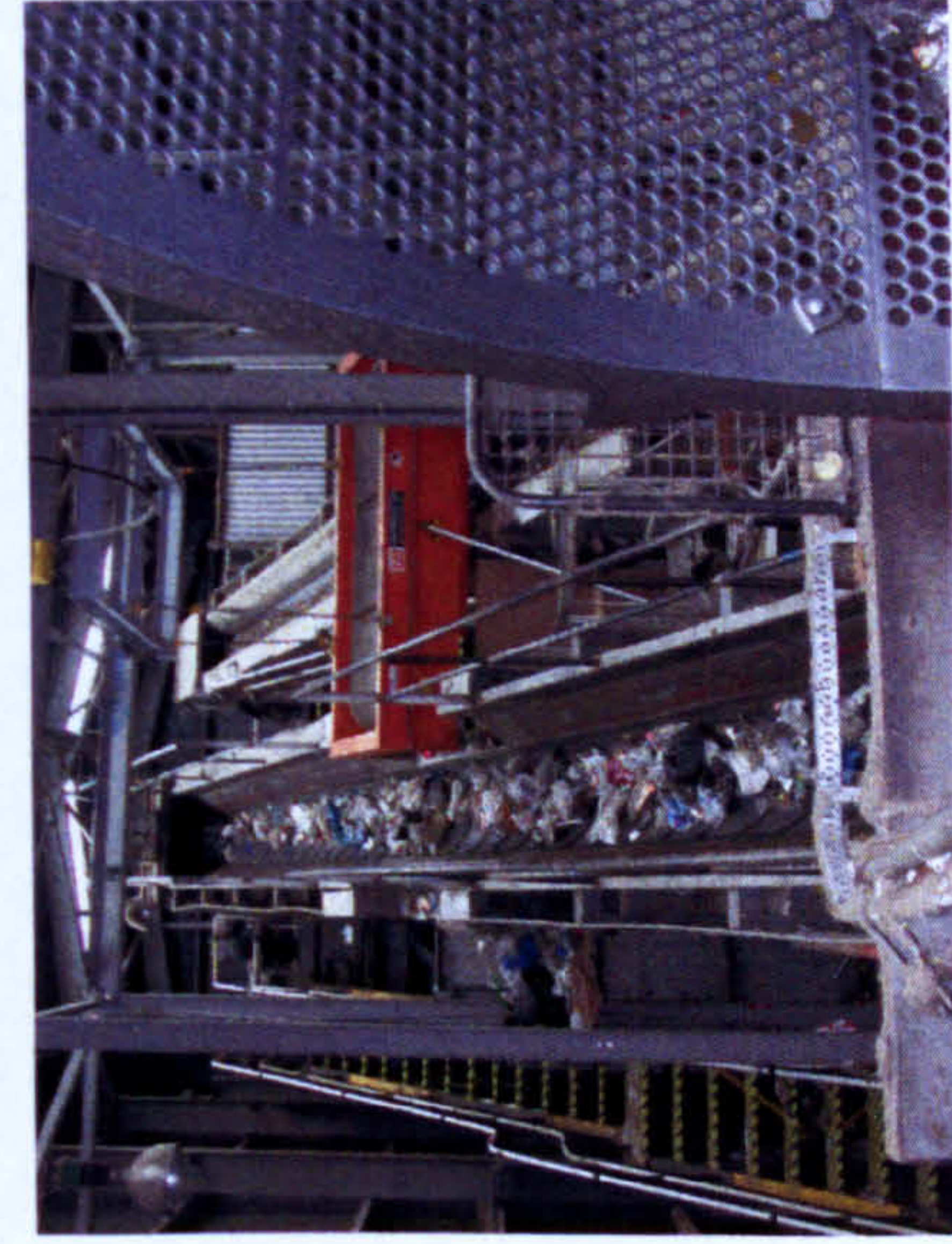


Figure 2.2: Photographs showing the loading of the DANOR® drum (top left), the outside of the DANOR® drum (top centre), the inside of the DANOR® drum (top right), the > 40 mm reject stream (bottom left), the <40 mm MSW Fines (bottom centre) and reject stream being sent to landfill (bottom right).

The Green Waste to be mixed with the MSW Fines was sourced locally from a municipal composting site located at Chwilog in south-west Gwynedd, near to Criccieth and Pwllheli. This site uses open windrows (ca. 3 m tall, 20 m in length) to compost Green Waste collected from the local area by Gwynedd County Council (Figures 2.3 and 2.4).



Figure 2.4: Composting of local authority collections of green waste in open windrows (upper panel) and turning of windrows (lower panel).

An in-vessel composting system was selected to treat the MSW Fines as open windrowing is not permitted on legislative grounds as it is not compliant with the 2005 Animal By-Products Regulations. Further, windrowing is more labour intensive and possesses a lower degree of process control.

The MSW-derived composts were produced using a commercially available CT-5® and EcoPOD® in-vessel composting system (ORM Ltd, Canterbury, Kent, UK and Ag-Bag International Ltd, Warrenton, OR, USA)(Figure 2.5). The in-vessel composter was filled with a CT-5® feed hopper (3 m³ feedstock volume) which uses a hydraulic ram to push the material through a filling chamber and into an extended 1.5 m diameter green plastic EcoPOD® vessel at a fill rate of approximately 3 m³ min⁻¹. At the same time, as filling the EcoPOD®, a rigid perforated plastic aeration pipe (7.62 cm diameter with 1.59 mm slits) is inserted into the base of the vessel to provide forced aeration. The aeration regime can be closely controlled by means of a timed fan (Ag-Bag International Ltd, Warrenton, OR, USA) running at a flow rate of 137 l min⁻¹ (Roberts *et al.*, 2007). The timed aeration regime was managed manually in order to maximise the rate of composting. In the initial 18 d, aeration was managed in order to maintain high temperatures and therefore optimal sanitation conditions (typically air pumped through the vessel for 2 min every 10 min). This was changed to 3 min on, 7 min off to compensate for the relatively wet feedstocks on day 19. On day 80, the duration of aeration was increased to 2 min on, 4 min off and the direction was reversed to extract moisture from the pile. This continued until the vessel was opened. Vents placed at distances along the sides of the EcoPOD® helped enhance the circulation of fresh air without danger of splitting the plastic vessel. Maturation of the composted materials was undertaken by opening the EcoPOD® and placing the different sections in static piles for a further 2 months. These piles were uncovered and unturned for the duration of the maturation period.

2.3.3. Waste characterisation

To characterise the waste materials, monthly samples of freshly produced MSW Fines were obtained from the DANO® drum facility and freshly delivered green waste from the Chwillog composting site. Standard chemical and physical analyses were carried out in triplicate on each replicate sample using the methods described below.

2.3.4. Compost sampling

MSW was collected from the municipal district of Dwyfor Meirionnydd, Gwynedd, UK and transferred to the Ffridd Rasus waste treatment facility. The MSW waste was then processed through the DANO® Drum pulverisation plant and the MSW fines collected. Green waste was obtained from the Dwyfor Meirionnydd district and shredded prior to use. Three different types of composts were produced from, (1) 15 t of shredded green waste (SGW) and 15 t of the fine fraction of MSW mixed together (MSW + SGW), (2) 30 t of the fine fraction of MSW alone, and (3) unsorted MSW black bag waste. These were placed in the in-vessel composter as described above.

During the active composting phase, triplicate independent samples were recovered from each treatment block by opening replicate holes in the vessel and removing material from a depth of 10 to 50 cm. After removal, the vessel was resealed using Duck® tape (Henkel Consumer Adhesives Inc., Avon, OH). Samples were removed from the vessel at day 0, 5, 7, 11, 18, 28, 39, 54, 75 and 83 when the vessel was opened. A final sample was taken at bagging from the mature static piles at day 145. Gaseous emissions were sampled using sterile glass syringes placed into the exhaust vents at the sample time as samples of compost were taken.



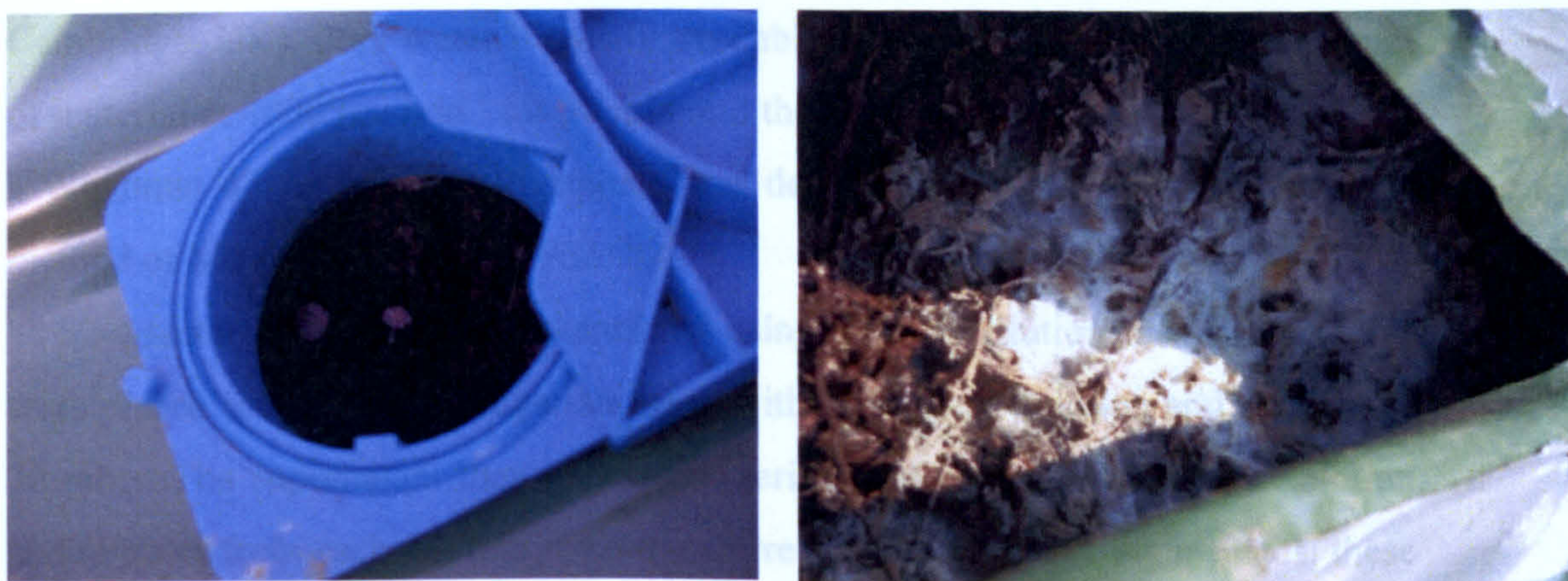


Figure 2.5: Photograph of the EcoPOD® in-vessel composting system showing the CT-5 Pod filling machine (top left), a filled vessel (top right), an air vent on the vessel (bottom left), and fungal growth during composting in the vessel observed when sampling (bottom right).

2.3.5. Chemical and physical analysis

pH (Hanna Instruments pH 209 pH meter) and electrical conductivity (EC) (Jenway 4010 EC meter) were determined in a 1:1 (v/v) deionized water:waste slurry, after mixing for 1 h. Moisture content was determined by drying samples at 80°C for 24 h and organic matter content was determined as loss-on-ignition at 450°C (16 h). Total C and N were determined using a CHN2000 analyser (LECO Corp., St Joseph, MI); prior to CHN analysis, samples were air dried (25°C, 14 d) before being ground in a T1-100 vibrating sample mill (Heiko Co Ltd., Fukushima, Japan) equipped with tungsten grinding vessels. Subsamples of the ground material were also taken for heavy metal analysis by way of digestion using a 1:4 HClO₄:HNO₃ concentrated acid mixture (MAFF, 1986). Heavy metals were determined by F-AAS using a SpectrAA 220 FS AAS in acetylene / air flame mode.

Nitrate and ammonium were extracted with 1 M KCl at a 1:5 (w/v) fresh sample:extractant ratio for 1 h on a reciprocating shaker at 250 rev min⁻¹ (Zhong and Makeschin, 2003). Nitrate and ammonium were determined using a San⁺⁺ segmented flow analyser (Skalar Inc., Norcross, GA).

Extractable K, Na, Ca and P were extracted using 0.5 M acetic acid using 1:5 (w/v) fresh sample:extractant ratio (Quevauviller, 1998). K, Na and Ca were analysed using a Sherwood 410 flame photometer (Sherwood Scientific,

Cambridge, UK). Pore water P (easily available P) was obtained by centrifugation of material at 4000 rev min⁻¹ for 30 min and the collected solutions filtered through a Whatman 42 filter paper. Phosphate was determined colorimetrically using the method of Murphy and Riley (1962).

Bacterial counts were determined using a serial dilution method. Samples were placed in sterile centrifugation tubes with ¼ Ringers Solution and shaken for 30 mins. The supernatant then underwent serial dilution with ¼ Ringers Solution and aliquots from each dilution were transferred to Chromogenic Agar plates; these promoted growth of *Salmonella* spp. and Faecal Coliforms which showed up as pink and green colonies respectively. Plates were incubated at 37°C for 24 h to promote growth; Colony Forming Units (CFUs) were expressed on a CFU/kg basis.

Fractionation data was calculated by air drying of samples (25°C, 21 d) and separation of the materials into the constituent fractions, recording the dry weights and calculation of the percentage of each material in relation to the overall sample weight. Water extractable dissolved organic C (DOC), dissolved organic N (DON) and total dissolved N (DN) were measured using a Shimadzu TOC-V-TN analyser (Shimadzu Corp., Kyoto, Japan). CO₂, N₂O and CH₄ concentrations within the exhaust emissions were measured using a gas chromatograph (GC) equipped with a flame ionisation detector.

2.3.6. Statistical analysis

All data was inputted into SPSS v12.0 (SPSS Inc., Chicago, IL), and a 1-way ANOVA performed. Tukey's Honestly Significant Difference (HSD) post hoc test was used to differentiate between the different samples. Means and standard errors were computed using Microsoft Excel 2003 (Microsoft Corp., Redmond, WA), and all graphical figures were produced in SigmaPlot v8.0 (Systat Software Inc., San Jose, CA). All values are expressed on a dry weight basis with the exception of gaseous emissions concentrations (parts per million – p.p.m.), bacterial counts (CFU kg⁻¹) and water extractable dissolved organic C and N (mg/l).

2.4. Results

2.4.1. Waste feedstock characterisation

This project monitored the temporal variability of the MSW Fines and Green Waste feedstock materials within the county of Gwynedd over a 6 month period. The chemical and physical compositions of mechanically-treated MSW Fines and shredded green waste are shown below.

2.4.2. Waste feedstock pH

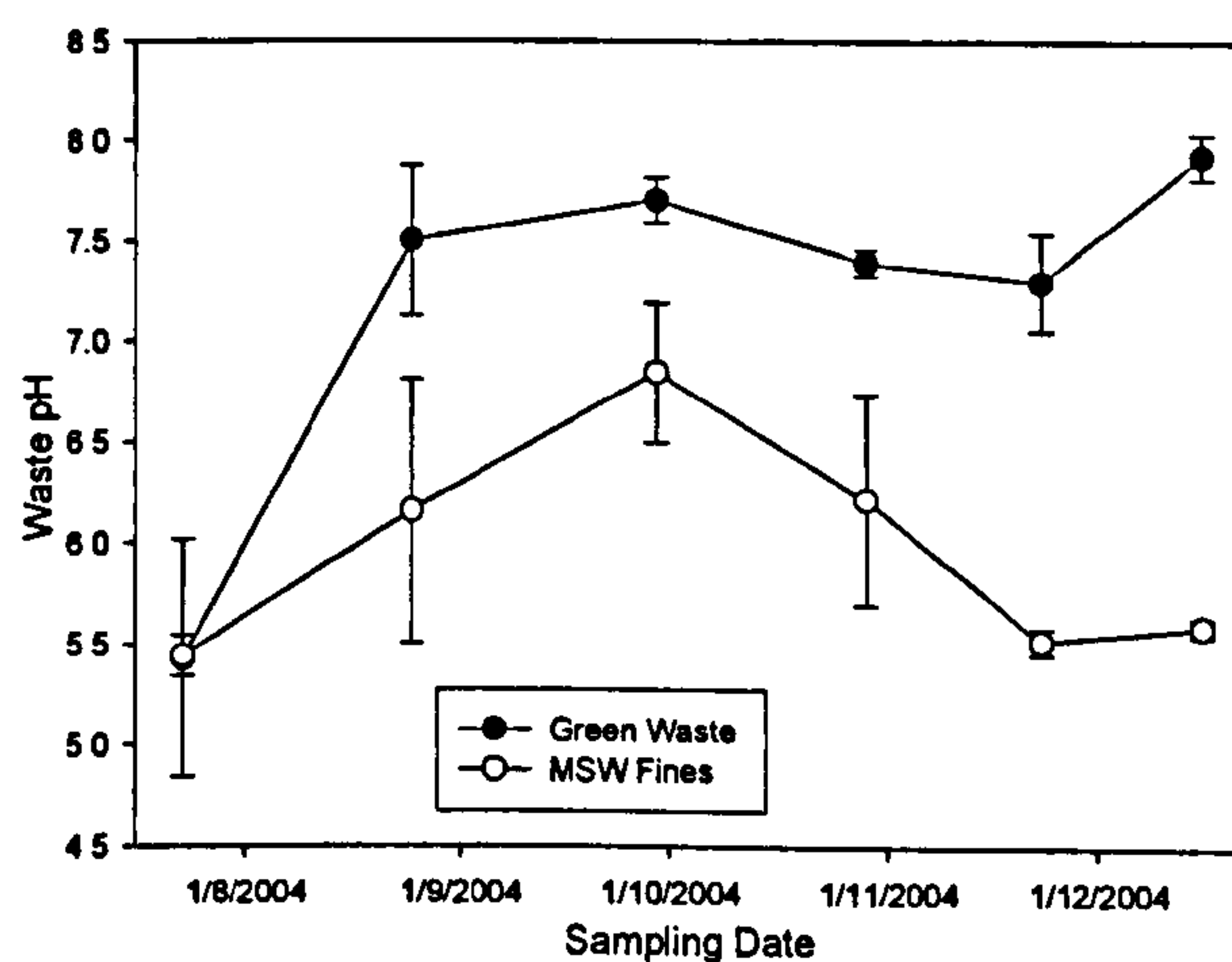


Figure 2.6: Changes in pH of MSW Fines and Green Waste sampled and tested over a 6 month period from two sites in Gwynedd, Wales. Bars are SEM.

The pH of the Green Waste was found to be higher than that of the MSW Fines and increased with time reaching a maximum of 7.9 ± 0.1 . In comparison, the pH of the MSW Fines increased to a maximum of 6.8 ± 0.4 in October before falling back to 5.6 ± 0.1 in December. The Green Waste had significantly higher pH levels than the MSW Fines ($p < 0.05$). No significant temporal differences were found between any of the individual of MSW Fines samples. The only significant difference for the individual Green Waste samples were that the first sample (July) was significantly lower than subsequent samples.

2.4.3. Waste feedstock electrical conductivity (EC)

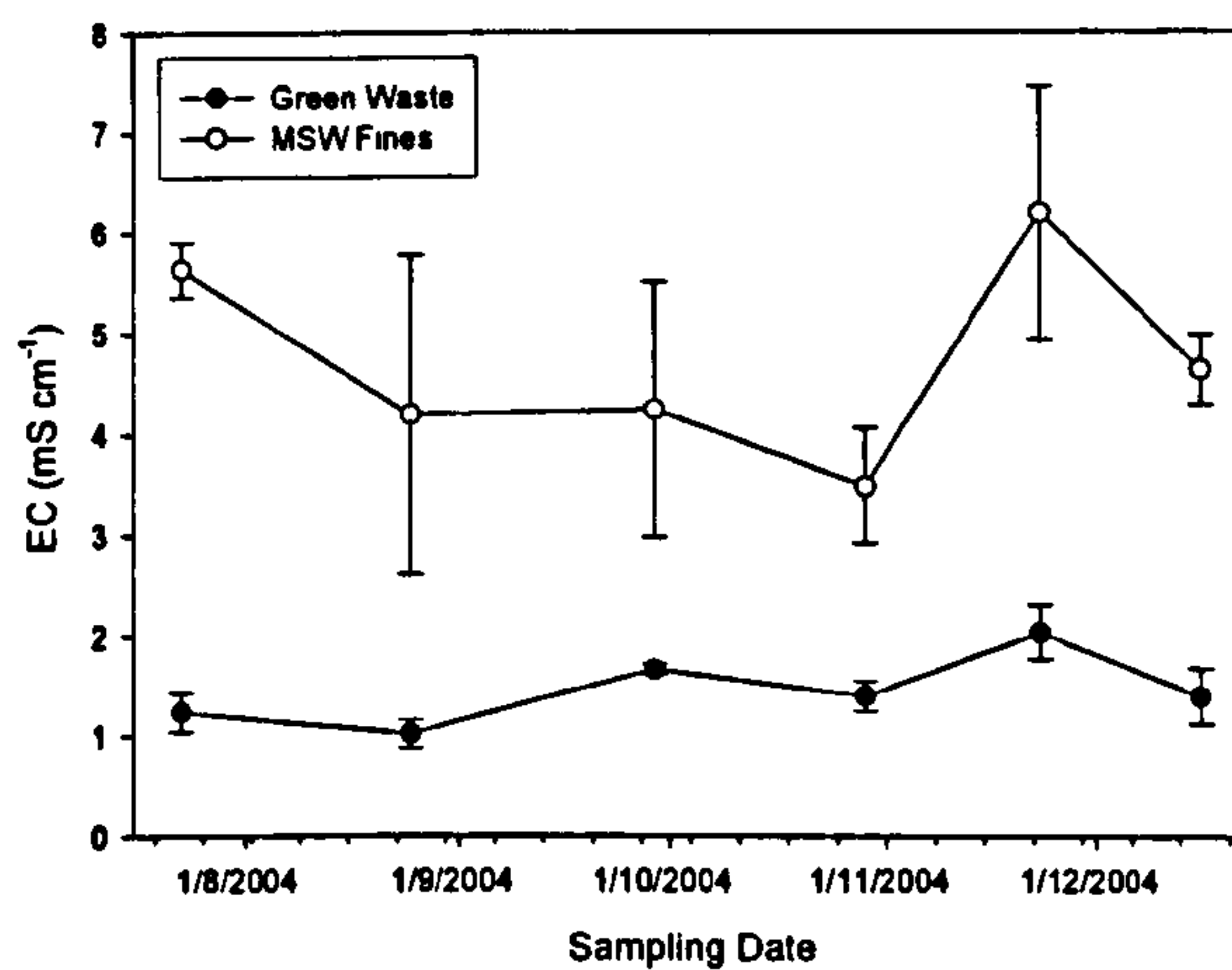


Figure 2.7: Changes in EC of MSW Fines and Green Waste sampled and tested over a 6 month period from two sites in Gwynedd, Wales. Bars are SEM.

Low variation in EC was recorded for the Green Waste during the 6 month sampling period with levels remaining relatively constant between 1.0 and 2.0 mS cm⁻¹. In contrast, greater variation was found in the MSW fines than in the Green Waste with EC rising from 3.5 to 6.2 mS cm⁻¹ over successive months. Although more variable, the EC of the MSW Fines was consistently higher than the Green Waste ($p < 0.0001$), possibly as a result of the higher concentrations of salts present within MSW Fines (e.g. household cleaning products and from kitchen waste; Baun *et al.*, 2004). No significant differences were identified between the monthly samples of the MSW Fines. The Green Waste did exhibit some differences, with the August sample having a significantly lower EC than the November sample.

2.4.4. Waste feedstock moisture content

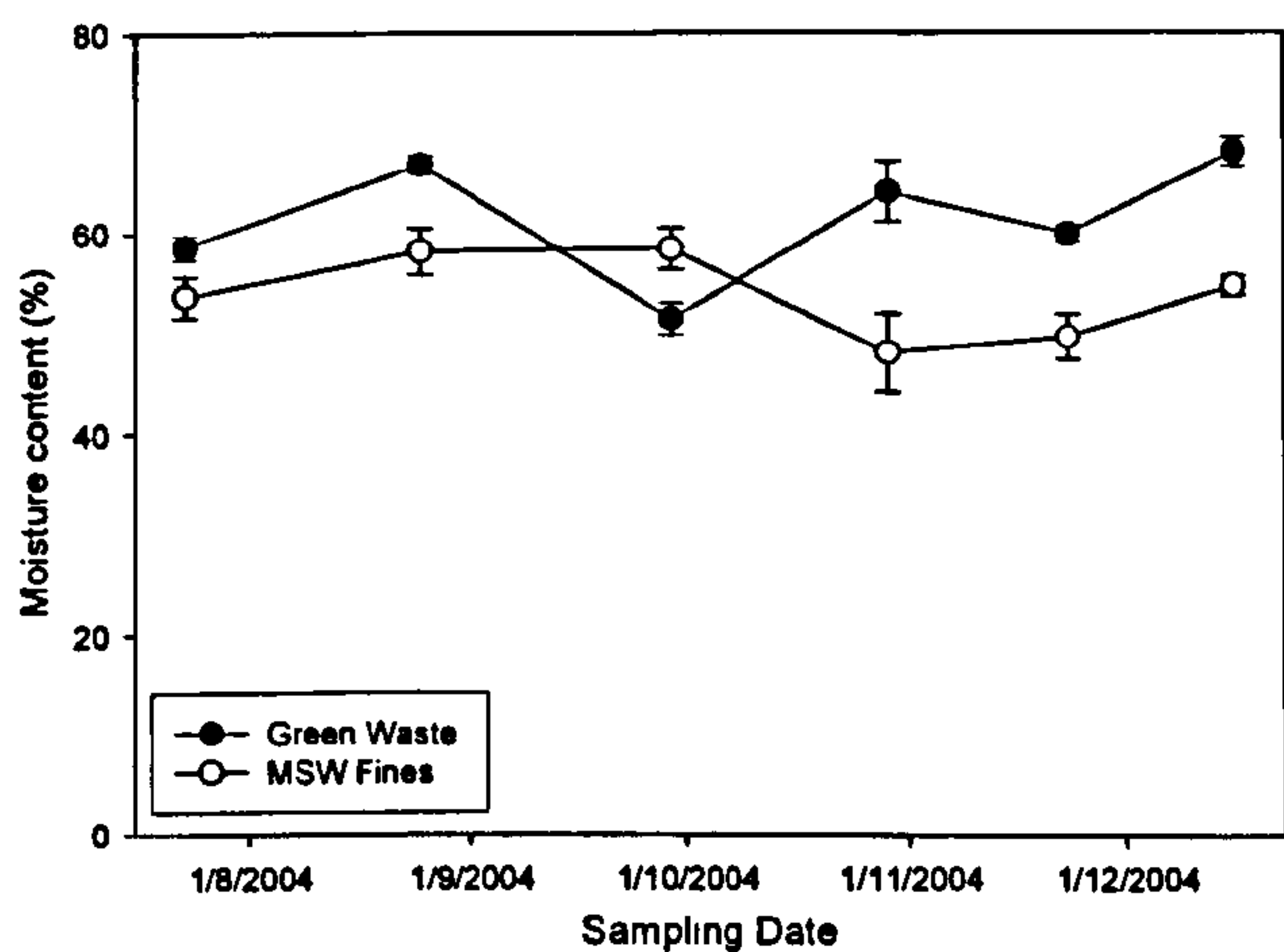


Figure 2.8: Changes in moisture content of MSW Fines and Green Waste sampled and tested over a 6 month period from two sites in Gwynedd, Wales. Bars are SEM.

The moisture content of both the Green Waste and MSW Fines remained relatively constant over the 6 month sampling period. Further, no significant differences were found between the monthly MSW Fines samples. The Green Waste samples taken in September and December were found to be significantly different, with the latter having the higher moisture content. No significant differences were apparent in moisture content between the two materials ($p < 0.05$).

2.4.5. Waste feedstock organic matter (OM) content

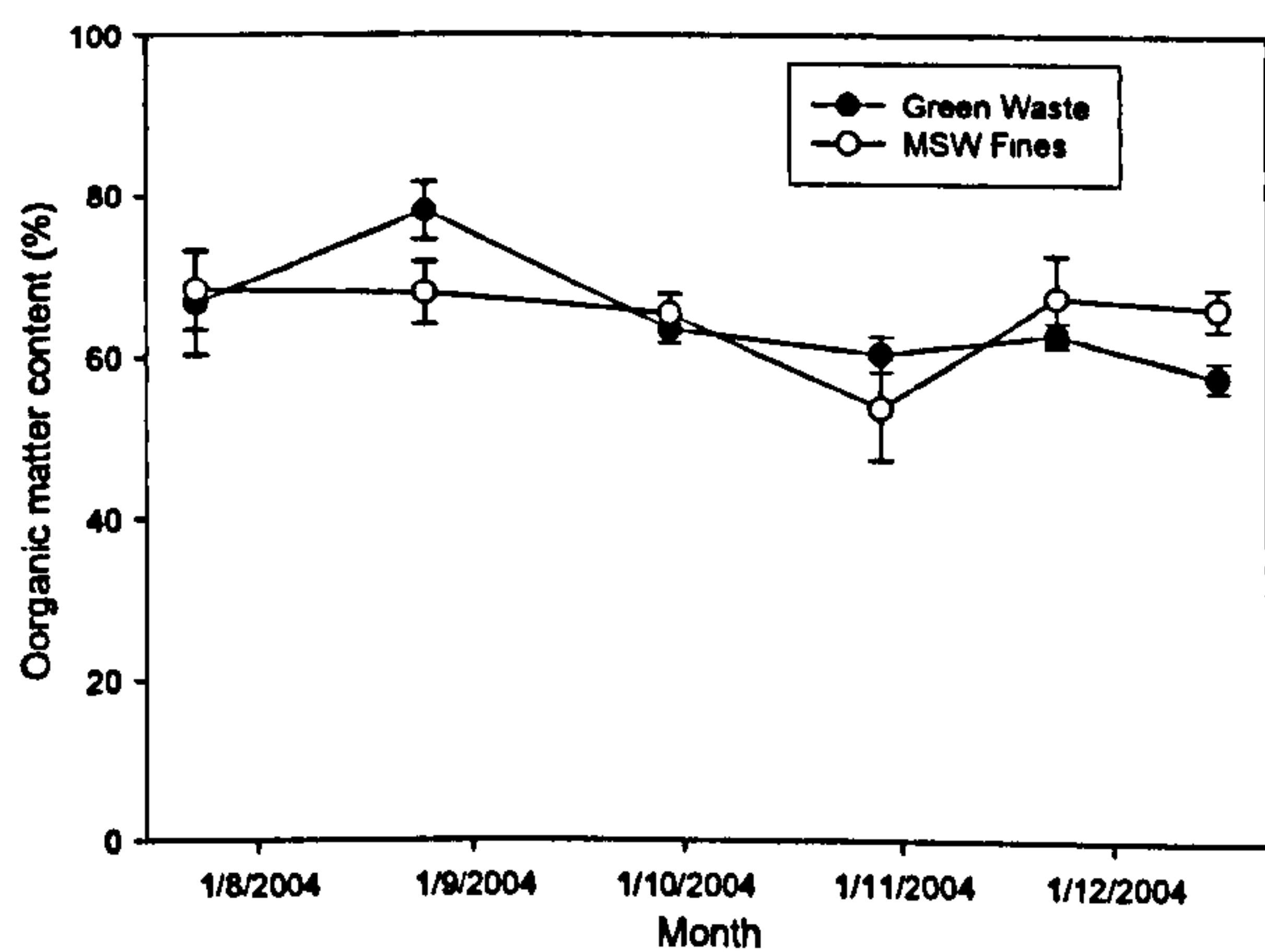


Figure 2.9: Changes in organic matter content of MSW Fines and Green Waste sampled and tested over a 6 month period from two sites in Gwynedd, Wales. Bars are SEM.

Organic matter contents of both materials remained above 60% for the majority of the analysis period. The differences between the materials were minimal for the majority of the trial; the August sample of Green Waste showed an increase in organic matter compared to the MSW Fines material. No significant differences were found within the monthly samples of each waste or between the two materials ($p > 0.05$).

2.4.6. Waste feedstock C/N ratio

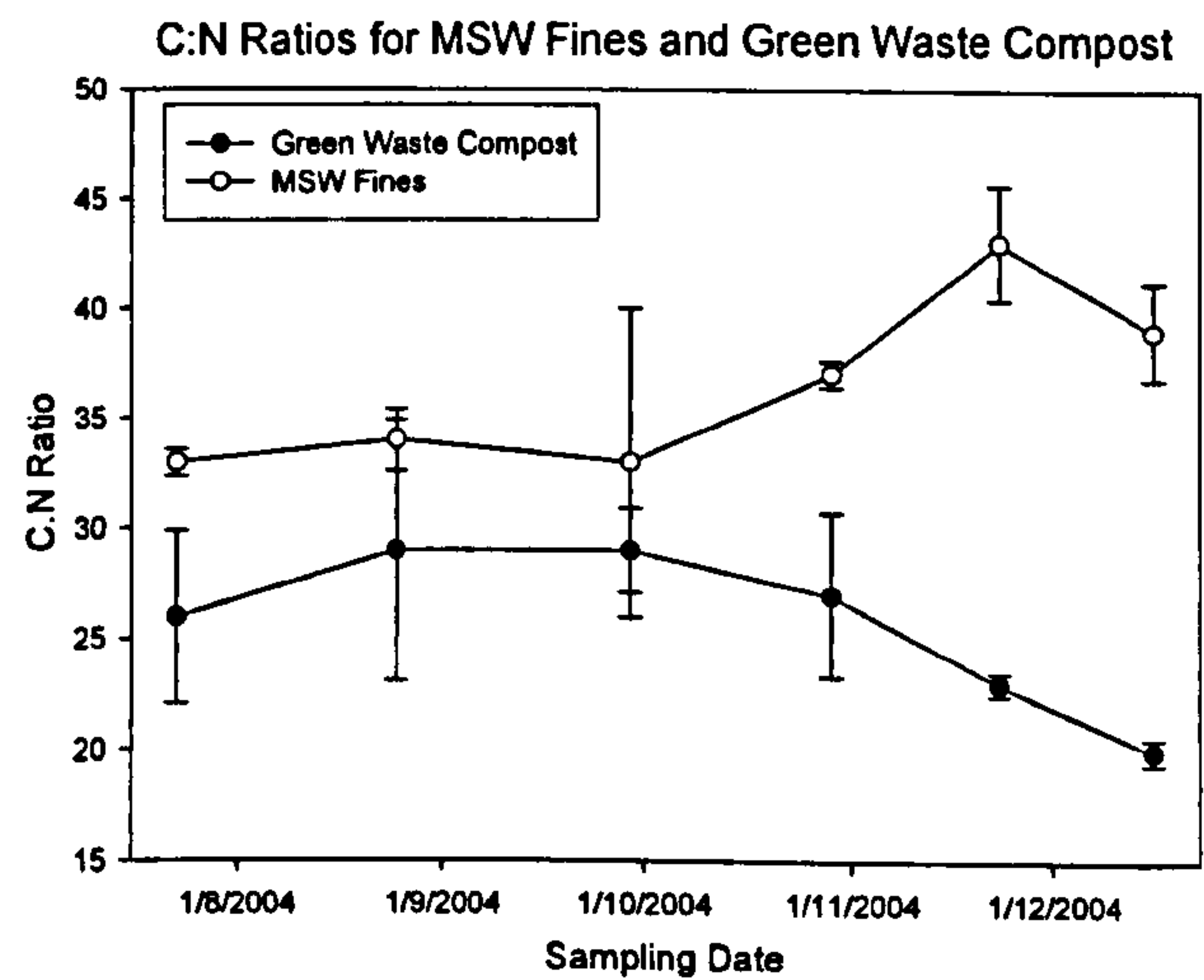


Figure 2.10: C/N ratio changes of MSW Fines and Green Waste sampled and tested over a 6 month period from two sites in Gwynedd, Wales. Bars are SEM.

MSW Fines had an increased C/N ratio in later samples compared to a decrease in Green Waste. No significant differences were found between the C/N ratios of the monthly MSW Fines and Green Waste samples. Overall, the C/N ratio was significantly higher in the MSW Fines than in the Green Waste ($p < 0.05$).

2.4.7. Waste feedstock nitrate

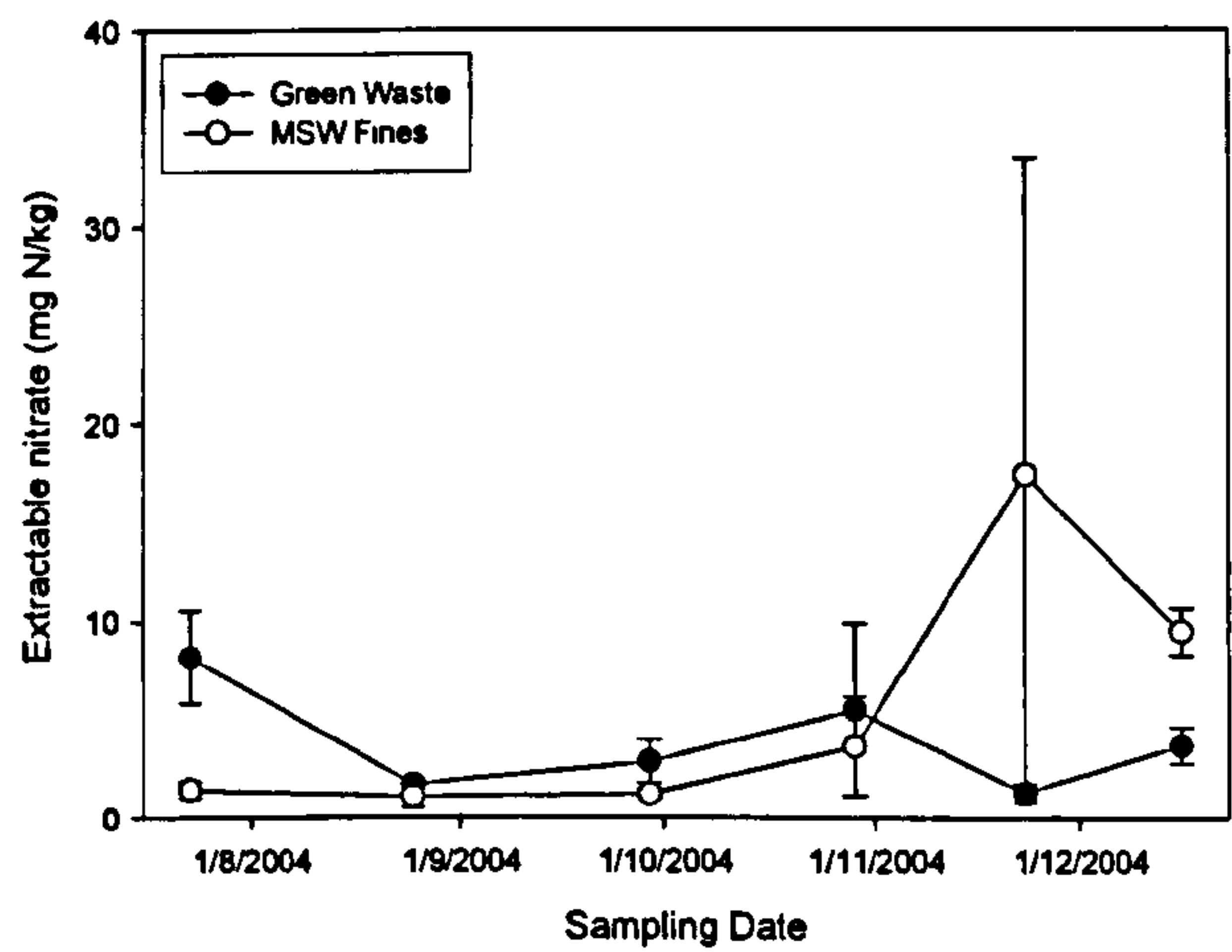


Figure 2.11: Changes in KCl-extractable NO_3^- concentration of MSW Fines and Green Waste sampled and tested over a 6 month period from two sites in Gwynedd, Wales. Bars are SEM.

There are very low levels of NO_3^- present in both wastes compared to NH_4^+ (see below) suggesting low levels of microbial nitrification in the wastes. There were no significant differences found between any of the MSW Fines or any of the Green Waste samples and no significant differences between the two materials ($p > 0.05$).

2.4.8. Waste feedstock KCl-extractable ammonium

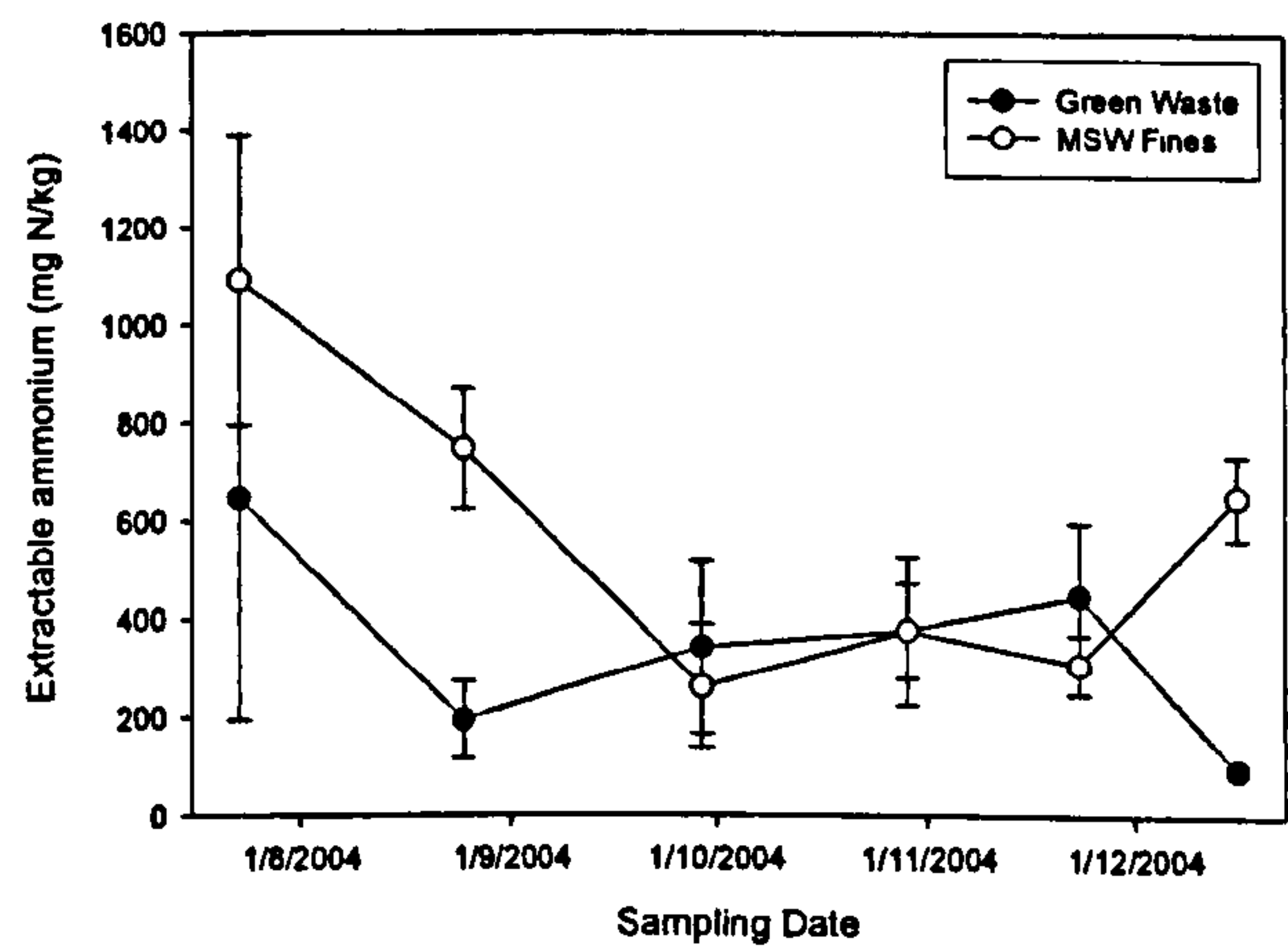


Figure 2.12: Changes in KCl-extractable NH_4^+ concentration of MSW Fines and Green Waste sampled and tested over a 6 month period from two sites in Gwynedd, Wales. Bars are SEM.

Initially, the MSW Fines tended to have a higher NH_4^+ concentration than the Green Waste. The concentrations recorded in both materials were far greater than the NO_3^- content. The July MSW Fines had a significantly higher NH_4^+ content than both the September and November samples. Conversely, no such differences were found between any of the Green Waste samples. No significant differences in NH_4^+ content existed between the materials ($p > 0.05$).

2.4.9. Waste feedstock phosphorus content

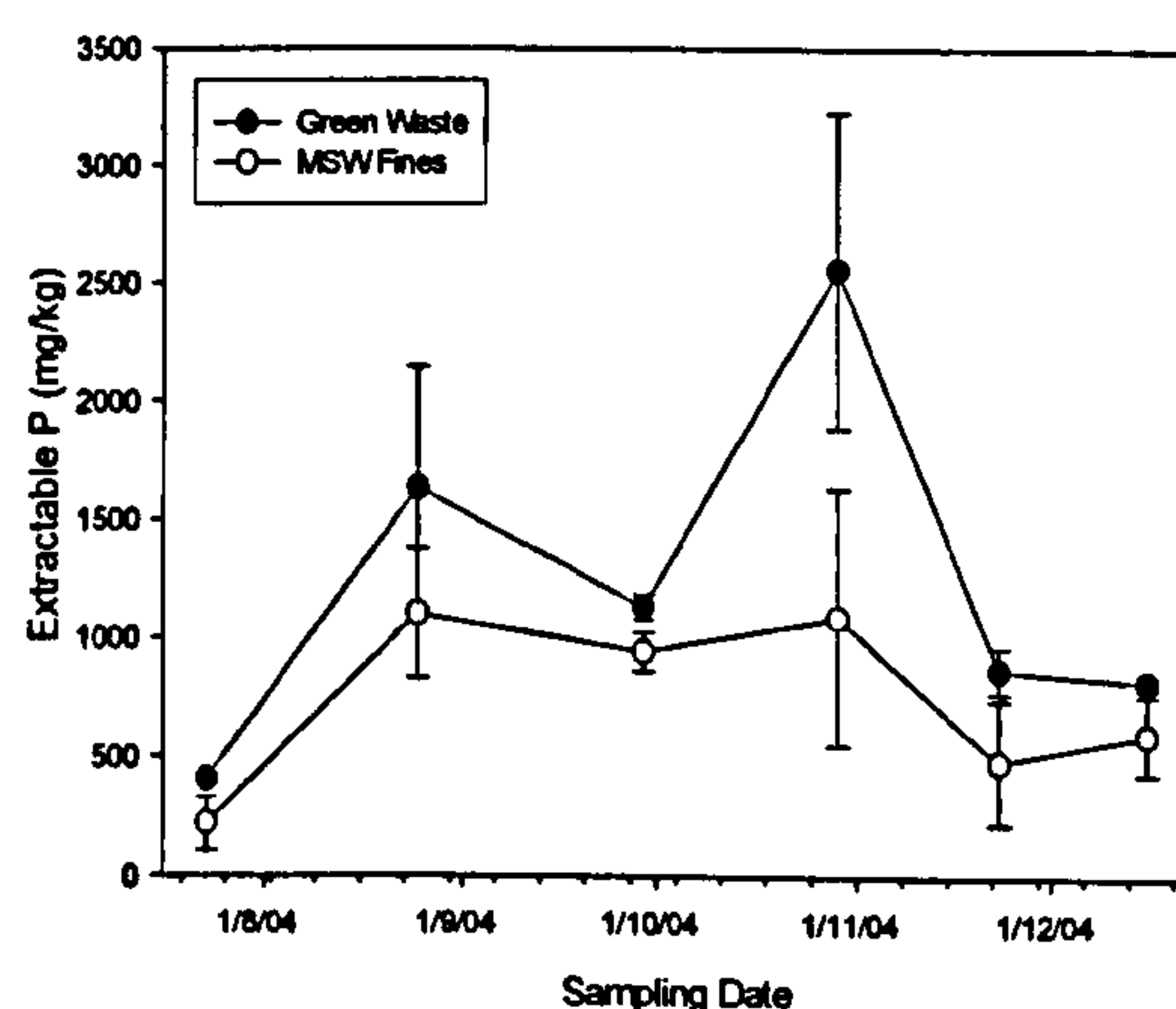


Figure 2.13: Changes in extractable P of MSW Fines and Green Waste sampled and tested over a 6 month period from two sites in Gwynedd, Wales. Bars are SEM.

Overall, the Green Waste contained a higher P concentration than the MSW Fines for the majority of the 6 month sampling period although the difference was small. No significant differences were identified between the two materials ($p > 0.05$). The MSW Fines samples taken in August and October were found to have significantly higher P concentrations than the July sample. For the Green Waste, the October sample was found to have a much greater P content than all of the other samples; the August sample was also observed to have significantly more P than the July sample.

2.4.10 Waste feedstock extractable metals

Ca was found to be higher in the MSW Fines than the Green Waste, with this difference being as much as four-fold in some samples. The opposite was apparent

for K. However, towards the latter monthly samplings, both materials were observed to have lower K and Ca levels than those taken initially. No significant differences were found either between the MSW Fines samples or between the Green Waste samples. Overall, the MSW Fines contained significantly greater levels of Ca and Na and lower K than the Green Waste ($p < 0.05$).

MSW Fines consistently contained higher levels of Na than the Green Waste, despite a spike in the latter during late October. The July MSW Fines sample contained significantly more Na than the other 5 samples, whilst the November and December samples had significantly less Na than the intervening months. The main difference found in the Green Waste was that November and December again had significantly lower Na contents than the other samples.

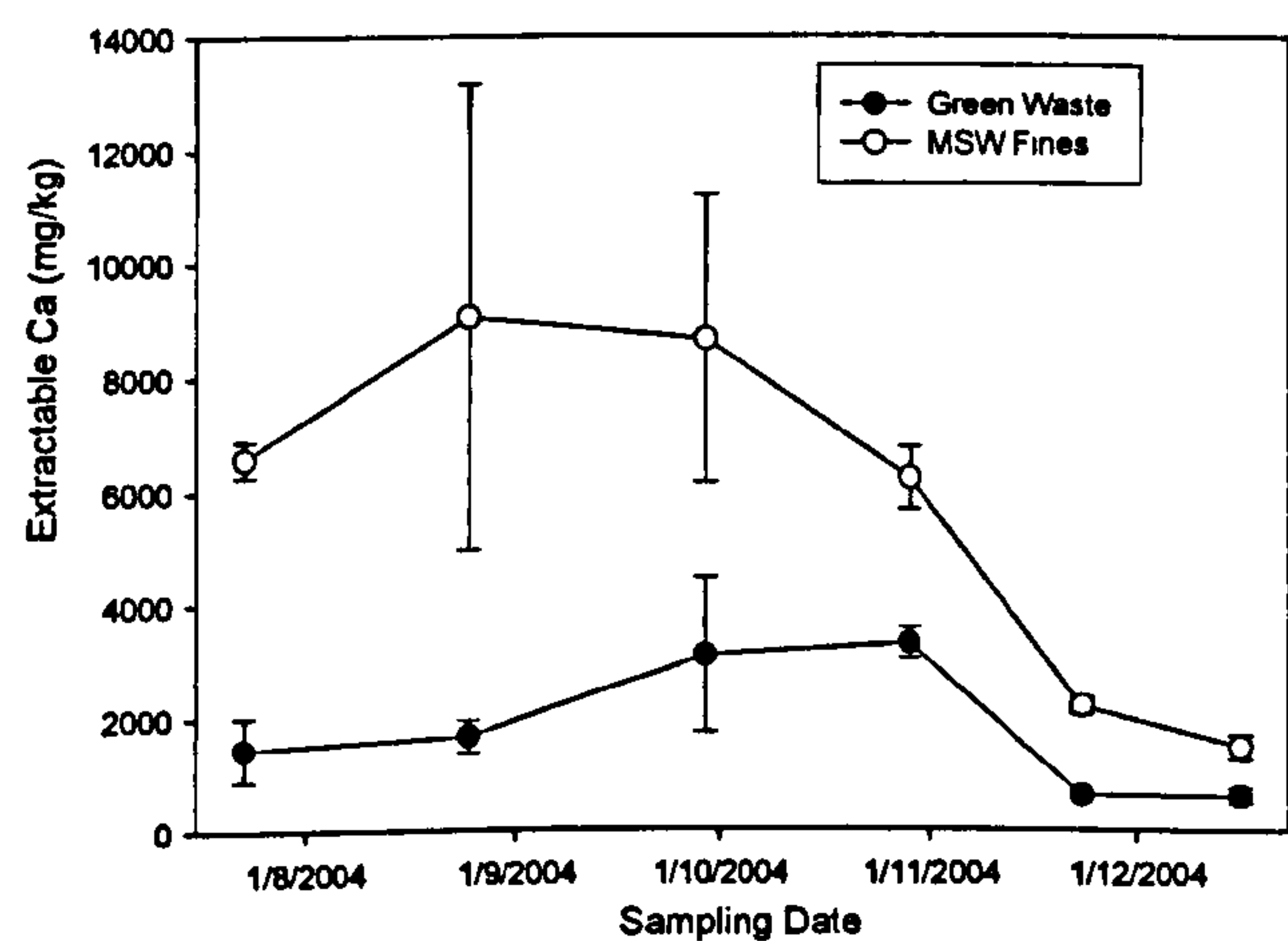


Figure 2.14: Changes in Ca concentration of MSW Fines and Green Waste sampled and tested over a 6 month period from two sites in Gwynedd, Wales. Bars are SEM.

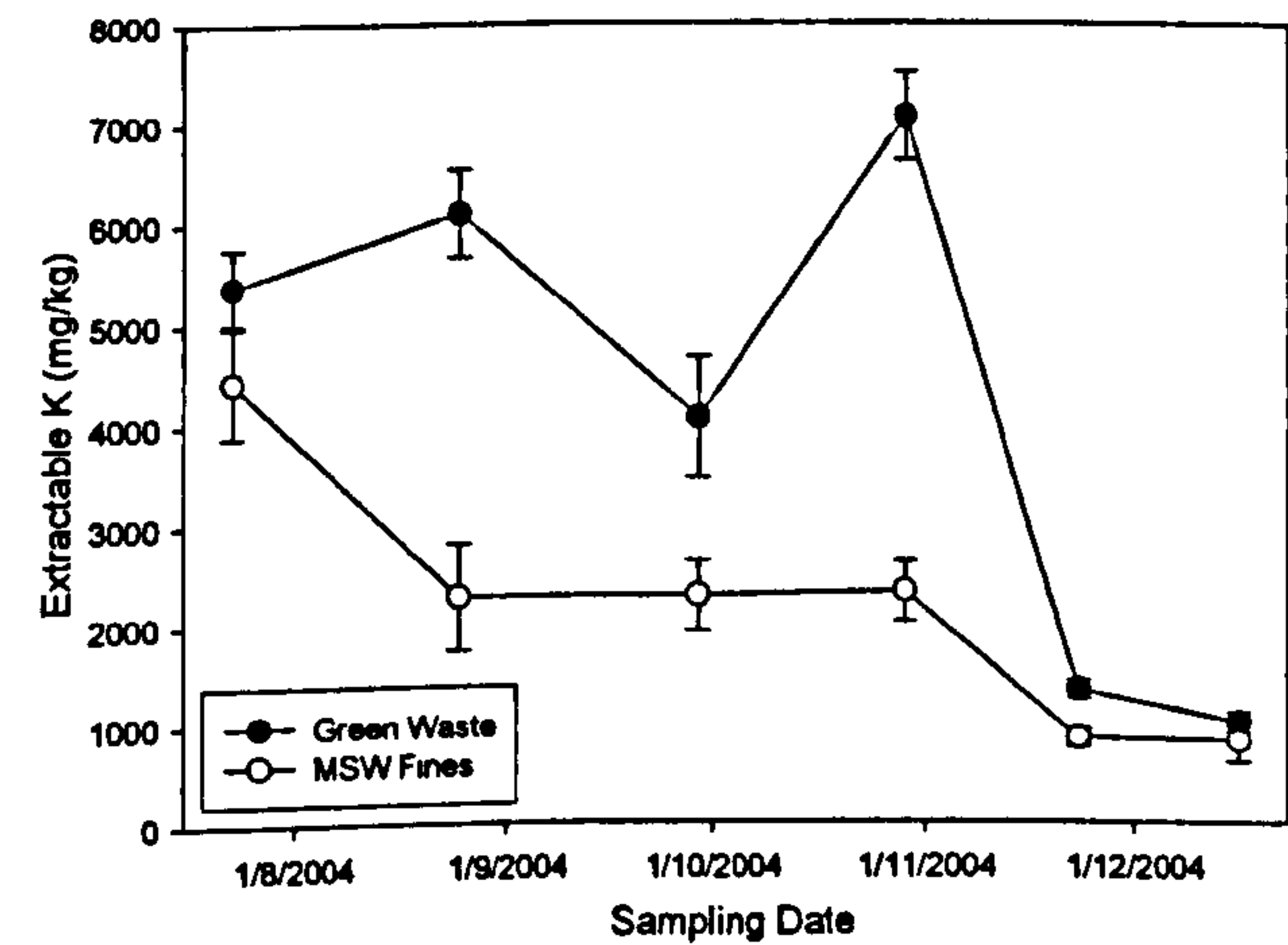


Figure 2.15: Changes in K concentration of MSW Fines and Green Waste sampled and tested over a 6 month period from two sites in Gwynedd, Wales. Bars are SEM.

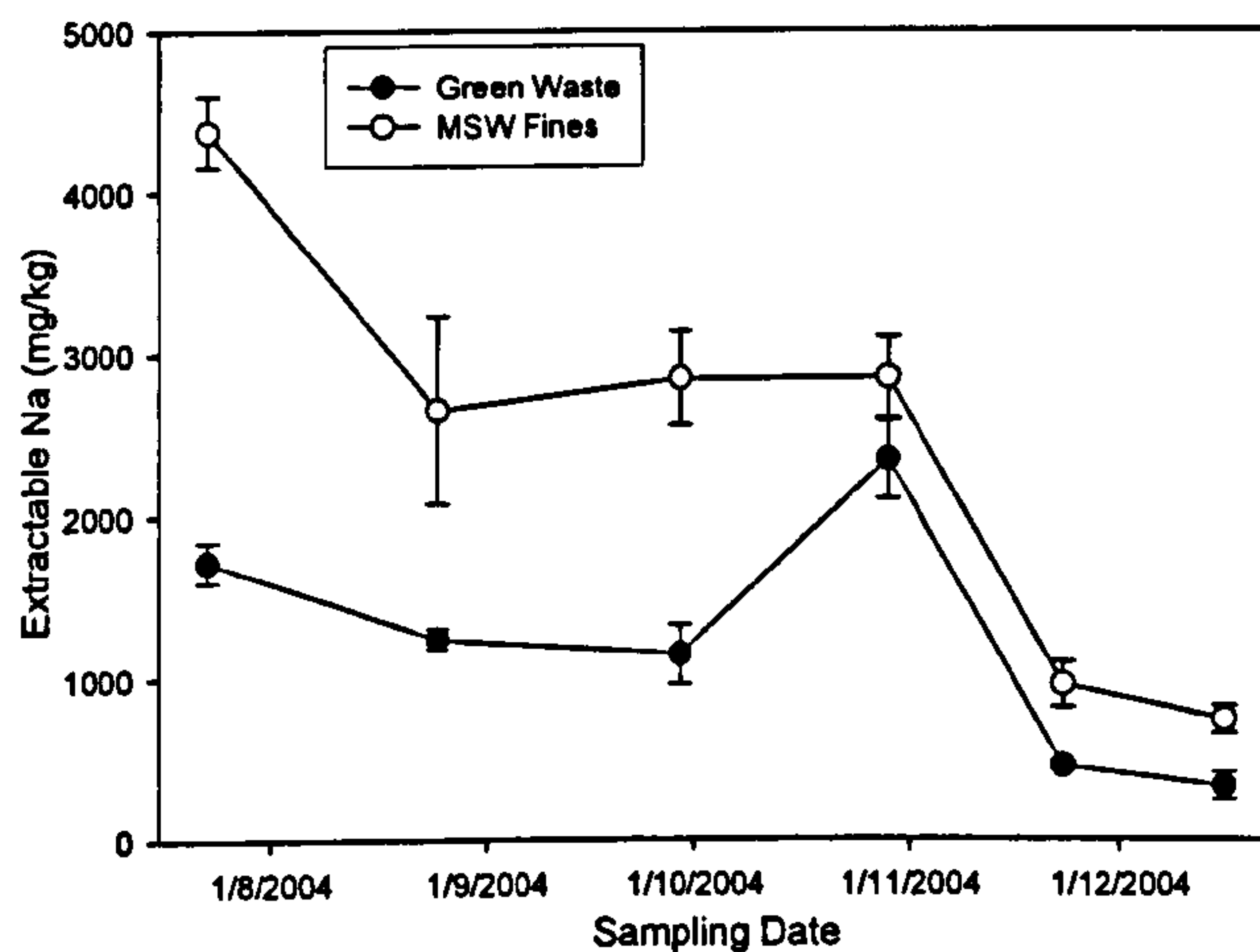


Figure 2.16: Changes in Na concentration of MSW Fines and Green Waste sampled and tested over a 6 month period from two sites in Gwynedd, Wales. Bars are SEM.

2.4.11. Waste feedstock bacterial counts

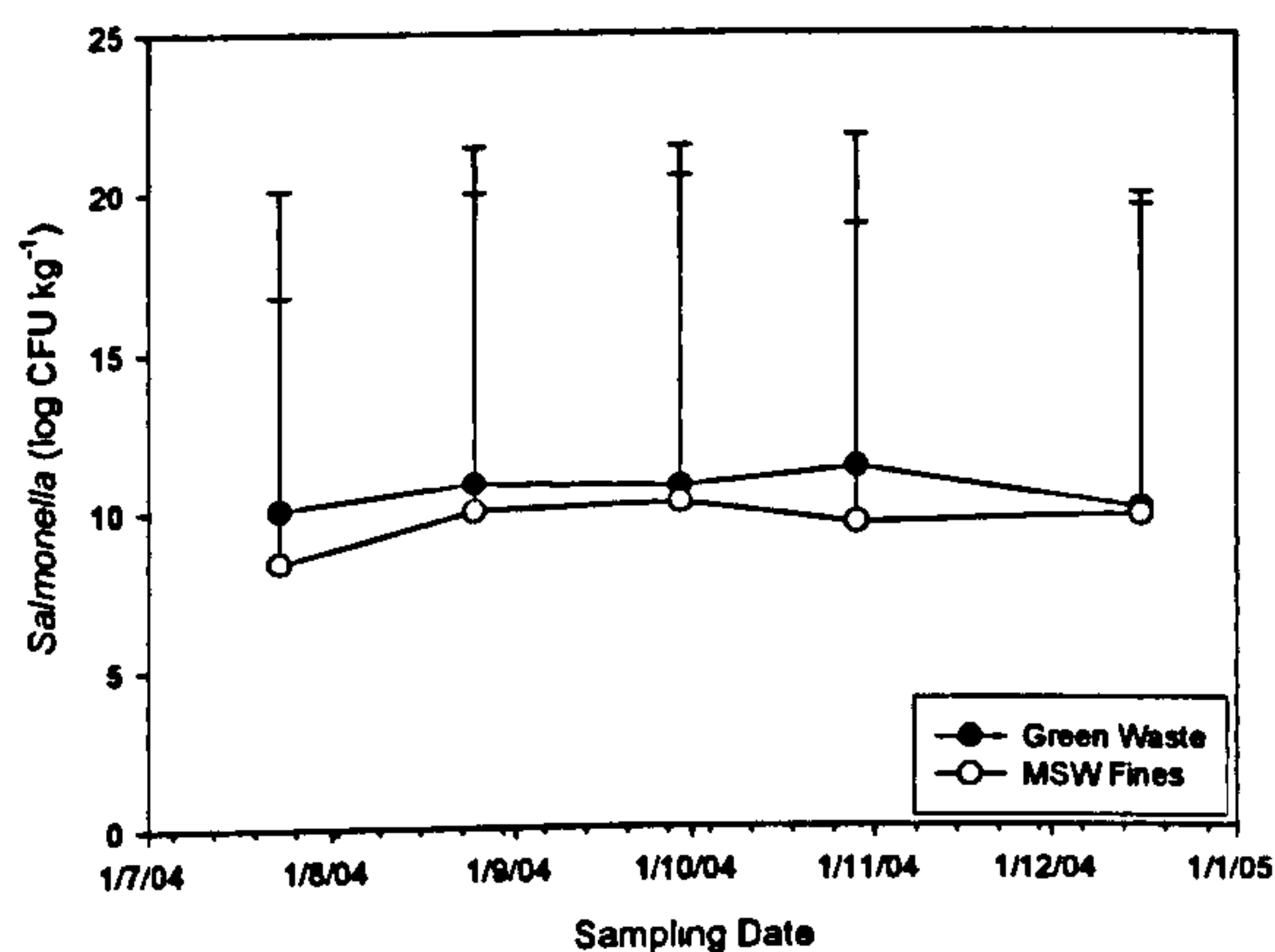


Figure 2.17: Changes in *Salmonella* spp. in of MSW Fines and Green Waste sampled and tested over a 6 month period from two sites in Gwynedd, Wales. Bars are SEM.

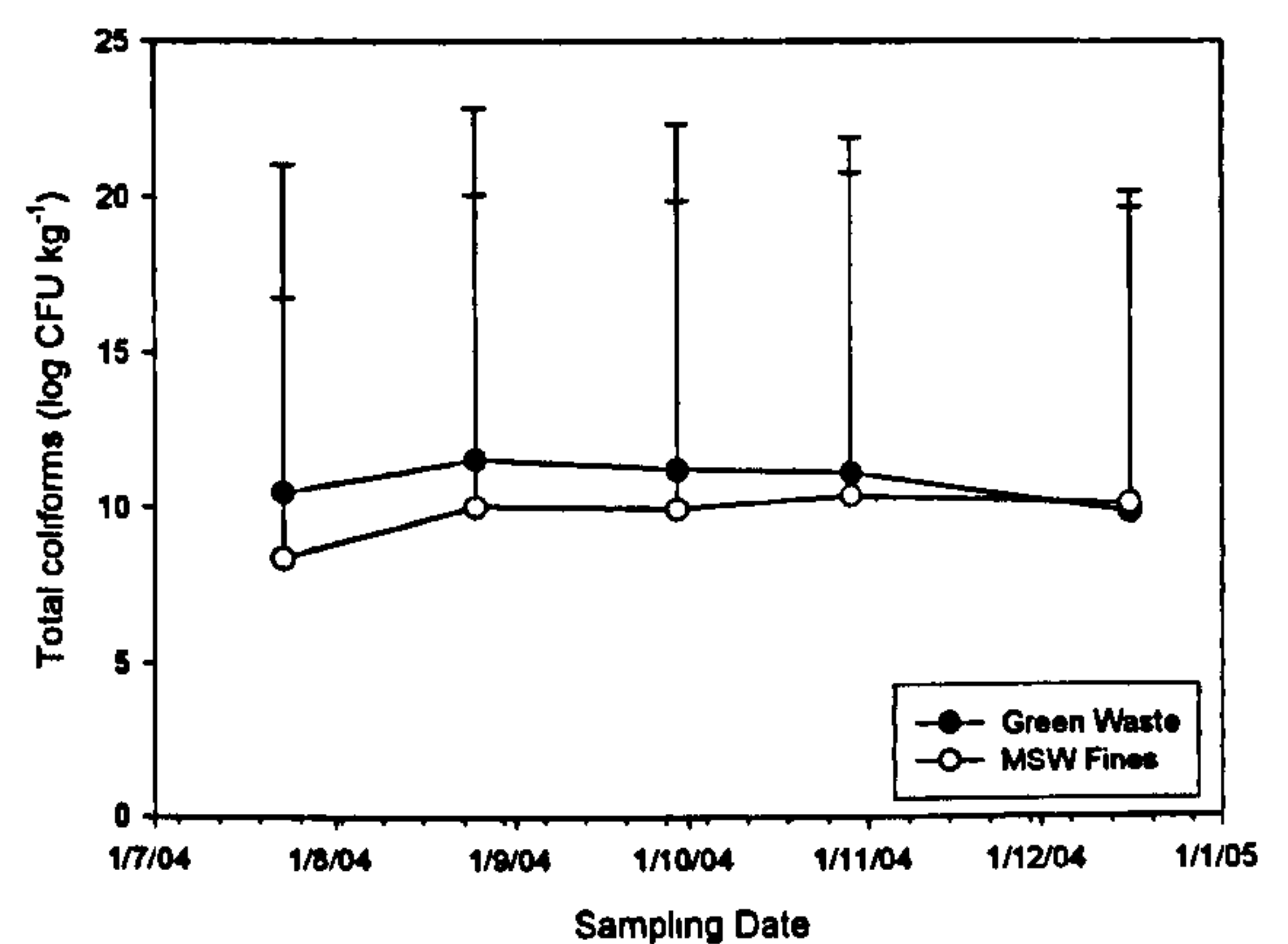


Figure 2.18: Changes in faecal coliforms of MSW Fines and Green Waste sampled and tested over a 6 month period from two sites in Gwynedd, Wales. Bars are SEM.

Salmonella and coliform counts were highly variable in both feedstocks in all the months sampled. Consequently, there were no significant differences between samples.

2.4.12. Waste feedstock fractionation data

MSW Fines were found to contain an average of $78 \pm 3\%$ biodegradable matter (by weight) in all samples along with a large amount of glass (ca. $15 \pm 3\%$; Figure 2.19). Here, biodegradable matter was classed as including paper and card, hence there was no separate data recorded for each of these materials.

Very little physical contamination was found in the Green Waste with only a small amount of plastic ($1.4 \pm 0.5\%$) or metal ($0.17 \pm 0.17\%$) present. In the majority of samples, no contamination was found. A strict policy is implemented by the local authority to prevent the disposal of non-green waste material into green waste collections; this effectively reduces the potential, and actual, physical contamination by metals, plastics and glass, often eliminating it all together.

The plastic ($p < 0.05$) and glass ($p < 0.01$) contents of the MSW Fines were higher than those in the Green Waste whilst the biodegradable fraction was higher in the Green Waste ($p < 0.01$); the results for metallic content did not show any such differences ($p > 0.05$).

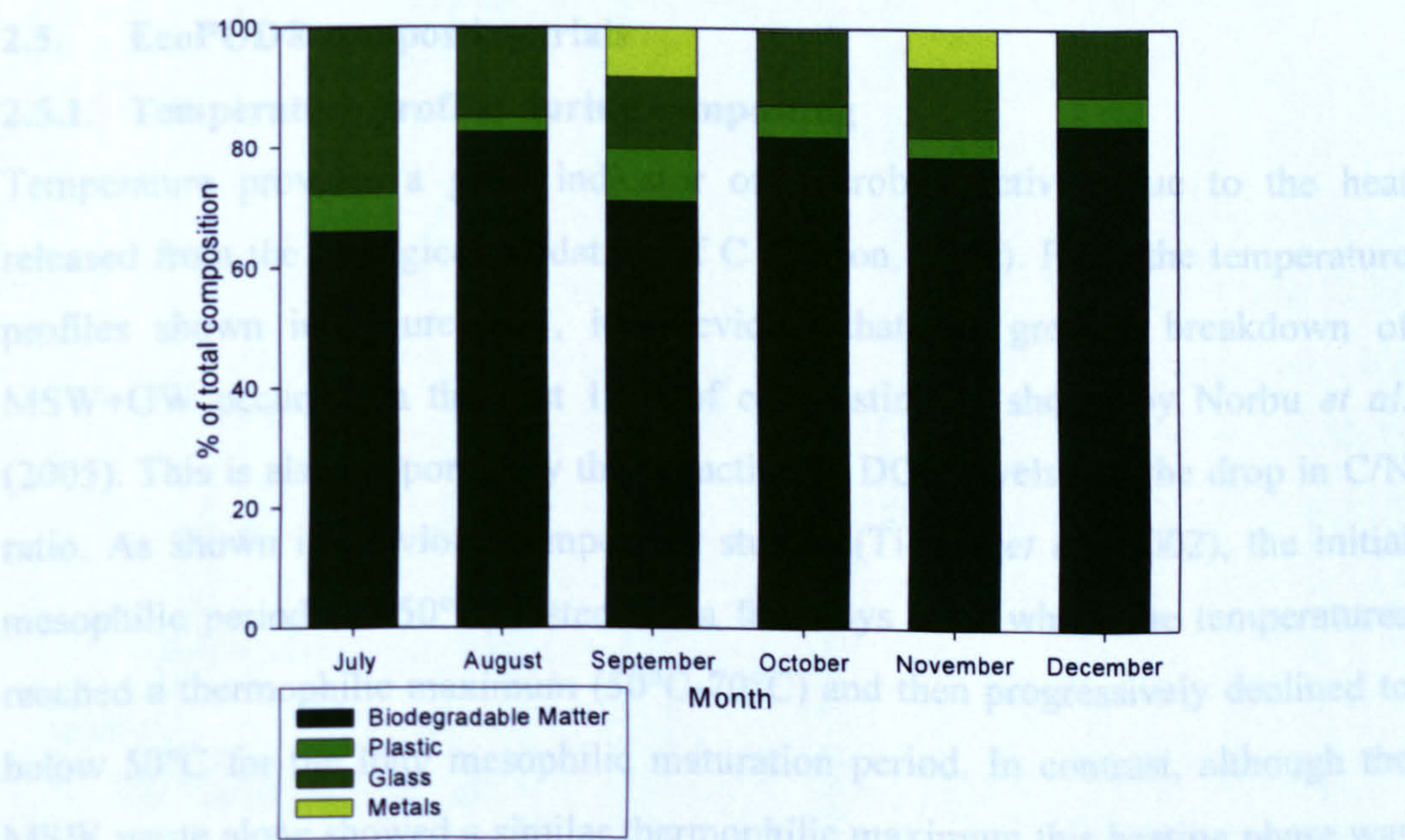


Figure 2.19: Changes in constituent fractions of MSW Fines sampled and tested over a 6 month period from a site in Gwynedd, Wales.

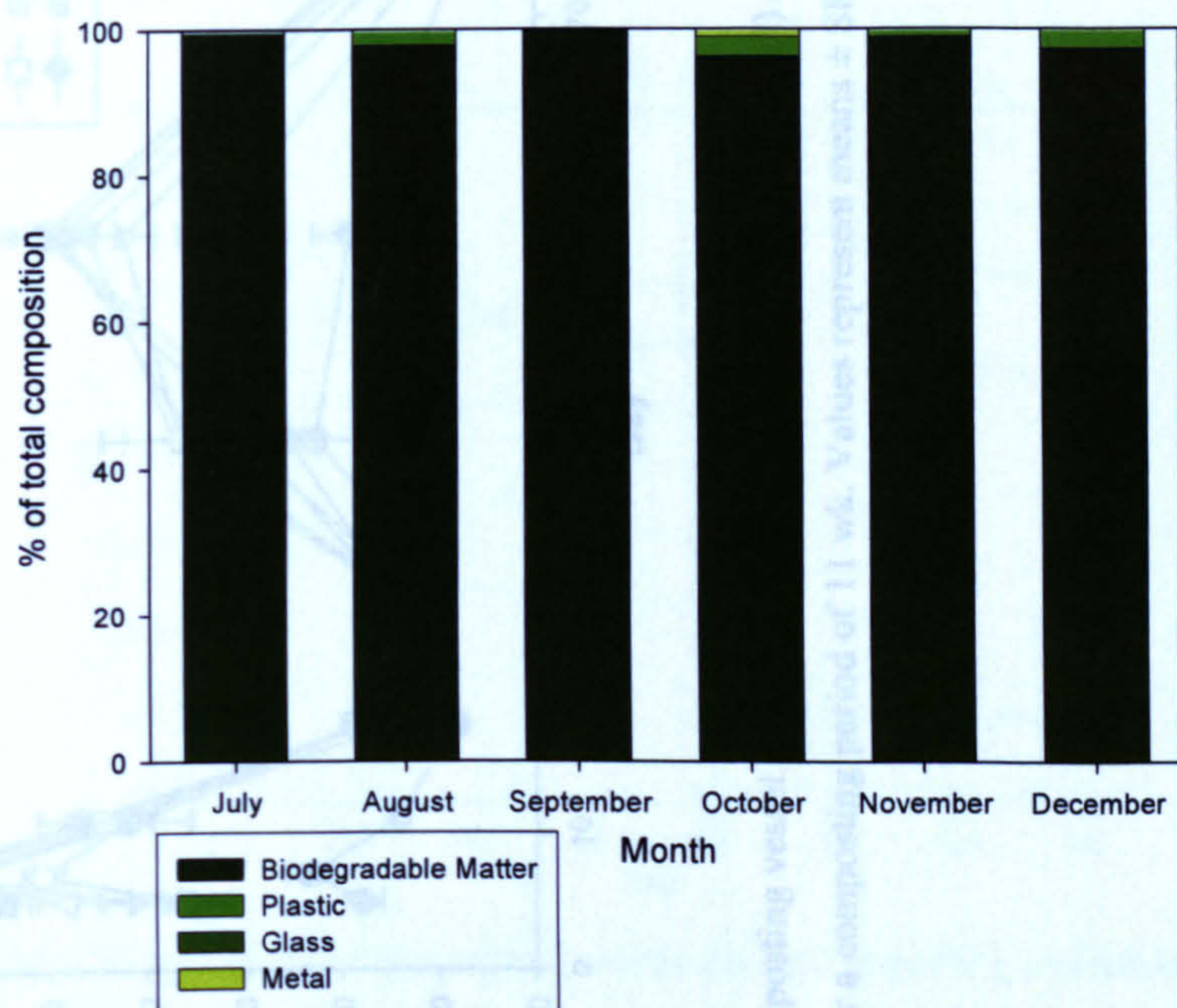


Figure 2.20: Changes in Constituent Fractions of Green Waste over a 6 month period sampled from a site in Gwynedd, Wales.

2.5. EcoPOD® composting trials

2.5.1. Temperature profiles during composting

Temperature provides a good indicator of microbial activity due to the heat released from the biological oxidation of C (Barton, 1979). From the temperature profiles shown in Figure 2.21, it is evident that the greatest breakdown of MSW+GW occurred in the first 10 d of composting as shown by Norbu *et al.* (2005). This is also supported by the reduction in DOC levels and the drop in C/N ratio. As shown in previous composting studies (Tilston *et al.*, 2002), the initial mesophilic period (30-50°C) lasted for a few days after which the temperatures reached a thermophilic maximum (50°C-70°C) and then progressively declined to below 50°C for the long mesophilic maturation period. In contrast, although the MSW waste alone showed a similar thermophilic maximum this heating phase was not sustained with temperatures rapidly falling after 7 d. Following this fall, the compost showed a secondary heating phase peaking about 2 months after the onset of composting. Statistical analysis revealed few differences in temperature with vessel depth over the course of the composting period for an individual treatment.

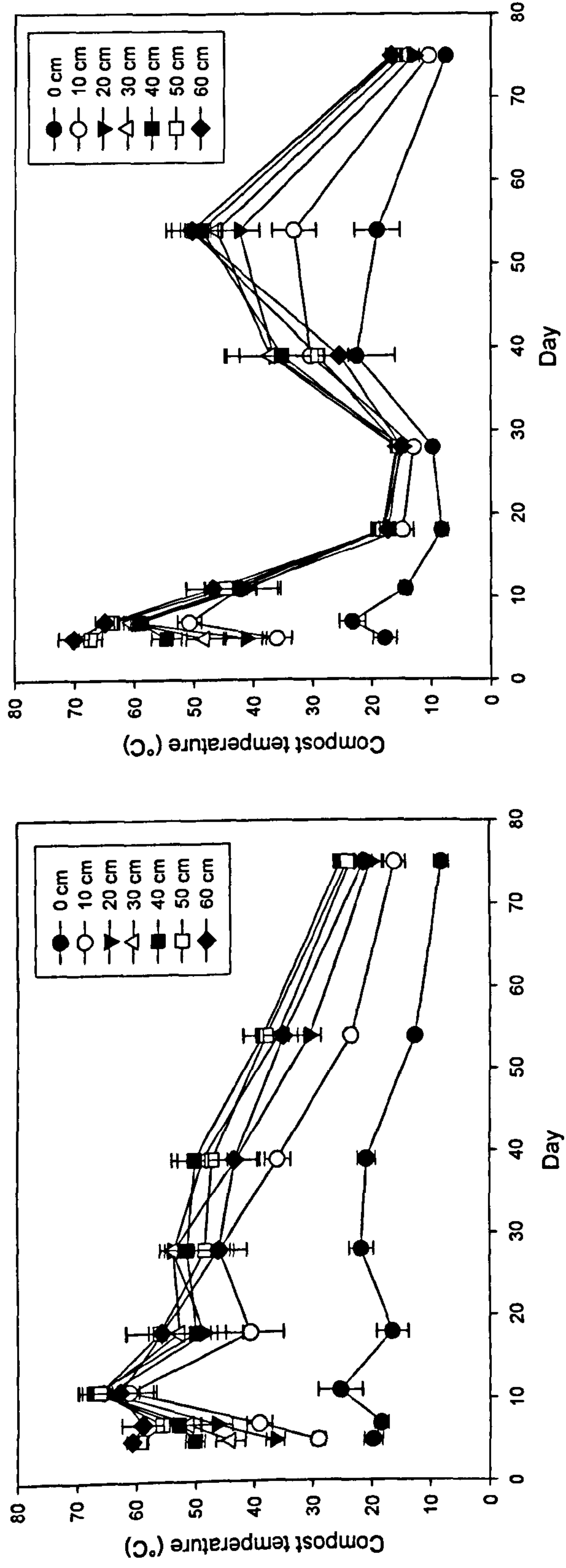


Figure 2.21: Temperature profile for different depths within the EcoPOD® composting vessel for the MSW Fines compost (left hand panel) and MSW Fines plus Green Waste (right hand panel) over a composting period of 11 wk. Values represent means \pm SEM.

2.5.2. Compost pH

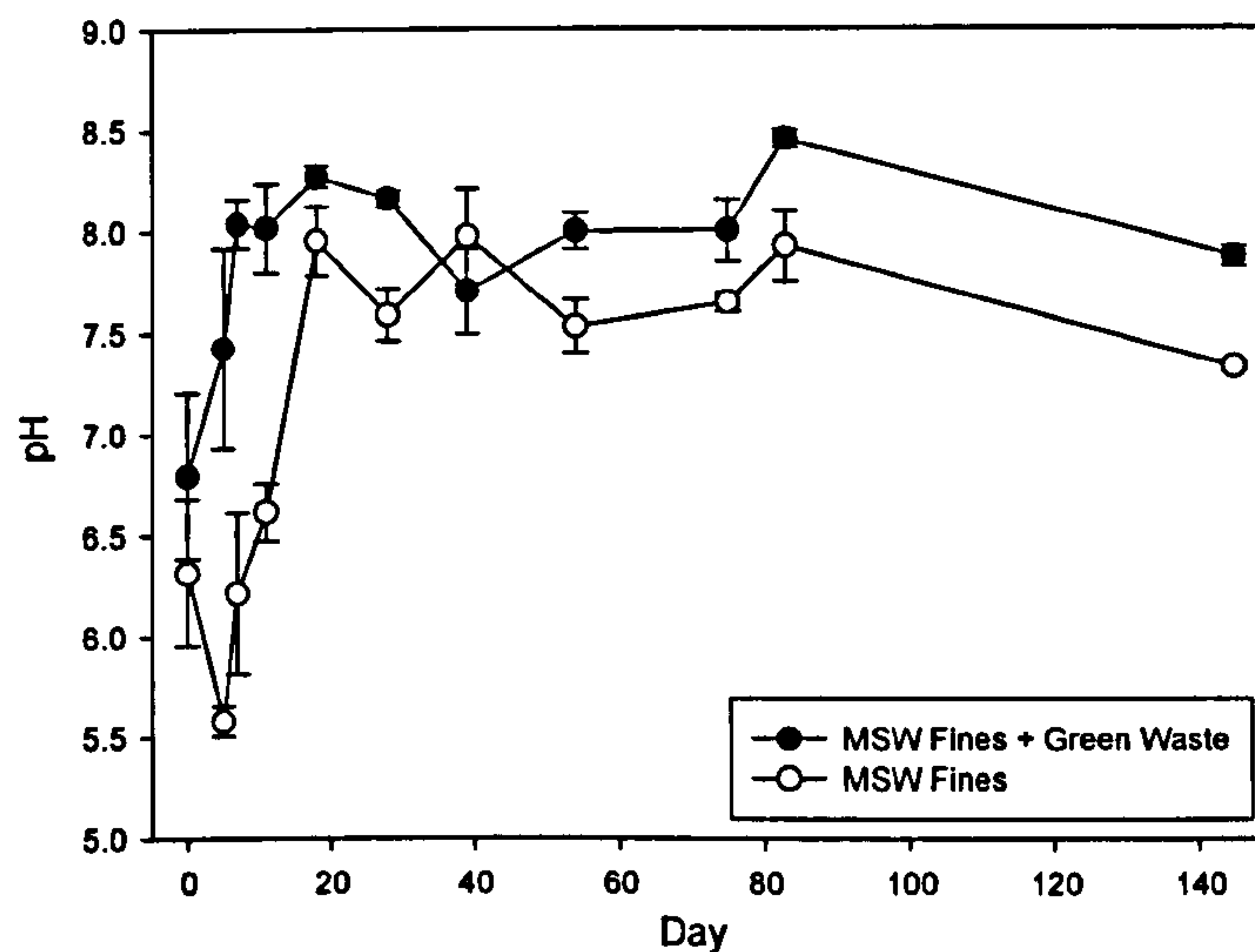


Figure 2.22: Changes in pH of two organic wastes during composting and subsequent maturation over a 5 month period. Values represent means \pm SEM.

During the period of composting (0-83 d), the pH of both the MSW Fines and MSW+GW mixture waste rose significantly. The first 3 wk of composting showed a distinct increase in the pH of both materials. The pH was generally higher in the MSW+GW mixture for the majority of the composting and maturation period.

Significant differences were found between the individual samples of the MSW Fines, with these being most prominent between the material that been composted for less than 3 wk and those that had undergone a longer treatment. For the MSW+GW mixture, some differences were found between samples but these differences were identified as not being related to the length of composting period. The two treatments had significantly different pH values across the trial, with the MSW+GW mixture being higher ($p < 0.005$).

2.5.3. Compost electrical conductivity

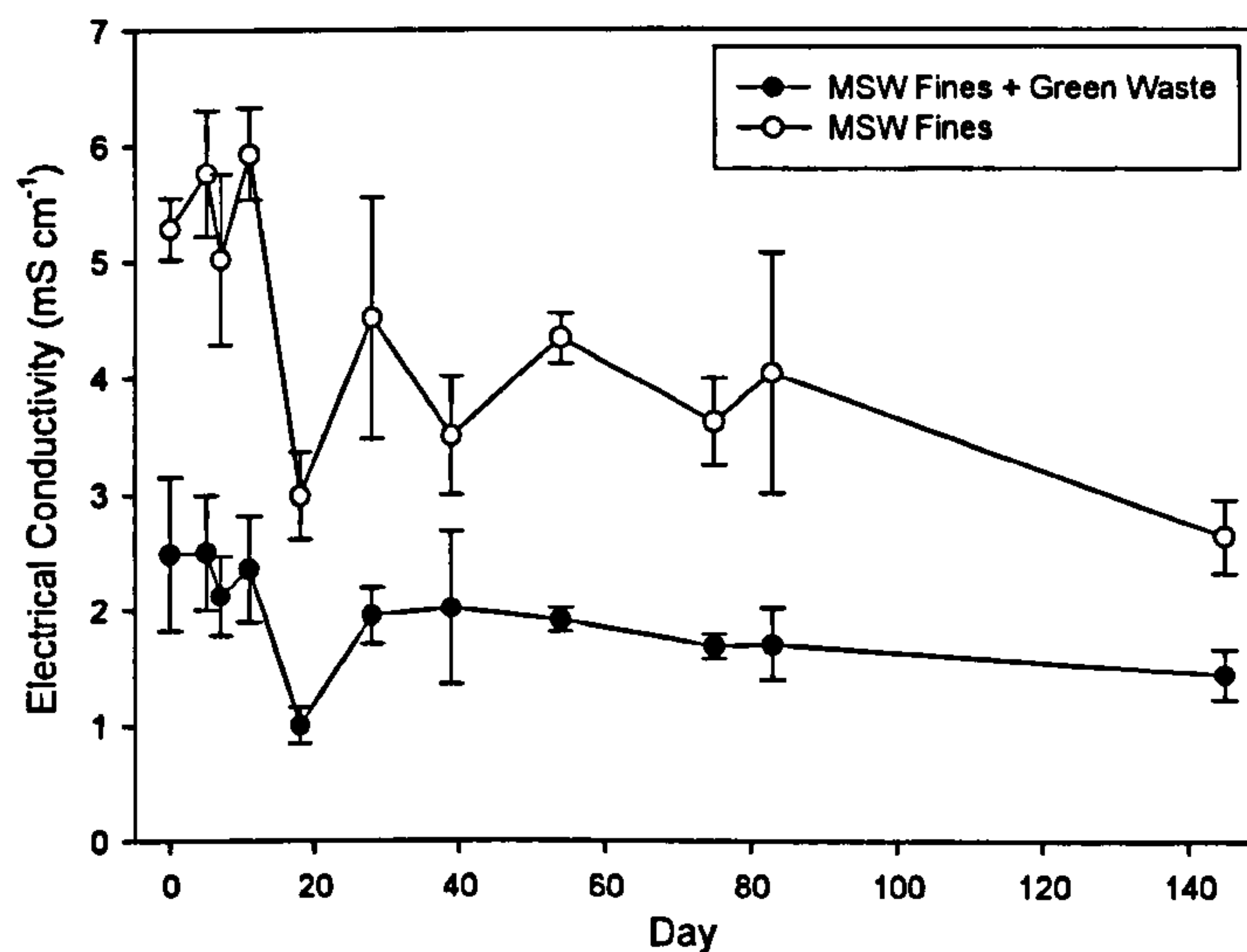


Figure 2.23: Changes in EC of two organic wastes during composting and subsequent maturation over a 5 month period. Values represent means \pm SEM.

EC was found to decrease in both materials during composting, with the MSW Fines having a much higher initial value than the MSW+GW mixture ($5.2 \pm 0.3 \text{ mS cm}^{-1}$ compared to $2.5 \pm 0.7 \text{ mS cm}^{-1}$). Such a change can be related to a fall in the levels of soluble salts present within the two materials, although MSW Fines still had a higher value at the end of maturation than the initial value (0 d) for Green Waste. Manios *et al.* (2004) found that composting plant-based materials (Olive leaves) resulted in a material with an EC of 3.7 mS cm^{-1} and that changes in the levels of this correlates with NO_3^- and K salts. The values obtained in the EcoPOD® study are similar to those for MSW Fines that had been composted for 11 wk in the study by Manios *et al.* (2004).

The EC readings of two of the early MSW Fines samples were significant higher than those of any subsequent samples. No such differences were found for the MSW+GW mixture. The EC of the Fines was significantly higher than the MSW+GW mixture ($p < 0.001$).

2.5.4. Compost KCl-extractable nitrate

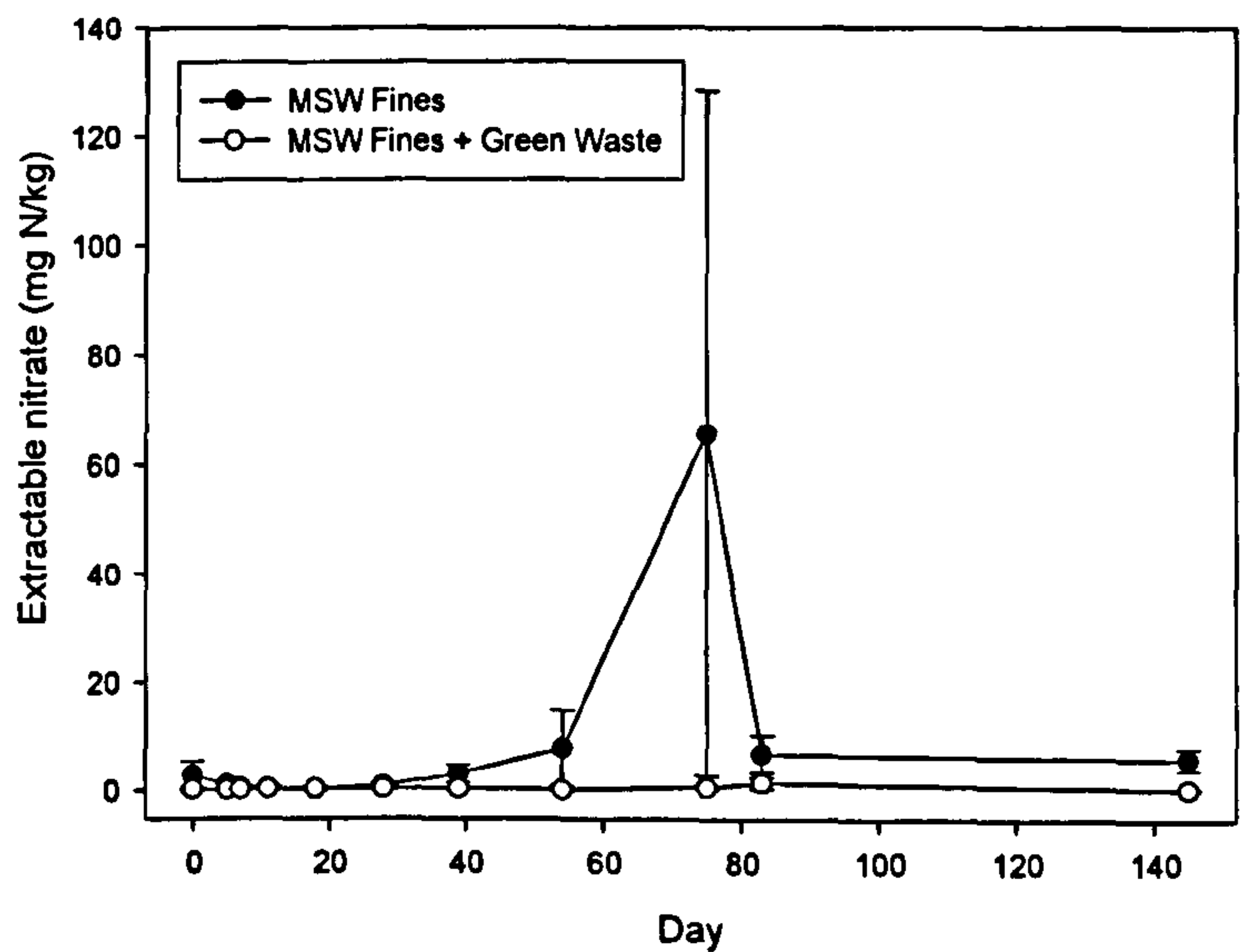


Figure 2.24: Changes in the nitrate content of two organic wastes during composting and subsequent maturation over a 5 month period. Values represent means \pm SEM.

Very little NO_3^- was found in the MSW+GW mixture for the duration of composting (83 d); although there was a slight increase observed during the early stages of maturation. MSW Fines was recorded as having a higher NO_3^- content throughout, often significantly higher, and was far more prone to variation between replicates than the mixed waste (shown by the large error bars). Such a low overall concentration of NO_3^- in both materials is most likely a result of a poor rate of conversion of NH_4^+ to NO_3^- due to unfavourable conditions within the composting vessel. No significant differences in NO_3^- levels were found either within or between treatments.

2.5.5. Compost KCl-extractable ammonium

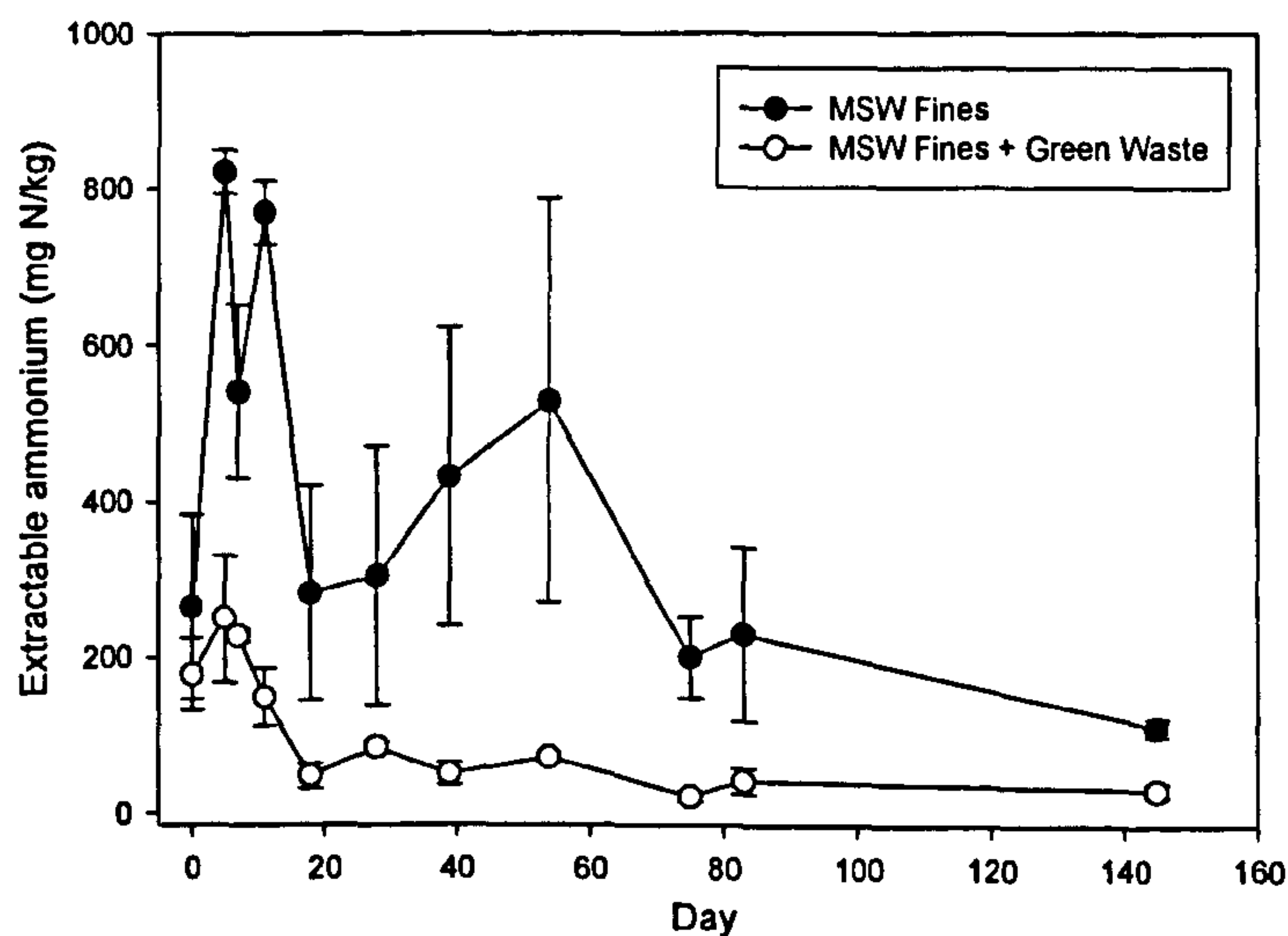


Figure 2.25: Changes in ammonium content of two organic wastes during composting and subsequent maturation over a 5 month period. Values represent means \pm SEM.

NH_4^+ concentration in the MSW+GW mixture decreased during composting and maturation, with very little within-sample variation. Comparatively, the MSW Fines compost increased in NH_4^+ during the initial stages of composting, before decreasing from around week 6 to week 12 in the latter stages of composting. The MSW Fines had significantly greater levels of NH_4^+ than the MSW+GW compost ($p < 0.001$).

2.5.6. Compost dissolved organic carbon (DOC)

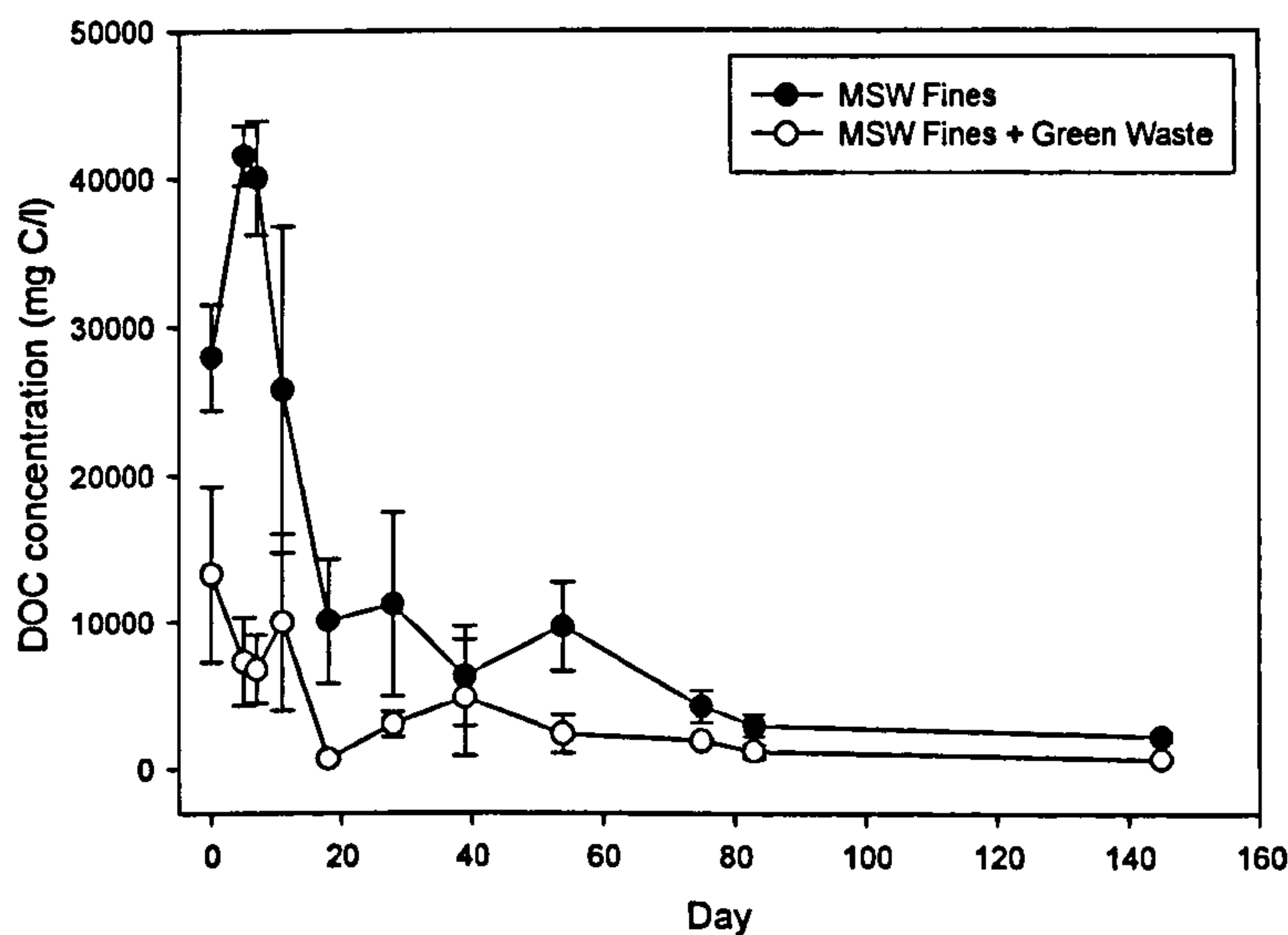


Figure 2.26: Changes in the DOC content of two organic wastes during composting and subsequent maturation over a 5 month period. Values represent means ± SEM.

The clear decrease in DOC over time indicates that the decomposition of organic carbon in the MSW Fines was at its fastest during the first 21 d after which the rate slowed. DOC also decreased in the MSW+GW compost although less so than in the Fines alone.

Significant differences were found between the two different materials with the MSW Fines possessing higher DOC concentrations ($p < 0.01$). Generally, the MSW Fines analysed during the first two weeks of composting were found to contain significantly greater DOC than the final sample (4 months composted). No such differences were identified between any of the MSW+GW compost samples.

2.5.7. Compost dissolved organic nitrogen (DON)

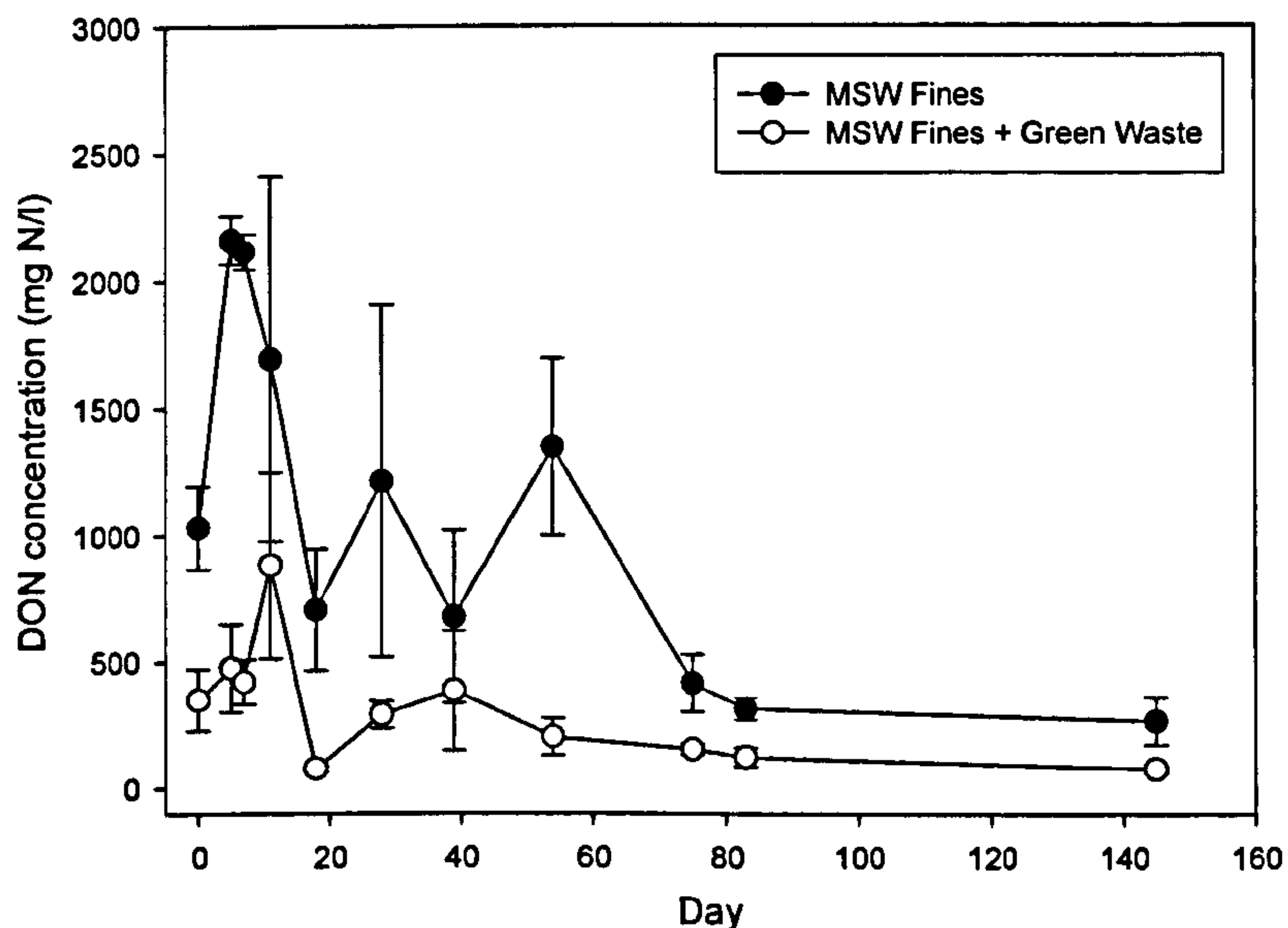


Figure 2.27: Changes in the DON content of two organic wastes during composting and subsequent maturation over a 5 month period. Values represent means \pm SEM.

DON was significantly higher in the MSW Fines than in the MSW+GW compost ($p < 0.001$). The only statistical difference was that the second sample (5 d) contained significantly greater amounts of DON than the final two MSW Fines samples. For the MSW+GW compost, it was shown that differences existed between two sequential samples; that taken at 4 wk composting contained significantly higher DON levels than the sample taken one week later. It was also evident that the sample at 4 wk composting was also significantly higher than the final sample (143 d).

2.5.8. Compost moisture content

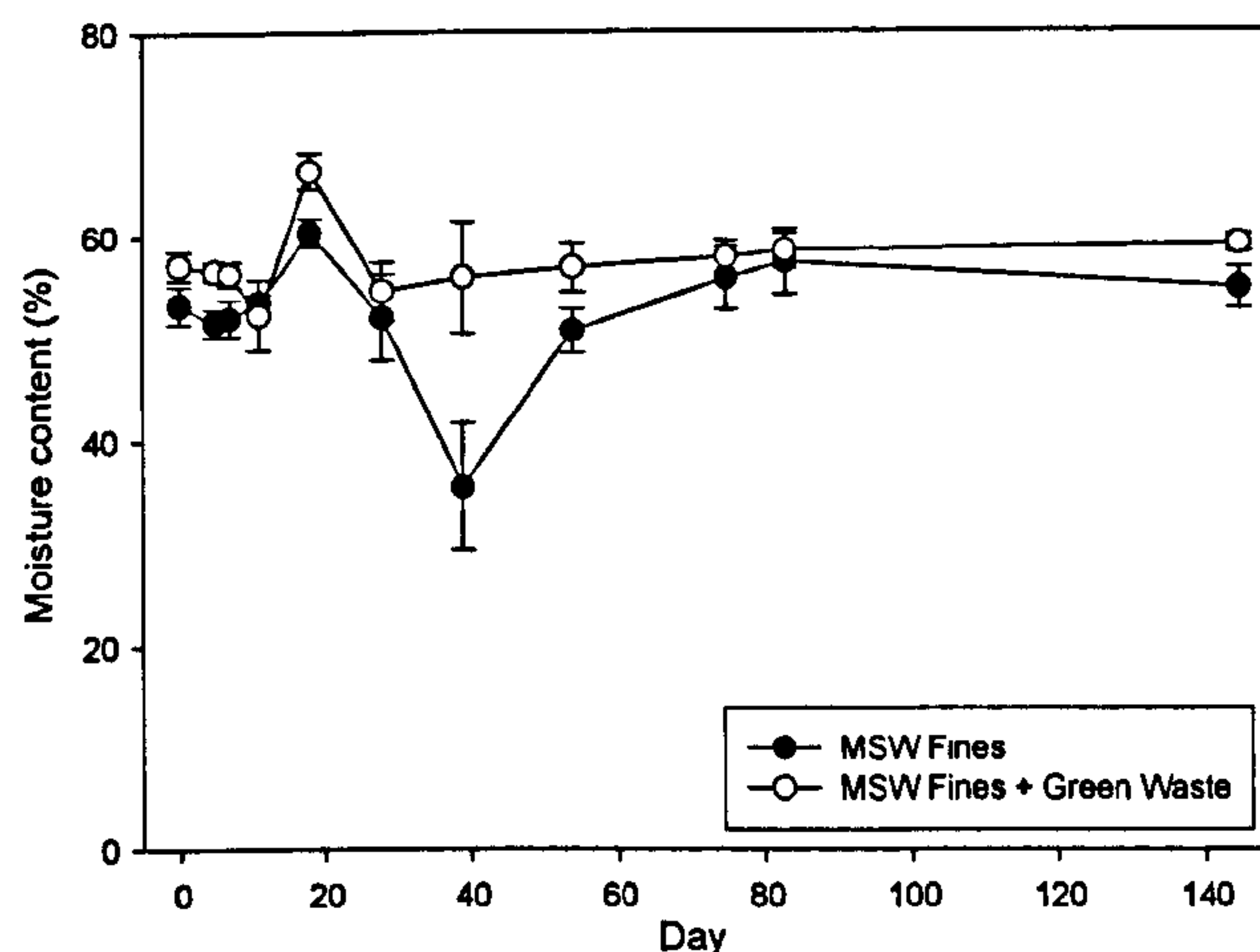


Figure 2.28: Changes in the moisture content of two organic wastes during composting and subsequent maturation over a 5 month period. Values represent means \pm SEM.

Moisture content of the substrates remained almost constant during the composting period. As no water was added to the vessel, this stability in moisture shows a good net balance between metabolic water produced from hydrolysis reactions in the EcoPOD® alongside that lost as steam in the outflow gas. No significant differences were identified between any of the MSW Fines samples taken here. The result is the same for the MSW+GW compost, despite there being large variations between week 3 and week 6. Despite this, the MSW+GW compost contained significantly higher moisture contents than the MSW Fines ($p < 0.05$).

2.5.9. Compost organic matter content

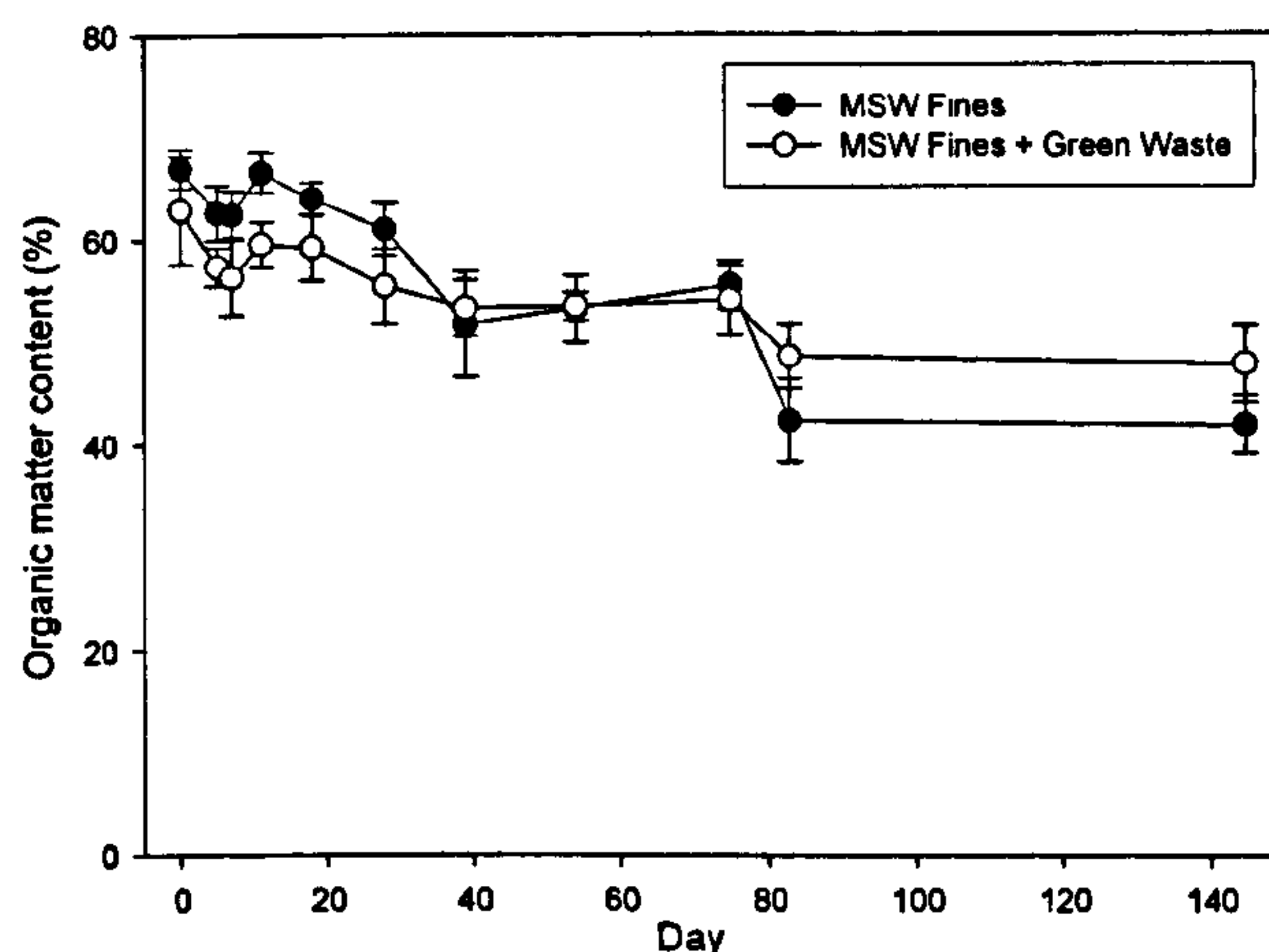


Figure 2.29: Changes in the organic matter content of two organic wastes during composting and maturation over a 5 month period. Values represent means \pm SEM.

Both composts showed a similar temporal trend in organic matter content which progressively fell over the active composting phase, becoming relatively constant during the maturation phase. For the MSW Fines data, it was shown that the final two samples (weeks 12 and 21) were found to contain significantly lower amounts of organic matter than those samples taken in the first 4 wk of composting. No such differences were identified in the data for the MSW+GW compost. No differences were found between the two compost types ($p > 0.05$).

2.5.10. Compost phosphorous

Acetic acid extractable P was found to decrease in both materials during composting, with the rate of decline being fastest in the first two weeks. The gradual fall in P concentrations continued through the maturation period. Statistical analysis revealed that the MSW Fines has significantly more acetic acid extractable P than the MSW+GW compost ($p < 0.05$).

Water soluble P followed a similar temporal trend to acetic acid-extractable P, albeit 3-fold lower. The MSW Fines possessed a higher P concentration than the MSW+GW compost during the active composting phase, but following maturation, the reverse was true. The first sample of MSW Fines contained significantly greater P levels than the rest of the subsequent samples. However, no such differences

were found for the MSW+GW compost. As occurred for the acetic acid-extractable P, the MSW Fines had significantly more P than the MSW+GW compost ($p < 0.05$).

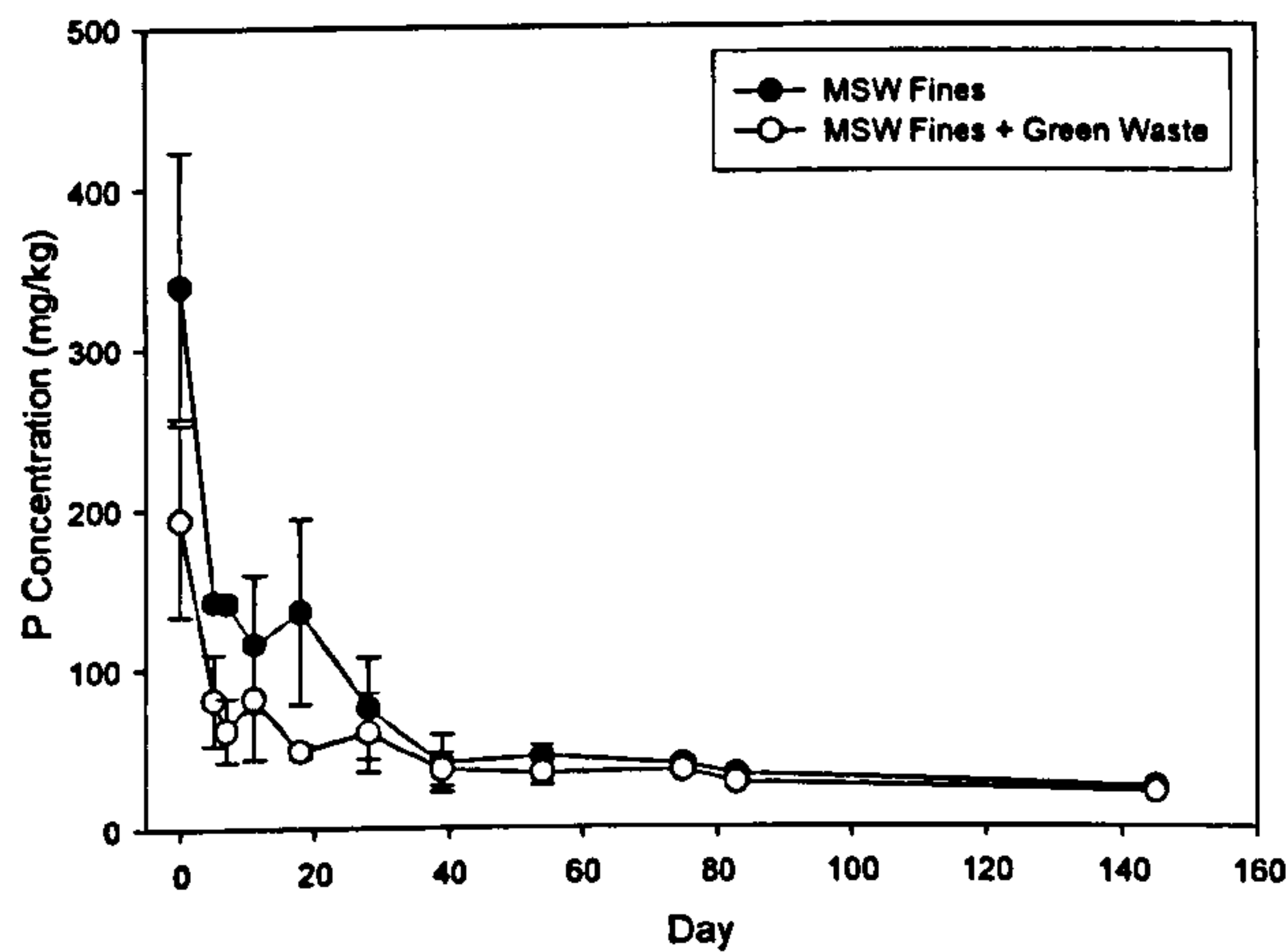


Figure 2.30: Changes in acetic acid extractable P content of two organic wastes during composting and maturation over a 5 month period. Values represent means \pm SEM.

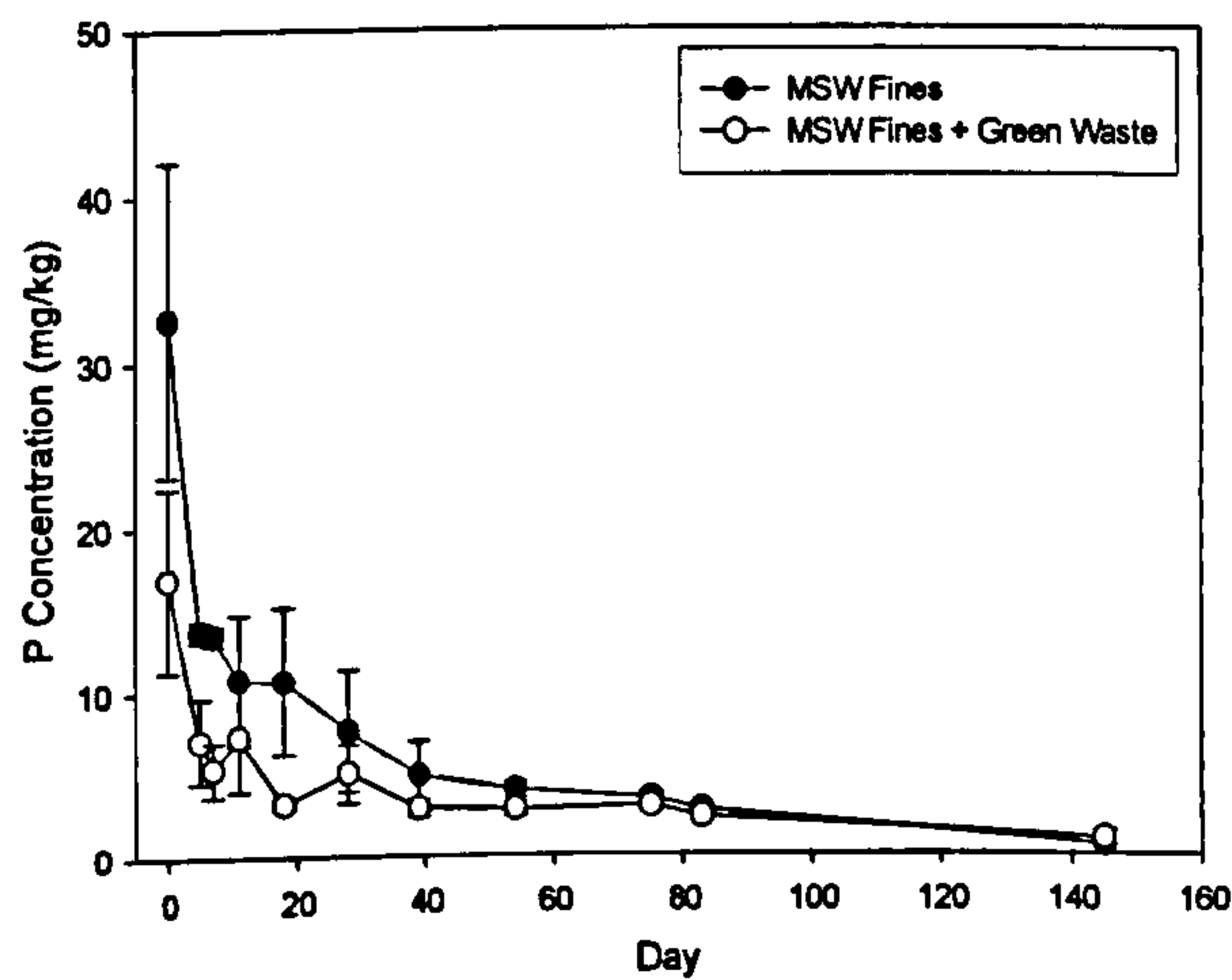


Figure 2.31: Changes in the water extractable P content of two organic wastes during composting and maturation over a 5 month period. Values represent means \pm SEM.

2.5.11. Compost C/N ratio

Compost C/N ratio progressively declined in both composts during the active composting and maturation period. There were no significant differences between the two compost types ($p < 0.05$). The first sample of both compost types had a significantly higher C/N ratio than the final sample.

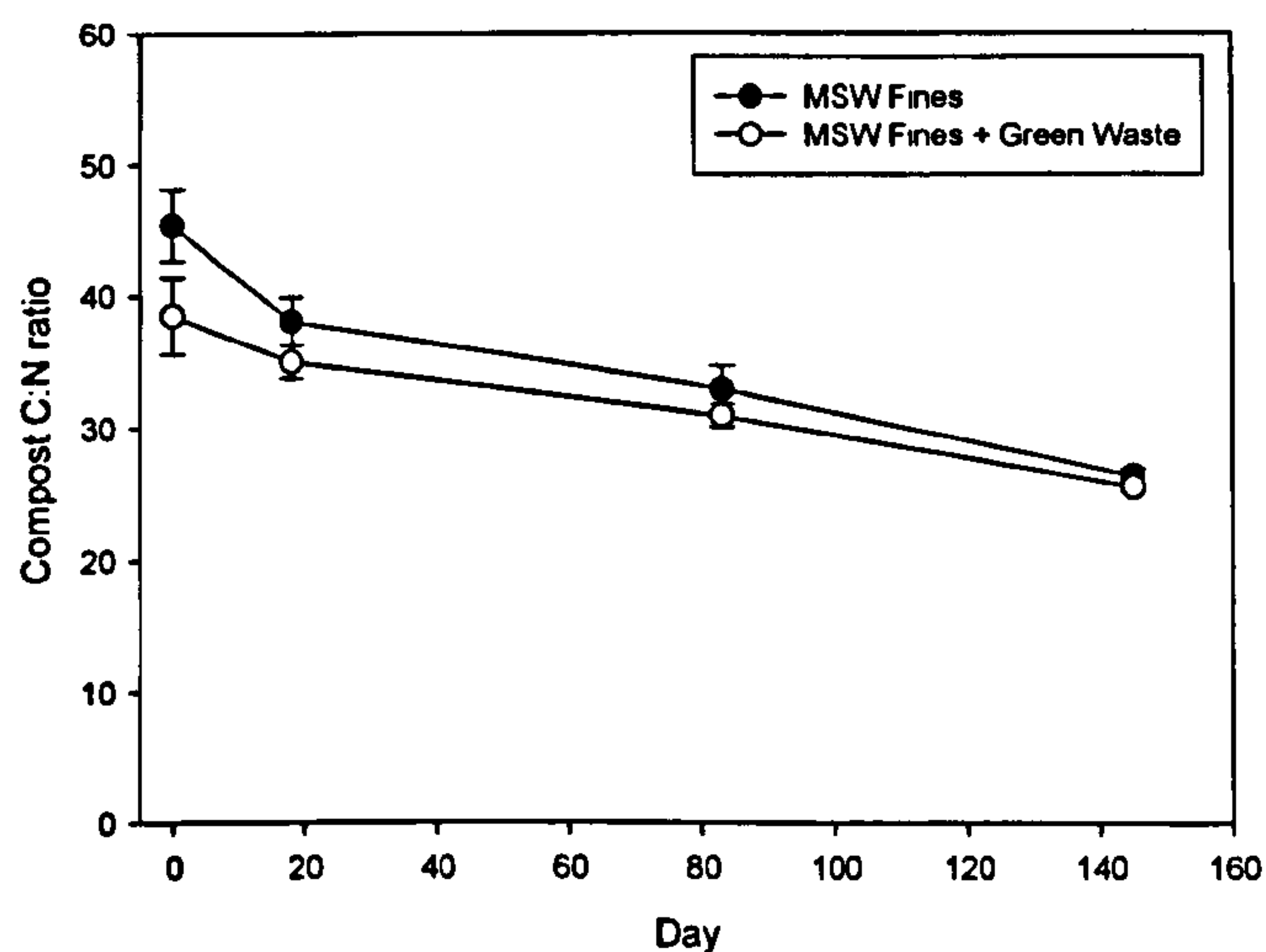


Figure 2.32: Changes in C/N ratios of two organic wastes during composting and maturation over a 5 month period. Values represent means \pm SEM.

2.5.12. Bacterial counts

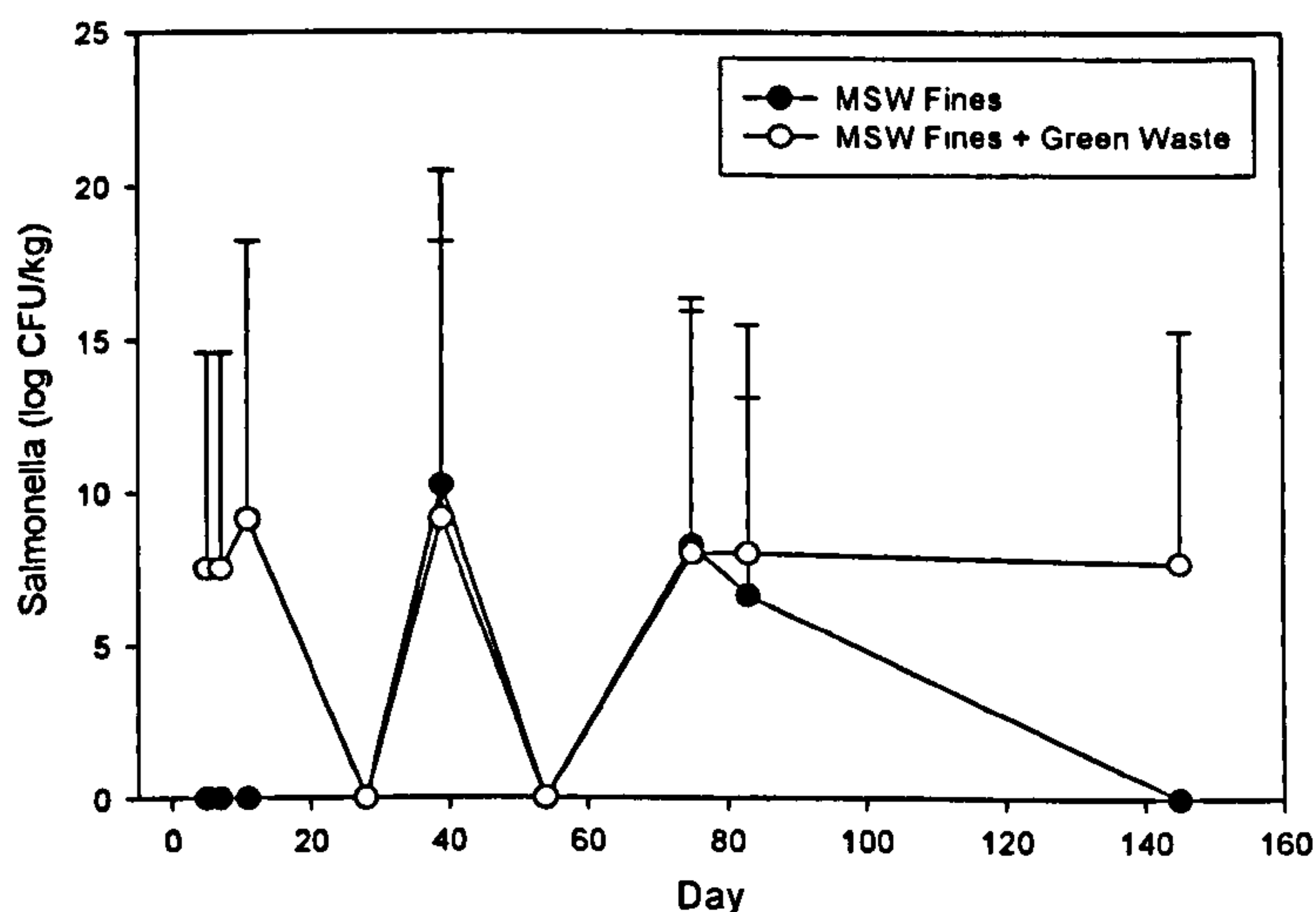


Figure 2.33: Changes in *Salmonella* counts of two organic wastes during composting and maturation over a 5 month period. Values represent means \pm SEM.

Salmonella and faecal coliforms were found to be present in the majority of compost samples, however, there was no distinct temporal pattern. MSW Fines were not found to contain any *Salmonella* during the early stages of composting, although by the sixth week, colonies were present. It also appeared that re-infection of both materials occurred between week 8 and week 11 after the former had

shown as being negative. It should be noted, however, that *Salmonella* can exist in a viable-but-non cultureable (VBNC) state (i.e. not detectable by the plating techniques employed here; Yeager and Ward, 1981). As there were large variations between samples both spatially and temporally, there were few significant differences either within or between treatments.

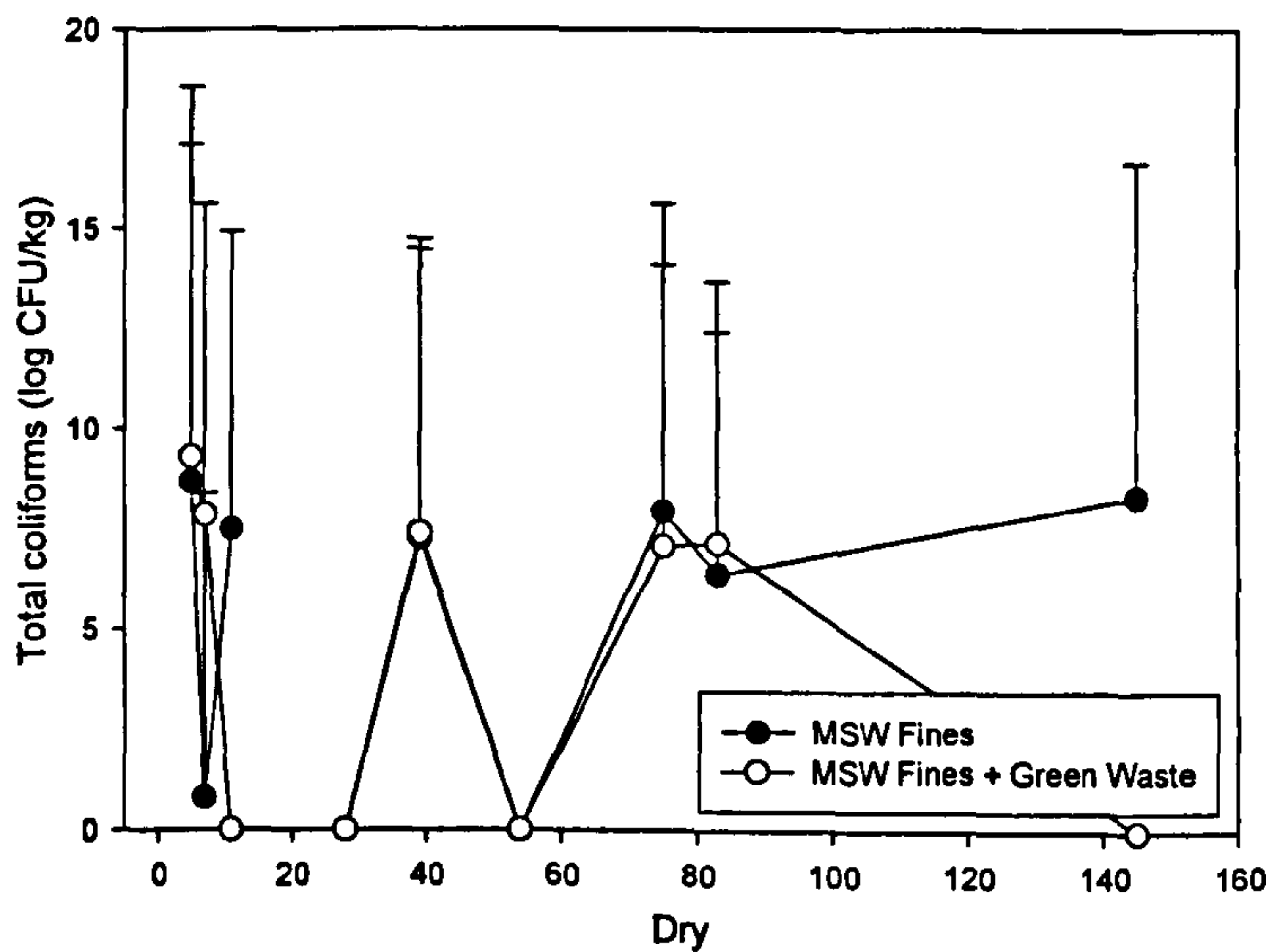


Figure 2.34: Changes in faecal coliform counts of two organic wastes during composting and maturation over a 5 month period. Values represent means \pm SEM.

2.5.13. Compost acetic acid extractable metals

2.5.13.1. Calcium

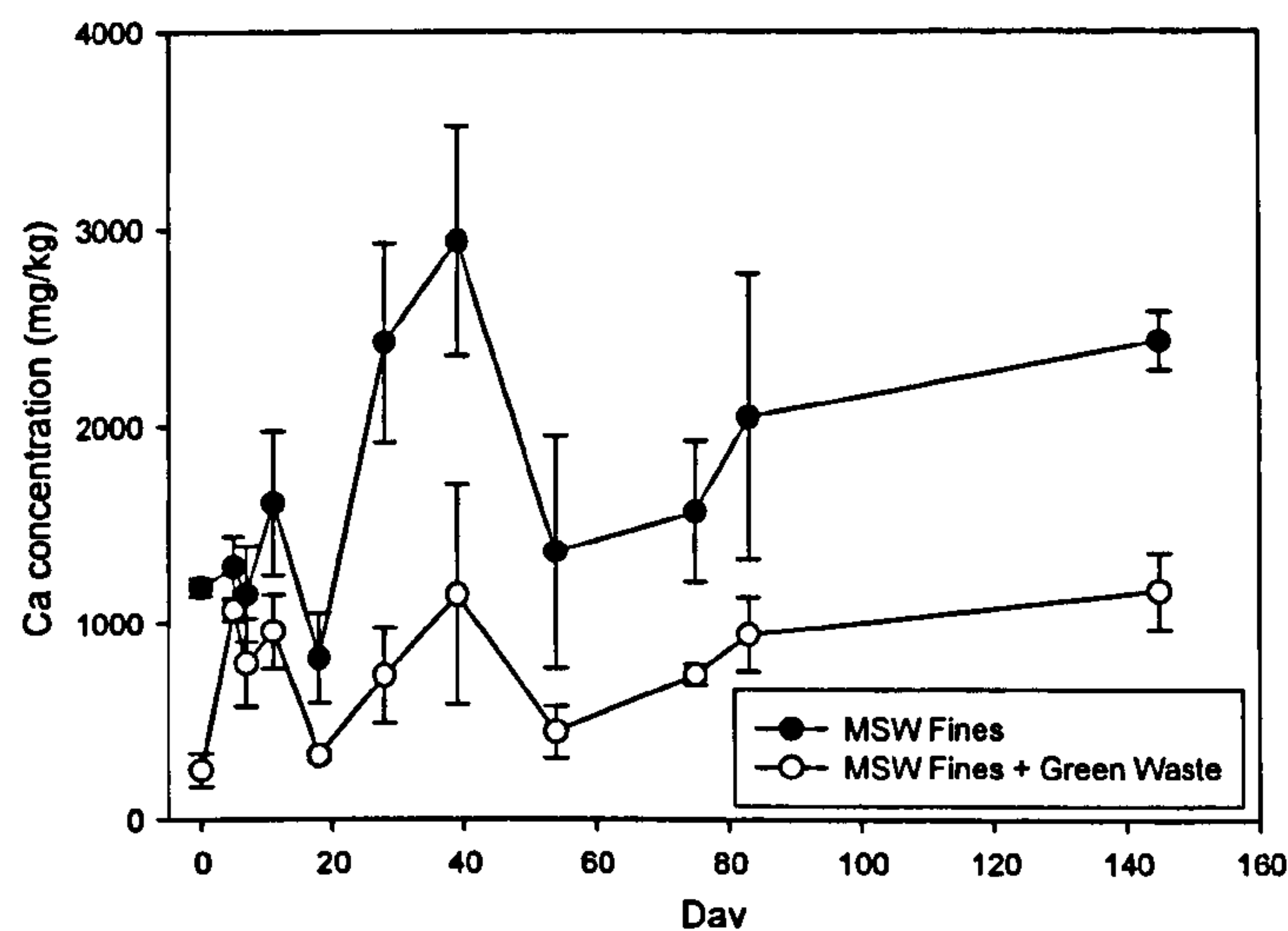


Figure 2.35: Changes in the extractable Ca content of two organic wastes during composting and subsequent maturation over a 5 month period. Values represent means \pm SEM.

Overall, there was an increase in extractable Ca in both compost types, with the Ca content in the MSW Fines consistently greater than the MSW+GW compost. No significant differences were found between any of the temporal samples in the MSW Fines. For the MSW+GW compost, the only difference was in the final sample (21 wk) and the first sample (0 wk). The MSW Fines had significantly more Ca present than the MSW+GW compost ($p < 0.005$).

2.5.13.2. Potassium

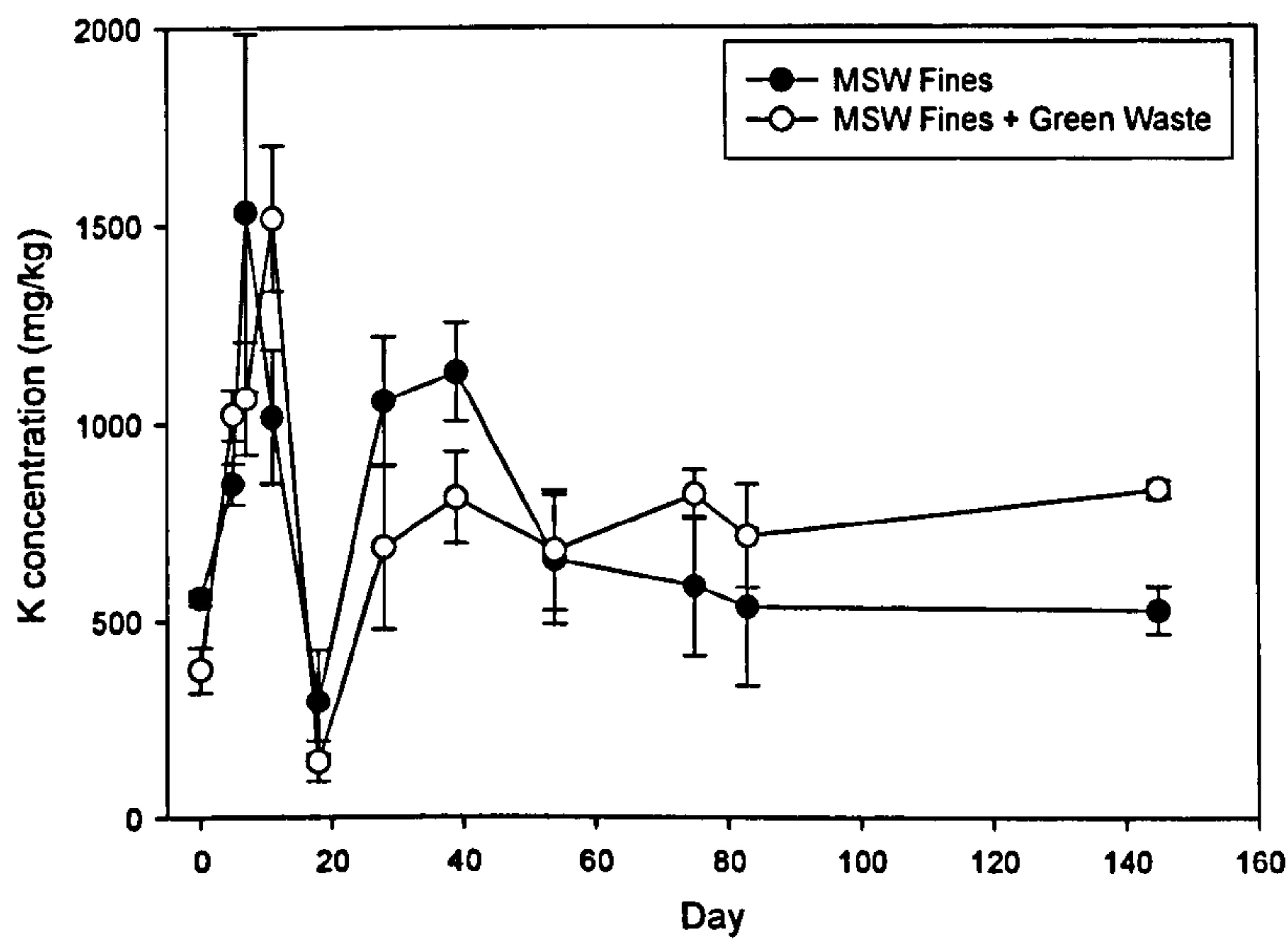


Figure 2.36: Changes in the extractable K content of two organic wastes during composting and subsequent maturation over a 5 month period. Values represent means \pm SEM.

Initially, extractable K showed a pulse in concentration (0-7 d), however, this progressively fell and concentrations became relatively stable during maturation. There were no significant differences between the MSW Fines and the MSW+GW compost ($p > 0.05$). In addition, there were no significant differences found between any of the individual MSW Fines samples. Differences were identified between some of the MSW+GW samples, with those taken around 3-4 wk of composting having significantly higher extractable K levels than the sample taken at the start of the process.

2.5.13.3. Sodium

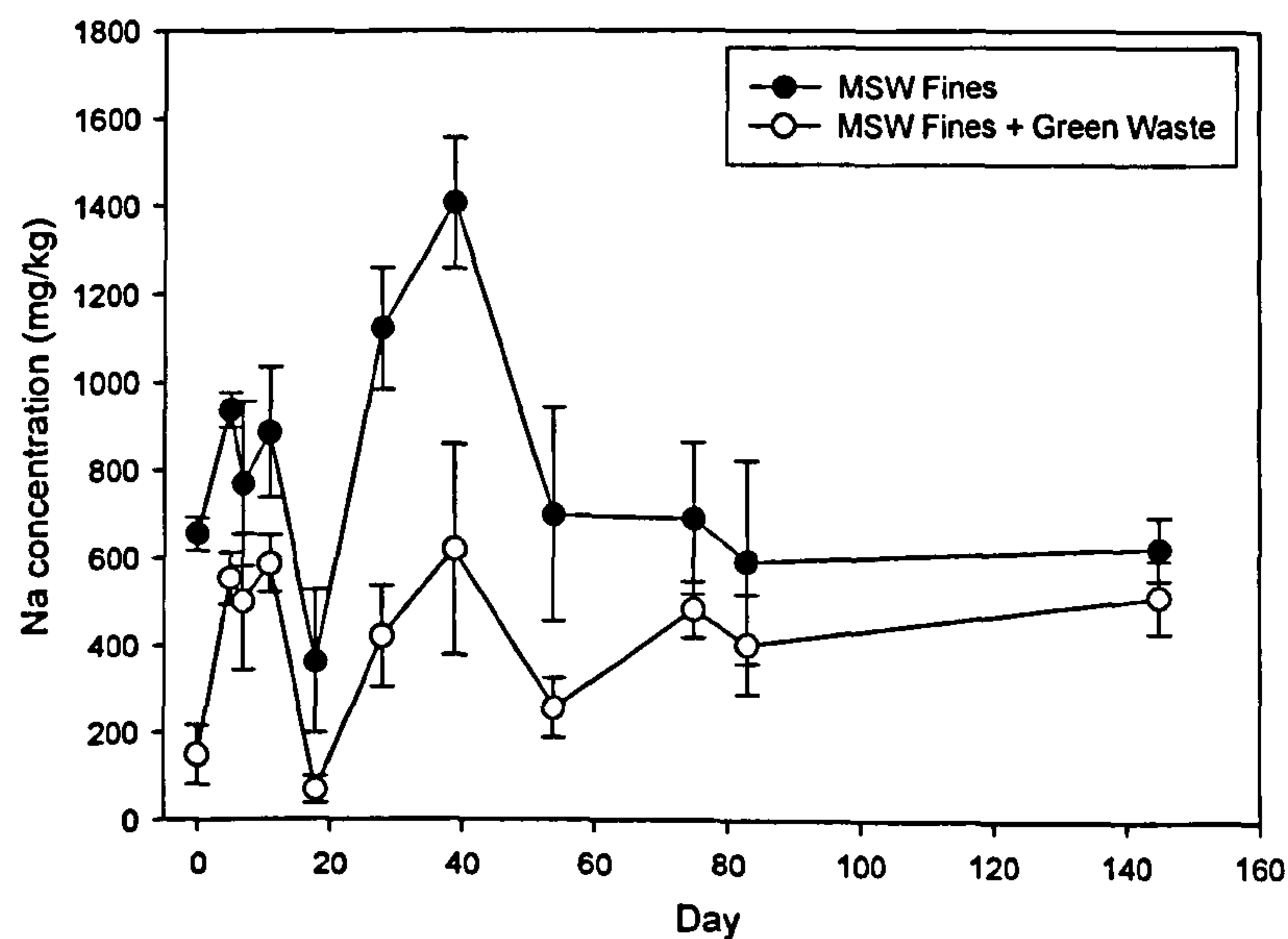


Figure 2.37: Changes in the extractable Na content of two organic wastes during composting and subsequent maturation over a 5 month period. Values represent means \pm SEM.

Extractable Na exhibited similar changes to Ca except that Na did not undergo a large increase during maturation. Yet again, MSW Fines were found to contain consistently higher concentrations of Na than the MSW+GW compost. No significant differences were found between any of the individual compost samples within a treatment. As for the Ca, the MSW Fines had significantly more Na than the MSW+GW compost ($p < 0.005$).

2.5.14. Gaseous emissions from the EcoPOD

2.5.14.1. Carbon dioxide emissions

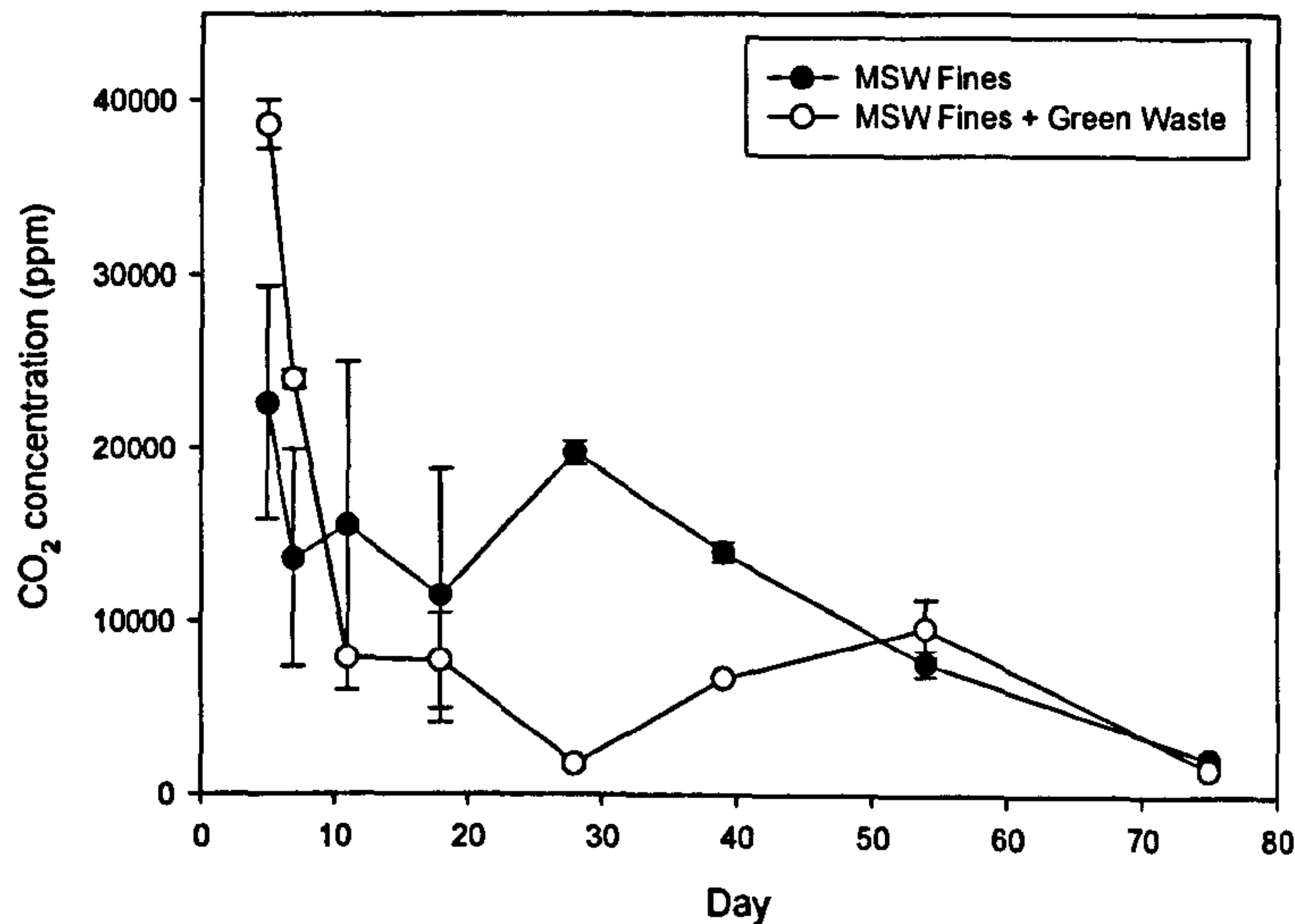


Figure 2.38: Changes in CO₂ emissions of two organic wastes during the in-vessel composting.

Values represent means \pm SEM.

CO₂ emissions from both materials progressively fell during the composting period, with the MSW Fines generally emitting higher concentrations of CO₂, apart from the initial 10 d where the MSW+GW compost was higher. No differences were found between the individual MSW Fines samples, whereas the data from the MSW+GW did show some significant differences. The major differences were that the emissions from the final monitoring date contained significantly lower CO₂ levels than the majority of the other samples; the first two monitoring points were also significantly higher than the other samples, and in turn were different to one another. There were no statistical differences between the emissions from the two materials ($p > 0.05$).

2.5.14.2. Methane emissions

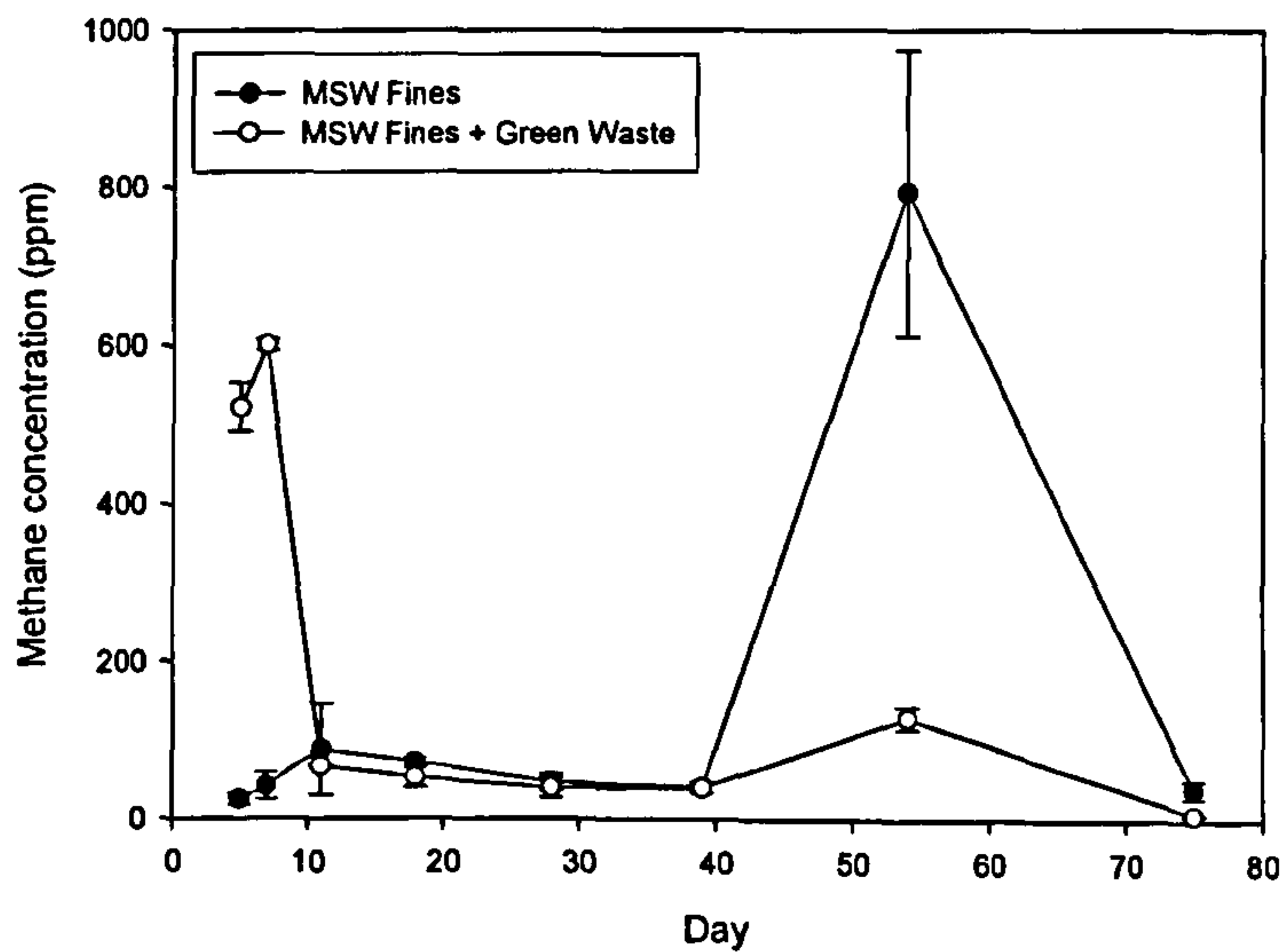


Figure 2.39: Changes in CH₄ emissions of two organic wastes during in-vessel composting. Values represent means \pm SEM.

CH₄ is produced by microorganisms which thrive in low oxygen, or oxygen free, environments, with moisture content being the major contributing factor which affects rates of production (Majumdar *et al.*, 2006). It has been recorded that anaerobic microsites can exist within waste material, which may explain the changes shown in Figure 2.39 (He *et al.*, 2000) although at times where moisture rose, the levels of CH₄ being emitted were not found to increase. Alternatively, larger anaerobic regions may have developed within the composting material (Hellebrand, 1998) rather than being a result of locally increased moisture. Compaction and consolidation of the material within the EcoPOD® is likely to have contributed to a reduction in aeration (by reducing the air spaces between particles) reducing CO₂ emissions and increasing CH₄ emissions, as seen around day 54 (Hobson *et al.*, 2005).

Statistical analysis showed that there were no significant differences between the two materials ($p > 0.05$). The penultimate sample analysed here (8 wk composting) was found to contain significantly higher CH₄ levels in its emissions than the other MSW Fines samples. As for the CO₂ data, the first two mixture samples had significantly greater CH₄ being emitted than all of the other samples;

the penultimate monitoring point was also shown to be significantly higher than the majority of the other samples.

2.5.14.3. Nitrous oxide emissions

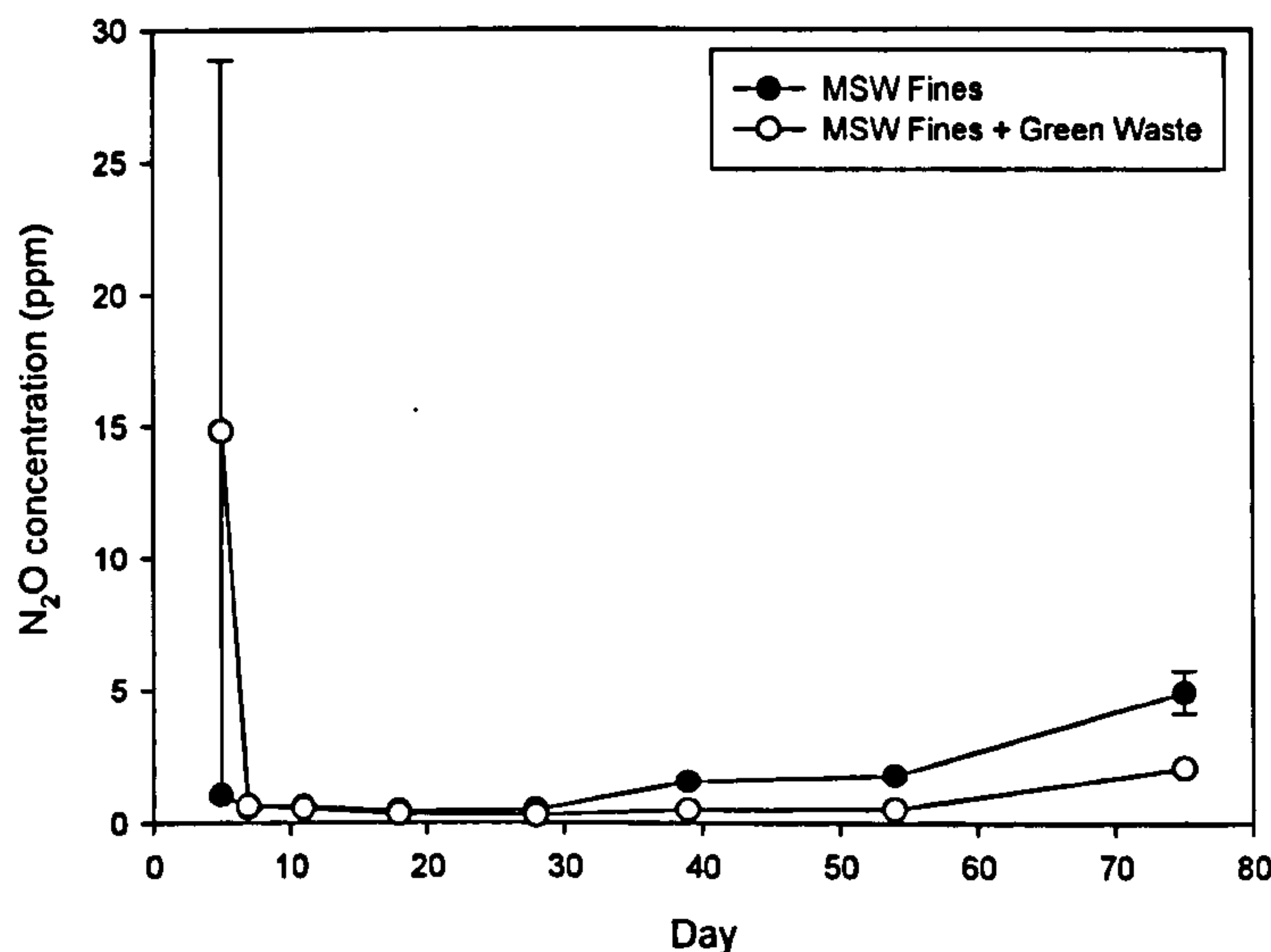


Figure 2.40: Changes in N_2O emissions of two organic wastes during in-vessel composting 3 month period. Values represent means \pm SEM.

N_2O emissions from composting can be attributed to volatilization occurring during denitrification (Hellebrand, 1998). The onset of anaerobic conditions within composting material can favour denitrification over nitrification; this can lead to NO_3^- being used by microbes as the terminal electron acceptor for respiration in the absence of O_2 resulting in it being reduced to N_2O (Hobson *et al.*, 2005). Hellmann *et al.* (1997) and Peigné and Girardin (2004) both state that N_2O can also be produced as a by-product of nitrification (an aerobic process); this depends on many other conditions being favourable. Hellmann *et al.* (1997) also state that N_2O production is generally higher at cooler temperatures than in the thermophilic phase of composting, something which can be seen here; emissions are higher at the start and end of composting, when vessel temperatures were lower.

The final sample of MSW Fines emitted significantly greater levels of N_2O than all of the previous samples. No such differences could be found in the

MSW+GW emissions. No differences were evident between the levels of N₂O emitted by the two materials ($p > 0.05$).

2.6. Discussion

2.6.1. Waste feedstock characterisation

The composition and amount of MSW produced at a regional scale is known to depend on a multitude of factors including: social factors (e.g. age, sex, income, educational level, size and status of family), residential structure patterns (e.g. one family houses vs. multi storey houses), and waste management and organisational patterns [e.g. whether household or commercial waste, bin size, collection system (door-to-door-system, drop-off-system, separate collection), waste fees; seasonal variations etc] (Plochl *et al.*, 2003; Shaw, 2008). Consequently, it is important to undertake a waste inventory at a regional level rather than relying on national statistics (Woollam *et al.*, 2006). Overall, the temporal results presented here indicate that both of the waste feedstocks had a fairly uniform composition despite their highly heterogeneous visual appearance even within an individual sample. This is a positive finding as it suggests that the composting results obtained here are not just specific to an individual batch of waste but are probably widely applicable to the MSW waste stream within a defined geographical region. From the data collected, it is obvious that there was much more spatial variability within the MSW Fines than the Green Waste (shown by the larger error bars). Although some small degree of temporal variability is expected in the MSW waste stream (e.g. Christmas versus summer; Zeng *et al.*, 2005), it is hypothesized that this will only result in small changes in the MSW Fines. For example, although the amount of paper disposed of at Christmas may increase the C/N ratio of the waste stream, this is likely to be balanced by a concomitant increase in food waste. Our results alongside those of Yousuf and Rahman (2007) support this view. Further, characterisation of the feedstocks revealed no major issues with respect to the potential viability of composting as a waste treatment method for the MSW fines (e.g. extreme chemical, biological or physical properties) and that sufficient potentially biodegradable material existed. Although waste odours are notoriously

difficult to quantify (Capelli *et al.*, 2008), the MSW fines also possessed no malodour in comparison to the raw material suggesting that pre-treatment and mixing in the DANO® drum did not induce any negative impacts. One significant issue that merits attention is that the composition of MSW is ever-changing as local authorities step up efforts to increase source-segregation of waste and recycling rates. In addition, markets for recyclables are also highly dynamic (Sharp and Luckin, 2006). In the short term this may increase the proportion of food waste in the MSW stream, however, this may decline as kerbside food waste collection systems are increasingly rolled out across the UK. In addition, waste composition may be affected if individual householders are levied on the weight of waste produced (e.g. increased removal of heavier items such as glass) (Shaw and Maynard, 2008). Significant changes in MSW composition in the long term (decade timescale) have also been reported elsewhere in Europe (Gidarakos *et al.*, 2006).

Table 2.1 below shows the composition of MSW in several studies conducted over the past two decades in several different nations:

| Study | Material as a percentage of the total waste sample | | | | | | |
|---|--|----------------------------|----------------------------|--------------------------|--------------------------|--------------------------|----------------------------|
| | Organics/Biodegradable | Paper | Plastic | Glass | Metal | Textiles | Other |
| Murphy and McKeogh (2004) | 36.0 (1995) 32.9 (1998) | 21.0 (1995) 19.5 (1998) | 10.0 (1995) 11.9 (1998) | 5.0 (1995) 5.5 (1998) | 3.0 (1995) 3.5 (1998) | 3.0 (1995) 2.9 (1998) | 22.0 (1995) 23.8 (1998) |
| Abu Qdais <i>et al.</i> (1997) | 49.0 | 6.0 | 12.0 | 9.0 | 8.0 | N/A | 16.0 |
| Bolaane and Ali (2004) | 67.9 | 12.5 | 4.5 | 6.4 | 6.5 | 1.3 | 1.2 |
| Ali Khan and Burney (1989) | 20-50 | 15-50 | 2-10 | 4-12 | 3-13 | 2-10 | 1-20 |
| Gidarakos <i>et al.</i> (2006) | 39.15 | 19.94 | 16.85 | 5.33 | 4.95 | 5.24 | 8.54 |
| Koufodimos and Samaras (2002) | 41.0 | 23.0 | 13.0 | 3.0 | 4.0 | 6.0 | 10.0 |
| Mee <i>et al.</i> (2004) | 38.0 | 25.0 | 10.0 | 8.0 | 4.0 | 3.0 | 12.0 |
| Grodzińska-Jurczak <i>et al.</i> (2003) | 53.0 | 19.0 | 4.0 | 8.0 | 3.0 | N/A | 13.0 |
| EcoPOD® Study | 77.6 (inc. paper) | - | 5.1 | 14.8 | 2.5 | N/A | N/A |
| Maximum/Minimum | 77.55/20.0 | 50.0/6.0 | 16.85/2.0 | 14.81/3.0 | 13.0/2.54 | 10.0/1.3 | 23.8/1.0 |

Table 2.1: Comparison of the composition of Municipal Solid Waste in several studies.

As stated, there are many different factors that can influence the composition of MSW. For example, Murphy and McKeogh (2004) compared composition of MSW across several years and noted that increased participation in recycling schemes helped reduce the amount of paper present.

Gidarakos *et al.* (2006) found that there was a large degree of variation within certain fractions of MSW, with the glass fraction reaching a maximum of 18% in some regions during autumnal months (these regions were mainly tourist areas). Plastics were found to make up a higher proportion of the total MSW sample in the autumn than the winter (around 10% greater). In general, this study cited the increase in tourist activity during the summer months increases the proportions of plastic, paper and glass. The study also noted that differences in the disposal habits and materials used (e.g. reusable bottles vs. non reusable bottles) also had a significant effect upon the relative proportions of the MSW fractions. A strong correlation between social activity and waste composition was also recorded.

Koufodimos and Samaras (2002) also noted that factors such as the type of industry present in a region influenced the waste composition, especially the amounts of plastic and paper present; areas with high levels of tourist activity also showed larger glass and plastic fractions. However, the changes in MSW composition are not only related to factors such as increased recycling; Grodzińska-Jurczak *et al.* (2003) noted that there had been increasing levels of plastic in Polish MSW during the 1990s as the country became more “Westernised” following the end of the Cold War.

It should be noted that the composition of MSW is highly variable from place to place, even between two neighbouring households; For example, Daskalopoulos *et al.* (1998) stated that there are a number of complex interrelationships that combine to affect the composition of municipal solid waste in many nations, further highlighting the potential for problems with disposal options.

In summary, the amount and composition of waste should be viewed as an ever-changing commodity in Wales. In addition, it is also fairly certain that the amount of waste entering the non-segregated MSW stream will continue to decline

over the next 20 years, particularly if Wales is going to meet its national and European waste diversion targets and its sustainable development goals. However, it is also likely that 'zero waste' is also unachievable based on current consumer attitudes and commercial marketing practices. Therefore treatment of MSW will always be required. The results presented here suggest that the chemical and physical properties of the MSW and green waste are amenable to composting.

Although the pH of both waste feedstocks were within the normal range for MSW and green waste (pH between 5 and 7; Schulze, 1962), some temporal variability was seen. These changes in pH probably relate to the length of storage time and the storage conditions before the samples were collected (e.g. residence time in the DANO® drum). If the samples were stored in a static pile (e.g. the collecting vessel from the DANO® trommel screen) then the production of volatile fatty acids may have reduced the pH. In contrast if the samples were continually agitated in the DANO® drum then the aerobic conditions may promote the rapid production of NH_4^+ from organic N which increases the pH (Bernal and Kirchmann, 1992). Although mineralization of the organic wastes increases immediately after collection (when a critical mass causes the temperature to rise) it is unlikely that decreases in pH are due to nitrification based upon the results NO_3^- concentrations presented here. However, the lower pH will help to reduce the amount of N lost through NH_3 volatilization which is promoted in high pH environments (pH > 8; Golueke, 1981).

The EC values were consistently higher in the MSW Fines compared to the Green Waste; a likely result of the amount of ionic substances present within the parent MSW (Van Assche and Uyttenbroeck, 1982; Villar *et al.*, 1993). While this is not of significance for the composting process, it may be of significance for the final compost quality where the high salinity may prevent end-uses involving non-halotolerant plants (e.g. horticulture). At present, there are no practical ways of reducing the salt content of MSW.

The optimum C/N ratio for feedstock to be composted is around 15-25 according to De Bertoldi *et al.* (1983). Outside of this range chemical amendments may be required to facilitate breakdown (i.e. addition of labile C such as straw or

waste paper or the addition of inorganic NH_4NO_3). This optimal C/N value was only achieved by both materials at the end of the maturation period suggesting that the conditions may have been sub-optimal for composting MSW fines alone. However, it must also be noted that some of the C contained in the MSW fines is non-labile (e.g. plastics) and that the C/N ratio of the truly compostable fraction may be lower than reported here. The C/N ratio of the MSW Fines was observed to increase towards the winter months; something that could necessitate the addition of extra N prior to composting if they were found to increase to critical levels above 50 (Schulze, 1962). This increase is possibly due to a reduction in readily degradable plant material within samples of the MSW Fines (e.g. less labile garden and food waste) and a relative increase in its paper content. Although the C/N ratio of green waste was expected to rise towards winter due to the preferential collection of more woody green waste (Rothwell and Hortenstine, 1969), this was not apparent from our results.

2.6.2. In-vessel composting

Composting is subject to much legislation at both the national and European level which is constraining its rapid expansion as a waste management treatment (i.e. due to restrictions in planning and public acceptance, waste quality and capital costs required for in-vessel infrastructure). This legislation is mainly related to the need to demonstrate destruction of human and livestock pathogens by reaching temperatures sufficient for their elimination. However, even in some commercial European MSW composting facilities this has not proved sufficient and concerns have been raised about the quality of MSW-derived composts (Lasaridi *et al.*, 2006).

In this study the composting of MSW in the presence and absence of green waste was undertaken. From the results presented here, it is clear that the addition of green waste aided the MSW composting process (as evidenced by the extended and more controlled single thermophilic phase). Emissions of CO_2 from the composting vessel decreased with time indicating a progressive decline in available substrate levels. This is supported by Rothwell and Hortenstine (1969) who found a

linear relationship between the levels of decomposable substrate present and the amount of CO₂ evolved. From the results obtained here we conclude that the MSW+GW material degraded faster than the MSW Fines alone. Utilising green waste (or similar materials) as bulking agents, even in relatively low proportions (e.g. 10% v/v), allows temperatures within composting vessels to reach higher levels compared to materials composted without bulking agents. This is caused by either the extra energy supplied by the added materials or by maintaining/increasing porosity (Manios, 2004; Dignac *et al.*, 2005; Manios *et al.*, 2007). Green wastes have also been recorded as having an “insulating” effect within a composting vessel, allowing greater heat retention (Manios, 2004). Wood chips, or similar, tend to have higher levels of lignins than fresher, greener, material, causing them to degrade slower and this may be problematic if the compost is destined for the marketplace. In this case, size screening of the compost may be required to eliminate this woody fraction.

The aerobic decomposition of green wastes is favoured over anaerobic breakdown due to the relative ease with which lignocellulosic matter degrade (Braber, 1995). As garden/green wastes have less physical contamination when compared to municipal wastes, their use as bulking agents is also favoured; this is due to the fact that garden wastes undergo source segregation rather than mechanical separation (as does MSW) (Braber, 1995). However, green wastes are subject to some degree of temporal variation in terms of their chemical and physical composition. This can even be the case in two samples of material taken from the same collection depending on the source of the deliveries (e.g. grass cutting from municipal parks/gardens versus roadside tree prunings; Benito *et al.*, 2005; Flavel and Murphy, 2006). One other potential problem with the use of bulking agents in composting is the cost of transporting the materials to the composting facility from source (e.g. collection centres) and the carbon cost associated with this transport (Manios, 2004; Manios *et al.*, 2007). If logistically possible, the Dano® drum could be used as the in-vessel composter as this can be agitated negating the need for a physical bulking agent.

The material treated here showed an initial increase in temperature reaching temperatures of around 60°C for around 7 d before decreasing steadily with time. Several studies have shown that thermophilic microbiological action is responsible for the initial temperature increase (De Guardia *et al.*, 2002; Mbuligwe *et al.*, 2002; Norbu *et al.*, 2005) and that this should be sufficient to eliminate bacterial pathogens but probably not Transmissible Spongiform Encephalopathies (TSEs/prions; Huang *et al.*, 2006). It is recognised that temperatures between 15-40°C allow mesophilic bacteria to colonise the material being composted; 40-70°C is the range in which thermophilic bacteria and fungi begin to dominate the degradation process (Mbuligwe *et al.*, 2002). In this trial, the composting process became thermophilic after 7 d, lasting for 3 wk. Although the temporal temperature patterns were similar, the duration of the thermophilic phase was less than reported by Bernal *et al.* (1998b, 1998c).

Various temperature targets for pathogen kill have been described. Some state that a period of several hours above 65°C can suffice, whilst some legislative acts tend to go further and give actual time requirements. For example, the 2005 Animal By-Products Regulations state that the whole of the material in the vessel must be maintained over 60°C for 48 h on two separate occasions if a material is to be deemed “safe”. In this study, the composting temperature appeared to be critically dependent upon depth within the vessel with the waste in contact with the vessel wall failing to meet the UK PAS-100 compost standard. This is a major issue as it prevents the compost being used off-site and therefore gives it a low economic value akin to landfill cover material (<£2 t⁻¹ for landfill cover versus £7 t⁻¹ for loose green waste and £8 t⁻¹ for bagged green waste; RWW, 2009). However, it must also be noted that the commercial in-vessel system used here was originally designed for treating farm wastes and is relatively small in comparison to other larger commercial in-vessel composting systems (e.g. NaturTech®, VCU Europa, Ortec systems etc).

The pathogen results presented here gave two significant findings. Firstly, it is apparent that the pathogens are highly spatially heterogeneously located within the waste. Secondly, the composting process failed to eliminate them. It is likely

that this is due to the low temperatures recorded around the vessel wall and the potential for continual re-infection of the remaining compost by transfer in the forced air stream (Gerba *et al.*, 1995). This re-infection hypothesis is supported by Kuhlman (1990) who found that *Salmonella* sp were “inactivated” after 20 d of composting and that temperatures of 60°C for 20 min were sufficient to kill the majority of this genus, although coliforms tend to be more heat resistant. Several studies investigating the survival and/or regrowth of *Salmonella* within composted materials have also concluded that composts have the potential to experience *Salmonella* regrowth at any point and that maturity is not a guarantee of bio-safety (Brandon *et al.*, 1977; Burge *et al.*, 1987; Sidhu *et al.*, 2001; Pietronave *et al.*, 2004). Russ and Yanko (1981) and Yeager and Ward (1981) have concluded that a moisture content of less than 20% is a critical threshold for *Salmonella* survival, below this the bacteria can become desiccated and exist in a dormant state even for periods longer than 12 months where they are highly resistant to heat treatment. In this study there were dry spots apparent within the compost where this may have occurred. Some studies have identified indigenous microflora within the composting matter as being a major factor in pathogen survival due to competition for resources (Burge *et al.*, 1987; Déportes *et al.*, 1998), although if conditions favour *Salmonella*, regrowth can occur in spite of competing organisms (Russ and Yanko, 1981). Christensen *et al.* (2002) discovered that *Salmonella* was absent in finished composts made using in-vessel methods and mention that re-growth/re-infestation may not occur at all. This study does, however, show that re-growth of faecal coliforms is liable to happen during the maturation/stabilization phase of composting.

Overall, compost pH increased with time in both materials, although an initial decrease occurred in the MSW Fines. While this could be caused by nitrification processes (Bernal and Kirchmann, 1992; De Guardia *et al.*, 2002) it is more likely to be due to the production of organic acids (Ten Brummerler and Koster, 1989). This is supported by He *et al.* (1995) who found that low pH values often result from the presence of low molecular weight organic acids produced from the incomplete decomposition of food wastes. Benito *et al.* (2003) noted that

pH increases occur during composting due to the degradation and mineralization of organic compounds within the materials and particularly the production of NH_3 from organic N. The results presented here suggest that the initial composting conditions may not have been optimal for the MSW Fines in isolation (i.e. a lack of O_2 delivery to parts of the vessel).

Electrical conductivity is commonly used as a rapid measure of soluble salts present within the compost. Both materials composted in this project showed a decrease in EC with composting time. This response could be attributable to (1) leaching of soluble compounds in condensate passing vertically through the vessel, (2) loss of soluble salts (e.g. NH_4^+ as NH_3), and (3) immobilization of salts due to imbalances in microbial nutrient supply and a rapid growth of the microbial biomass (Benito *et al.*, 2003).

Nitrate concentrations increased slightly in the MSW+GW compost over time; however, there was much greater variation within the MSW Fines compost. Due to the spatially heterogeneous nature of MSW Fines, the degree of variation is unsurprising, with individual replicates containing high amounts of NO_3^- influencing the final readings (shown by the error bars). An increase in NO_3^- can be related to the conversion of NH_4^+ to NO_3^- by nitrification. Typically, this is used as a good indicator of compost maturity as nitrification often only occurs when the thermophilic phase has halted. In addition, nitrification can be inhibited at high pH with optimal rates observed in the region pH 6-7 (Amlinger *et al.*, 2003; Terry *et al.*, 1981). As the changes in inorganic N concentrations constituted a relatively small part of the overall dissolved N (DN) present, the changes in DON and DN need to be analysed together and a comparison drawn. It was discovered that there was a significant decrease in the levels of organic N during composting of both materials; normally this would be accompanied by an increase in the inorganic forms (NH_4^+ and NO_3^-) as the organic N is mineralized.

The C/N ratios of both wastes showed a progressive decline during composting (from 45 to 25) which is in line with published reports on composts reaching maturity (He *et al.*, 1995). If the untreated wastes were applied directly to soil (which typically has a C/N ratio of 11-14) they would immobilize intrinsic soil

N and this would be expected to be highly detrimental to crop growth (Busby *et al.*, 2007). Composting therefore offers real benefits in removing large amounts of the labile C as CO₂ whilst retaining more of the N (albeit with some volatile loss as NH₃). The relatively low amounts of available N (i.e. NO₃⁻ and NH₄⁺) in the mature compost suggest that most of the N present is either undecomposed or more likely has become immobilised in the microbial biomass (Sikora *et al.*, 1980). This view is supported by Makaly Biel *et al.* (2000) and Terry *et al.* (1981) who showed that high C/N ratios encourage immobilisation of inorganic N during the early stages of composting while Tester *et al.* (1979) state that the rate of N mineralization is inversely related to the C/N ratio.

2.6.3. Heavy metal content

Although composts may contain many organic pollutants, their inorganic heavy metal content remains one of the key criteria for determining compost quality. Many studies have monitored the heavy metal content of different composted materials including MSW and Green Wastes. Table 2.2 provides a comparison of previous studies with the EcoPOD® study conducted here:

| Study | Material | Metal content (mg/kg) | | | | | | | |
|--|-------------------------------------|-----------------------|-------------|---------------|--------------|----------------|-------------|-----------|-----------|
| | | Cadmium (Cd) | Copper (Cu) | Chromium (Cr) | Iron (Fe) | Manganese (Mn) | Nickel (Ni) | Lead (Pb) | Zinc (Zn) |
| Wong <i>et al.</i> (1983) | Refuse Compost | - | 550 | - | - | 1179 | - | 361 | 1488 |
| | Activated Sludge | - | 329 | - | - | 148 | - | 87 | 2431 |
| Zorpas <i>et al.</i> (2000) | Dewatered Sewage Sludge | 2 | 258 | 552 | 5098 | 150 | 41 | 326 | 1739 |
| | Composted Dewatered Sewage Sludge | 2 | 205 | 578 | 4118 | 168 | 45 | 335 | 1801 |
| | Organic Fraction of MSW | - | 130 | 118 | 4201 | 106 | 12 | 113 | 263 |
| | Composted Organic Fraction of MSW | - | 128 | 120 | 4001 | 110 | 12 | 114 | 264 |
| Iglesias Jiménez and Pérez García (1991) | Urban Refuse | 1 | 148 | 23 | 6487 | 112 | 25 | 94 | 184 |
| | Urban Refuse + Sewage Sludge | 3 | 90 | 59 | 14725 | 279 | 82 | 196 | 303 |
| Lima <i>et al.</i> (2004) | Selectively collected Organic Waste | - | 25 | 23 | - | - | 5 | 5.6 | 58 |
| | Non-selected wastes | 1.9 | 65 | 51 | - | - | 9 | 53 | 151 |
| | MSW Compost | 2.05 | 270 | 42.5 | - | - | 20 | 244 | 374 |
| Keeling <i>et al.</i> (1994) | Organic Household Waste Compost | 0.27 | 30 | 21 | - | - | 16 | 28 | 130 |
| García-Gil <i>et al.</i> (2000) | MSW Compost | <0.2 | 548 | 83 | 11662 | 175 | 81 | 681 | 1325 |
| EcoPOD® Study | MSW Compost | 0.6 | 159 | - | - | - | 38 | 351 | 256 |
| | MSW Fines + Green Waste | 0.2 | 109 | - | - | - | 23 | 87 | 173 |
| RANGE | | <0.2 – 3 | 25 – 548 | 23 – 578 | 4001 - 11662 | 106 – 1179 | 5 – 82 | 6 – 681 | 58 – 2431 |
| UK PAS 100 Specification Limits | | <1.5 | <200 | <100 | - | - | <50 | <200 | <400 |

Table 2.2: Comparison of the heavy metal content of several composting studies carried out on different materials to the UK PAS 100 Specification Limits.

Table 2.2 shows that there is a wide degree of variation between different composting feedstocks in terms of their heavy metal contents. The figures obtained from the EcoPOD® study were within the ranges of values recorded in the 7 previous studies. In comparison to the UK PAS 100 Compost Quality Specification Limits, the MSW Compost from the EcoPOD® contained lower levels than the limits for all metals, with the exception of Pb (average of 351 mg/kg compared to an upper permissible limit of 200 mg/kg). However, the mixture of MSW Fines and Green Waste has a much lower Pb content and subsequently meets the PAS 100 Limits for Pb and all other metals tested. It can be concluded therefore that addition of Green Waste to the MSW Fines appears to reduce the total amount of heavy metals present (i.e. by dilution). Generally, the MSW Fines compost and the MSW Fines + Green Waste compost contained both lower amounts of heavy metals than found in the previous studies.

2.7. Conclusions

- Composition of the MSW Fines used for this experiment had higher levels of biodegradable material than other studies, but also had higher levels of glass present. This is most likely due to the fact the material had been screened prior to use.
- MSW Fines experienced a greater degree of variation between samples, and even between replicates compared to the MSW Fines/Green Waste mix.
- MSW is a highly heterogenic material that varies in terms of physical and chemical composition and that these variations have many possible causes.
- The use of an EcoPOD® to compost MSW Fines on its own and when mixed with Green Waste proved effective for composting and reducing waste volume.
- The presence of human pathogens within the composted material and their apparent regrowth during the EcoPOD® process is the biggest potential issue arising here in terms of effectiveness of the process and defining end uses for the material.

- Greenhouse gas emissions reduced during the period of composting, resulting in a lower potential for landfill gas production should the material be subsequently landfilled.
- The C/N ratio of the MSW fines feedstock was outside the range considered optimal but this was most likely due to the presence of non-labile C in the form of plastics.
- The addition of a Green Waste bulking agent to the MSW allowed a faster rate of MSW decomposition; this is probably due to increased aeration amongst other factors.
- Vessel temperature was dependent upon depth within the composting material, with that nearer the surface often cooler than the core of the vessel.
- Green Waste appeared to reduce the heavy metal content of the composting material to levels below the limits of the PAS100 specifications.

2.8. Further Work

It is possible that composting of MSW Fines produced using other sorting techniques, taken from a different geographical location or even composted using another in-vessel method may produce different results in terms of chemical composition changes. As previously mentioned, knowing the physical composition of a sample of MSW could also help identify the causes behind trends in certain results such as C/N ratio. As such, composting MSW Fines taken from an alternative source could, potentially, deliver significantly different results. In particular, continual agitation of the material (as occurs in a Vertical Composting Unit (VCU) in vessel composter) would allow better aeration and moisture control of the MSW fines. Further work is also required to establish whether pathogens can realistically be eliminated during the composting process. In terms of the carbon economy of the process it would also be good to undertake a full life cycle assessment of the MSW composting process in comparison to direct landfill. This should also incorporate the Life Cycle Analysis (LCA) costs associated with the addition of bulking materials. With respect to potential contamination, it is also

clear that the analysis should also include persistent organic pollutants (POPs; e.g. PAHs, PCBs, dioxins) within the feedstock and final compost. It must be noted, however, that limits for POPs in compost do not exist. In addition, it would also be useful to measure bioaerosols during compost production to assess the human exposure risk.

2.9. References

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Evaluation of a MSW-based feedstock from an MRF for use in co-composting with Green Waste.

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Evaluation of a MSW-based feedstock from an MRF for use in co-composting with Green Waste.

3.1. Abstract

The majority of household waste produced in the UK is disposed of to landfill, but as there is an increasing move towards treating waste prior to landfilling in an attempt to recover and reuse materials contained within the household waste. Materials Recovery Facilities (MRF) are installations which utilise various methods of separation to remove constituents of waste for recycling (e.g. metals, plastic etc.) or for composting (biodegradable matter – kitchen or garden waste). This study utilised an MRF based in Mid Wales to remove the biodegradable fraction from local Municipal Solid Waste collections for use as feedstock for in-vessel composting in combination with Green Waste. The feedstock materials were characterised prior to composting. The aims of this study were firstly to determine the composition of MSW Fines and Green Waste prior to composting. Secondly, to evaluate changes in chemical composition of two mixtures of MSW Fines and Green Waste (50/50 and 70/30 – volume/volume) when composted using in-vessel methods. Thirdly, to evaluate the suitability of the in-vessel composting technology employed here for use with the feedstocks used. The study showed that temperatures could be reached using this technology that are sufficient to meet the requirements of legislation such as the ABPR (2005), with the 70/30 mixture producing favourable results with respect to sanitization potential. Using several standards of maturity indication (e.g. C/N ratio), the two mixtures tested here could not be labelled as being completely mature and stable, hence the potential end uses for such material are severely limited. The results obtained from this study highlight that there appears to be a trade off between the rapid throughput of feedstock materials against the quality of the compost produced.

3.2. Introduction

Composting of MSW is a commonplace waste treatment practice in many countries worldwide (Elango *et al.*, 2009; Lu *et al.*, 2009). The compost produced at the end of the process varies widely in both composition and economic value from nation to nation and even between neighbouring municipalities (Abu Qdais *et al.*, 1997). This variation is caused by the intrinsic differences in the composition of the parent MSW which in turn is regulated by many factors including legislation, waste infrastructure, monetary issues, and the disposal habits of households which is influenced by their financial status, attitude to recycling/the environment and consumption habits (preferences for certain foods etc) (Abu Qdais *et al.*, 1997; Daskalopoulos *et al.*, 1998; Koufodimos and Samaras, 2002; Grodzińska-Jurczak *et al.*, 2003). At present, MSW derived compost generally has a low economic value due to its perceived 'dirty' nature and the often poor quality of the compost. However, rarely is the composting process optimized for MSW and this could be responsible for the poor quality of the final product (O'Brien *et al.*, 1995).

In-vessel composting is used to treat many types of wastes and allows a greater degree of process control of key factors such as moisture content and aeration, compared to windrow composting or other methods. It may also allow better containment of bioaerosols and odours which are known to be negative issues associated with MSW composting (Kissel *et al.*, 1992; Nikaeen *et al.*, 2009). Such process control is also important, especially when the material being composted has to meet the strict requirements of legislation such as the 2005 EU Animal By-Products Regulations.

MSW was chosen as the main feedstock for this project, more specifically, MSW Fines were chosen. MSW Fines were sourced from a Materials Recovery Facility (MRF) operated by LAS Waste Ltd at Lampeter, Ceredigion. The site takes in household waste collections from the local area and passes the material through a series of operations to remove recyclable/reusable materials (such as paper, cardboard and metals). Contained within the facility is a trommel screen which is used to extract the majority of biodegradable material contained within the household waste stream.

The trommel in this MRF consists of a large rotating drum, around 10-15 m in length with a perforated outer containing holes of 50 mm diameter, through which any items smaller than this size pass into large collecting hoppers. The material contains a large amount of biodegradable matter giving it a good potential for use as a compost feedstock yet there is often physical contamination in the way of broken glass, plastic residues or metallic items. Due to local authority kerbside collections of recyclable items, this contamination is generally lower than in other regions, however, it still represents a problem with the potential end uses of any compost produced (Brinton, 2005). A key question is whether these MSW Fines will compost better in the presence of green waste. Economically, it is desirable not to include green waste or another bulking agent (e.g. chipped wooden pallets) prior to composting the MSW Fines due to the extra capital and operational expenditure incurred and the extra time and labour required in sourcing the material. However, conversely, the added green waste may improve the quality of the final product (Chong, 2000). Therefore a critical evaluation of the use of extra agents in the composting of MSW Fines is required.

In this study an in-vessel treatment technology was used, however, in this case the system was specifically designed for achieving the UK PAS 100 compost standard. The technology also afforded greater control over certain elements of the composting process such as temperature and aeration.

The objectives of this work were as follows:

- Determine the composition of MSW Fines and Green Waste prior to composting;
- Evaluate the changes in chemical composition of two mixtures containing differing proportions of MSW Fines and Green Waste during in-vessel composting and maturation;
- Evaluate the suitability of the in-vessel composting method utilised here for the composting of MSW Fines and Green Waste.

3.3. Materials and methods

3.3.1. Site description

The commercial MRF owned by LAS Waste Ltd is located on the outskirts of Lampeter, Ceredigion on the A485 (Tregaron Road)(Fig. 3.1). It receives a combination of municipal mixed black bag waste and source-segregated recyclates. All of these pass through the MRF which is shown visually in Figure 3.2 and represented schematically in Figure 3.3. Samples of MSW Fines were obtained from the standard MRF line for the composting trials.

Figure 3.1: Location of LAS Waste Ltd MRF, Lampeter, Ceredigion.

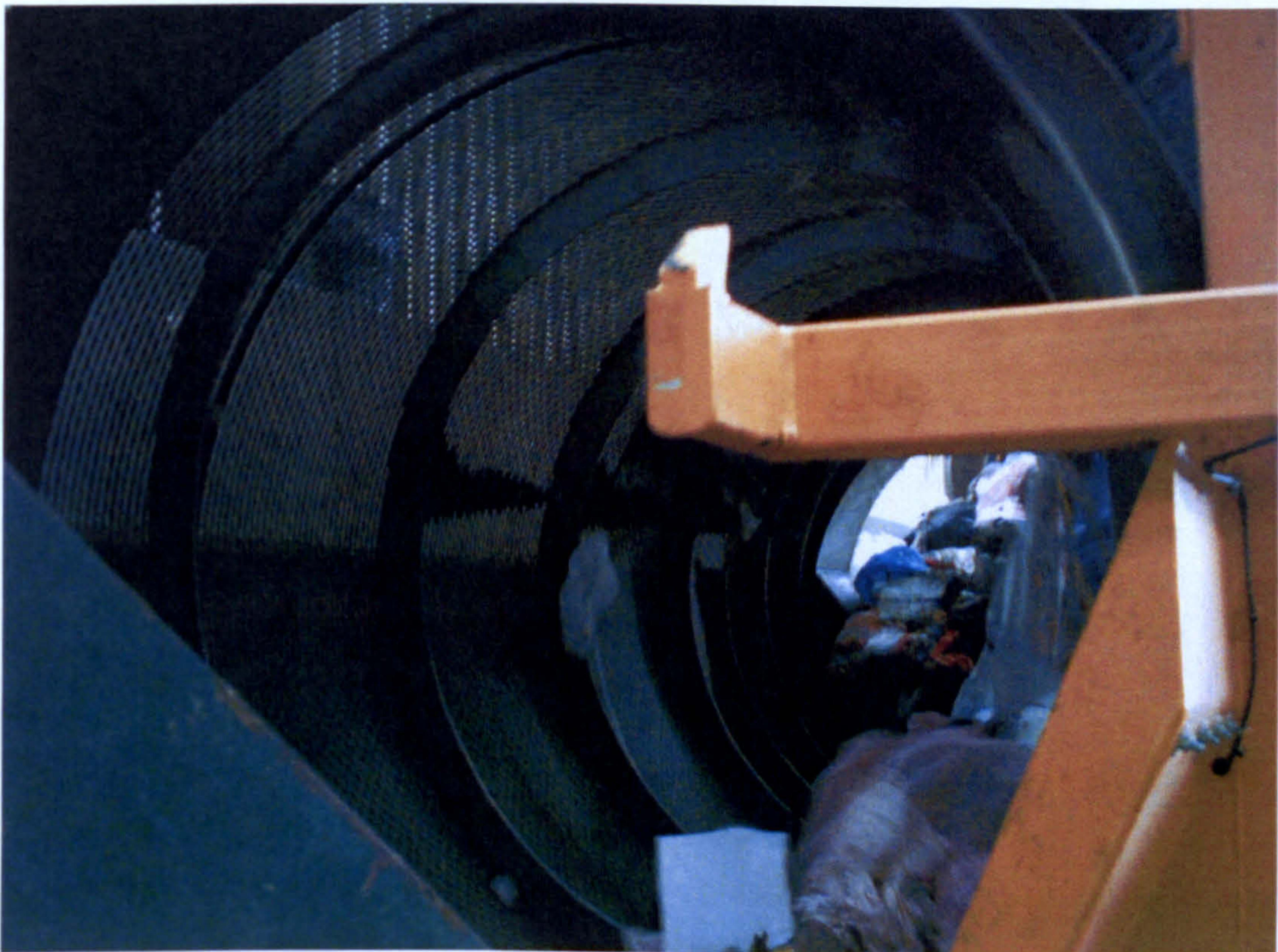


Figure 3.2: Trommel screen located in the LAS Waste Ltd MRF, Lampeter, Ceredigion.

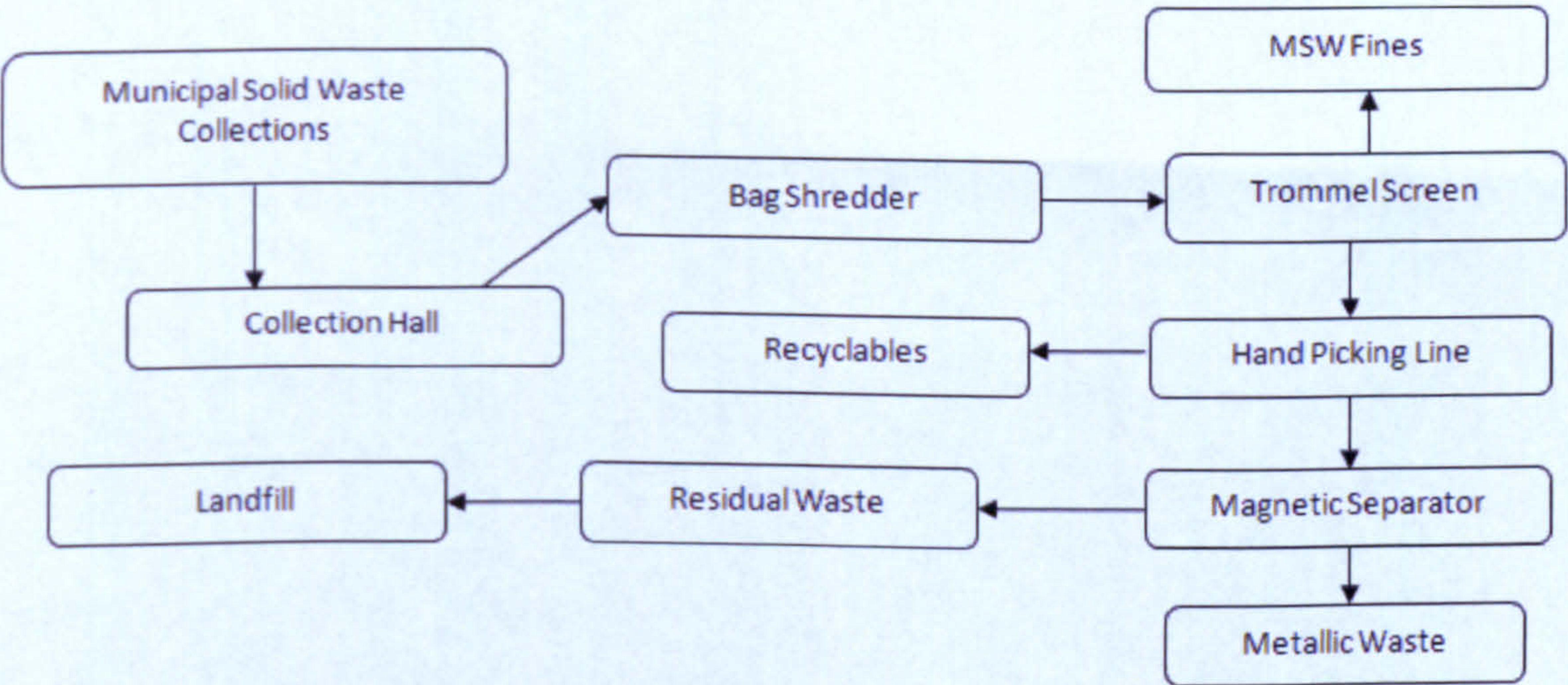


Figure 3.3: Schematic flow diagram showing the movement of waste materials at the LAS Waste Ltd MRF, Lampeter, Ceredigion.

The Vital Earth composting facility is located near to the hamlet of Sutton, near Market Drayton, Shropshire (off the A41). The site receives both Green Waste and

Kitchen Waste material collected by North Shropshire District Council fortnightly (alternating with household waste collections) from the kerbside. In this study, the technology was adapted to treat MSW Fines.

Figure 3.4: Location of the Vital Earth Composting Facility, Sutton, nr. Market Drayton, Shropshire.

The in-vessel composting system was based around converted agricultural trailers fitted with false floors, through which air is forced using a pump system attached to the back of the trailer. The exhaust pipes are connected to stationary filter beds, which help to reduce odorous emissions from the system. For ease of use, the trailers can be moved by tractor and are fitted with hydraulic tops and rears to allow filling and emptying to be easily performed. In line with the 2005 ABPR (DEFRA, 2005), composting material must reach temperatures greater than 60°C for 48 (continuous) h and this must be repeated; the hydraulic lids and backs help make the loading/emptying for this much easier. A computer control system constantly

monitors the temperature within the vessels and allows the operator to check how long the material has been over the target temperatures.



Figure 3.5: Vital Earth trailer-based in-vessel composting system at Sutton, Shropshire.

At this composting site, a colour coding system is used to identify if a vessel was on the first or second rotation of 48 h at over 60°C, coloured plates on the vessel are swapped once the first rotation has been completed.

1. 50% MSW Fines and 50% Green Waste (termed 50/50 in the following text)
2. 70% MSW Fines and 30% Green Waste (termed 70/30 in the following text)

Each of these mixtures (25 t vessel⁻¹) was produced by loading the MSW Fines and Green Waste into a tractor driven mixing unit fitted with automatic weighing scales. Material was then loaded into the composting vessels using an agricultural elevator before the vessels were moved from the loading hall and connected to the computer control system and the power and air supplies. The control system

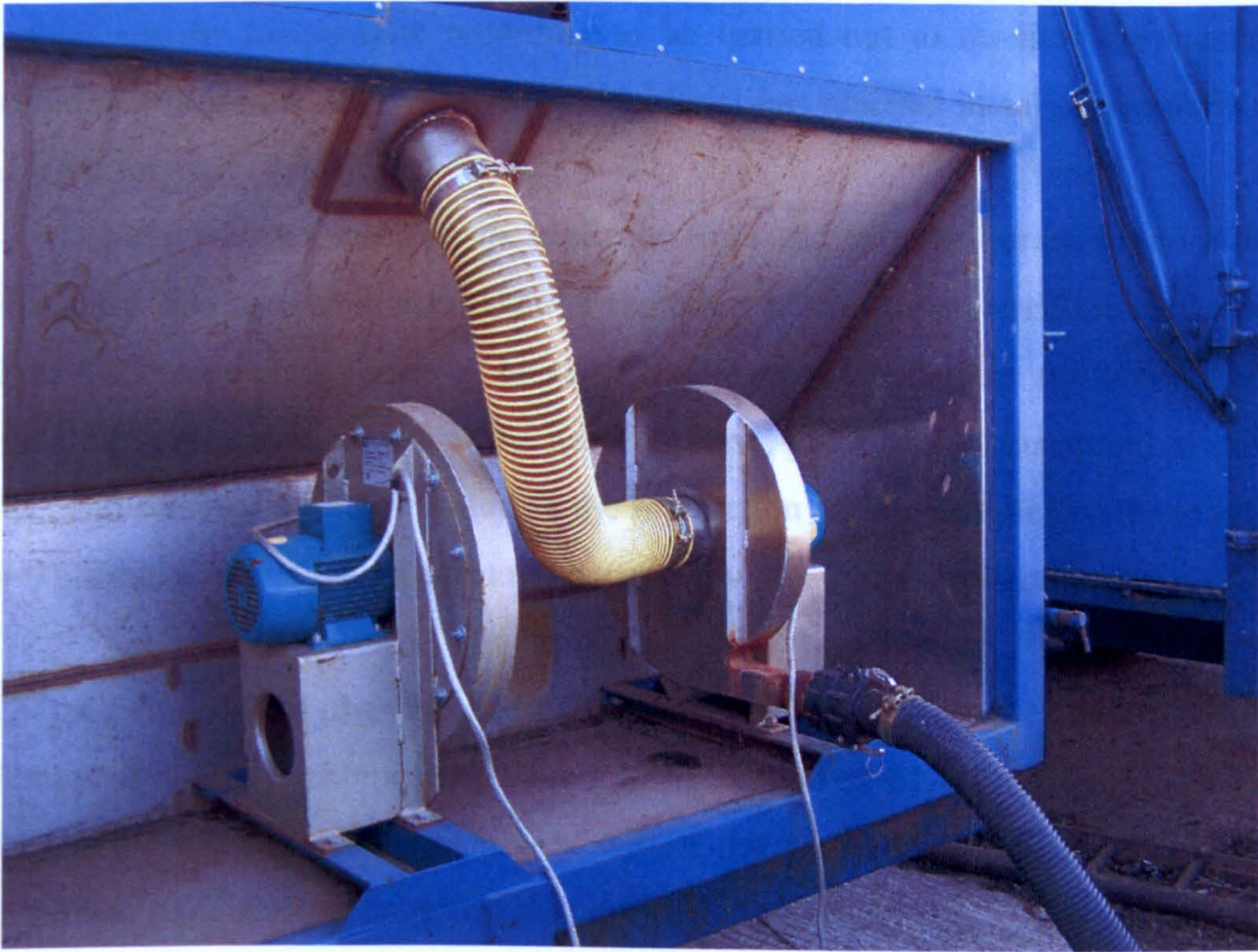


Figure 3.6: Ventilation system of the in-vessel composting system at Sutton, Shropshire.

Figure 3.6 shows the fan assembly connected to the back of the composting vessel. Air is pulled through the vessel by the fan, where other in-vessel systems force air into the composting material. The black pipe in the bottom right of the picture is linked to a stationary filter bed.

Two combinations of MSW Fines and Green Waste feedstocks were created for the purpose of these trials:

1. 50% MSW Fines and 50% Green Waste (termed 50/50 in the following text)
2. 70% MSW Fines and 30% Green Waste (termed 70/30 in the following text)

Each of these mixtures (25 t vessel^{-1}) was produced by loading the MSW Fines and Green Waste into a tractor driven mixing unit fitted with automatic weighing scales. Material was then loaded into the composting vessels using an agricultural elevator before the vessels were moved from the loading hall and connected to the computer control system and the power and air supplies. The control system

enables regular temperature recording to be carried out to monitor the changes within the vessel and identify when the material has reached its target temperature of 60°C for 48 h (twice).

Material was sampled in triplicate from the vessels during loading and unloading. Samples were also taken at the point where the material was unloaded from the composting vessel to be turned prior to reloading for the second heat cycle. Following completion of the heating phase, the material was stored in static piles and sampled at intervals during the maturation phase. Samples were stored at 4°C prior to analysis which normally occurred within 24 h of collection.

3.3.2. Chemical and physical analysis

Temperature of the composting vessel was recorded at 24 h intervals by way of thermistors located at several points within the composting vessel. pH (Hanna Instruments pH 209 pH meter) and electrical conductivity (EC) (Jenway 4010 EC meter) were determined in a 1:1 (v/v) deionized water:waste slurry, after mixing for 1 h. Moisture content was determined by drying samples at 80°C for 24 h and organic matter content was determined as loss-on-ignition at 450°C (16 h). Total C and N were determined using a CHN2000 analyser (LECO Corp., St Joseph, MI); prior to CHN analysis, samples were air dried (25°C, 14 d) before being ground in a T1-100 vibrating sample mill (Heiko Co Ltd., Fukushima, Japan) equipped with tungsten grinding vessels.

Nitrate and ammonium were extracted with 1 M KCl at a 1:5 (w/v) fresh sample:extractant ratio for 1 h on a reciprocating shaker at 250 rev min⁻¹ (Zhong and Makeshin, 2003). The supernatant was removed by centrifugation of material at 4000 rev min⁻¹ for 30 min and the collected solutions filtered through a Whatman 42 filter paper. Nitrate and ammonium were determined using a San⁺⁺ segmented flow analyser (Skalar Inc., Norcross, GA). Extractable K, Na, Ca and P were extracted using 0.5 M acetic acid using 1:5 (w/v) fresh sample:extractant ratio (Quevauviller, 1998). K, Na and Ca were analysed using a Sherwood 410 flame photometer (Sherwood Scientific, Cambridge, UK). P was obtained by extraction, firstly with sodium hydrogen carbonate (NaHCO₃) buffered to pH 8.5, and also

with deionised water using a 1:5 (w/v) fresh sample:extractant ratio for 1 h on a reciprocating shaker at 250 rev min⁻¹. The supernatant was removed by centrifugation at 4000 rev min⁻¹ for 30 min and the supernatant filtered through a Whatman 42 filter paper. Phosphate was determined colorimetrically using the method of Murphy and Riley (1962). Water extractable dissolved organic C (DOC), dissolved inorganic C (DIC) and dissolved organic N (DON) were measured using a Shimadzu TOC-V-TN analyser (Shimadzu Corp., Kyoto, Japan).

3.3.3. Statistical analysis

All data were inputted into SPSS v12.0 (SPSS Inc., Chicago, IL), and a 1-way ANOVA was performed. Tukey's Honestly Significant Difference (HSD) post hoc test was used to differentiate between the compost treatments. Means and standard errors were computed using Microsoft Excel 2003 (Microsoft Corp., Redmond, WA), and all graphical figures were produced in SigmaPlot v8.0 (Systat Software Inc., San Jose, CA). All nutrient values are expressed on a dry weight basis (mg/kg) with the exception of water extractable DOC, DIC and DON which are expressed in mg/l.

3.4. Results

3.4.1. Waste characterization results

Table 3.1 summarises the results of the chemical analyses conducted upon the MSW Fines and Green Waste feedstocks collected from the two sites prior to composting. Overall, there were some significant differences in the chemical characteristic between the two feedstocks, however, these were generally small.

| | MSW Fines | Green Waste | Significance |
|--|-----------|-------------|--------------|
| pH | 6.0 ± 0.2 | 7.1 ± 0.2 | $p < 0.01$ |
| EC (µS/cm) | 2.8 ± 0.4 | 1.4 ± 0.3 | $p < 0.05$ |
| NO ₃ ⁻ (mg N/kg) | 3.6 ± 1.1 | 3.7 ± 2.5 | <i>NS</i> |
| NH ₄ ⁺ (mg N/kg) | 41 ± 4 | 9.3 ± 3.5 | $p < 0.001$ |
| Olsen P (mg/kg) | 83 ± 16 | 76 ± 13 | <i>NS</i> |
| Extractable Ca (mg/kg) | 0.5 ± 0.1 | 0.2 ± 0.1 | $p < 0.05$ |
| Extractable K (mg/kg) | 0.3 ± 0.0 | 0.6 ± 0.0 | $p < 0.01$ |
| Extractable Na (mg/kg) | 0.4 ± 0.1 | 0.2 ± 0.0 | <i>NS</i> |
| C/N ratio | 30 ± 1 | 25 ± 1 | $p < 0.001$ |
| Moisture content (%) | 45 ± 3 | 57 ± 5 | <i>NS</i> |

Table 3.1: Results of chemical analysis of MSW Fines and Green Waste feedstocks prior to in-vessel composting. *NS* indicates $p > 0.05$.

3.4.2. In-vessel composter temperature data

Once the 50/50 waste mixture had been placed into the composting vessel, the temperature rose at a steady rate for the first 7 d until the ABPR temperature regulations had been met (Fig. 3.7). The rapid decrease shown on day 7 was caused by the unloading of the vessel to allow the composting material to be turned and reloaded. Once the material had been reloaded to the vessel, temperatures rose more slowly and took a further 2 wk to reach the required 60°C for over 48 h. The 70/30 waste mixture took a much shorter time period (overall) to obtain compliance with the 2005 ABPR; it also achieved a greater maximum temperature. Initially, the material took only 2 d to get above 60°C and remained around 67°C for a further 4 d until it was unloaded from the vessel, turned and reloaded. Once reloaded, temperatures rose swiftly again and remained above 70°C for 7 d; at this point, the temperature began to fall slightly and it was decided to unload the vessel and place the material in static piles to mature.

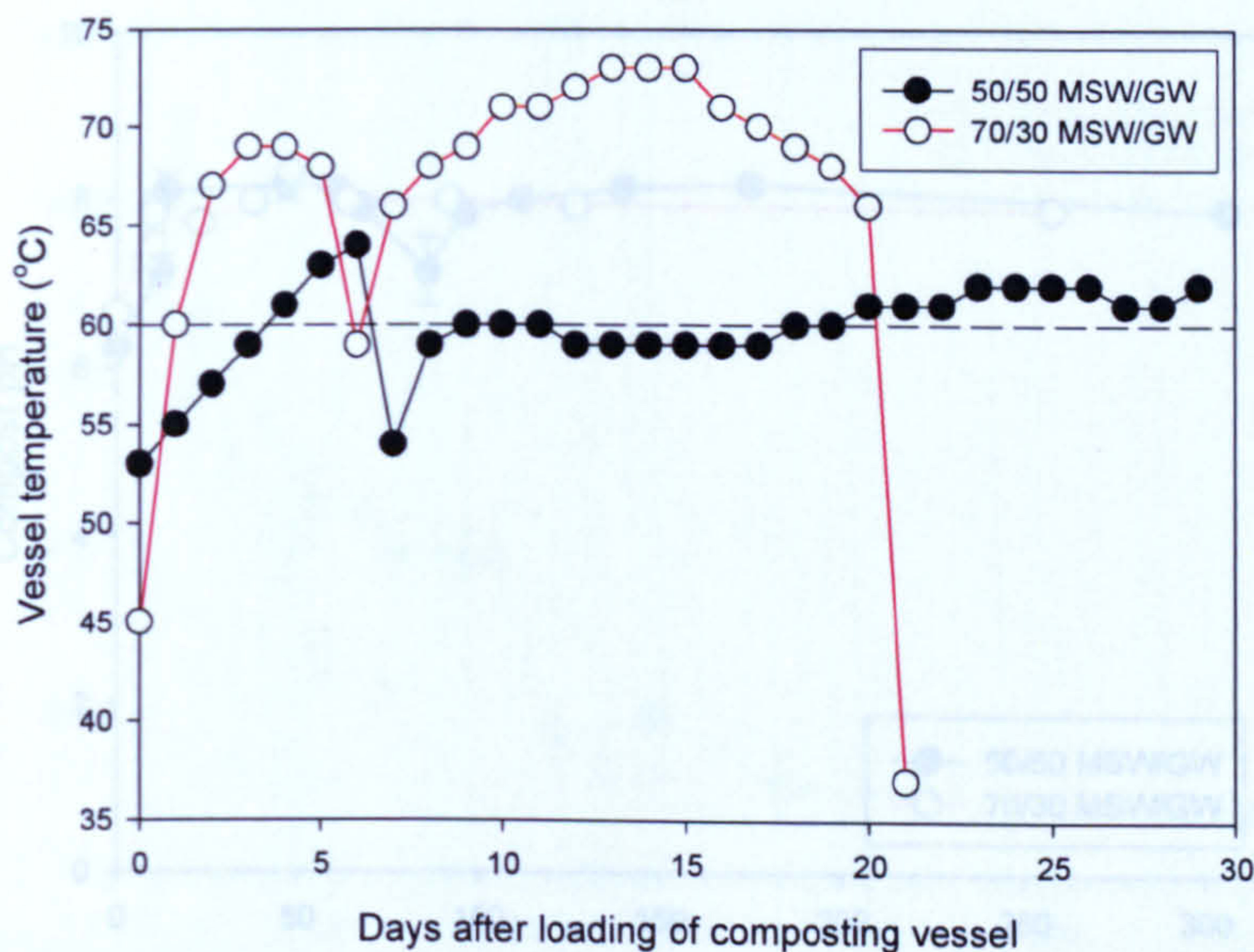


Figure 3.7: Changes in composting vessel internal temperature during composting of two mixtures of MSW Fines and Green Waste. The dotted line shows the ABPR temperature regulation standard.

3.4.3. Compost pH

The pH of the 50/50 waste mixture increased rapidly within the first 2 wk of composting (from 6.3 ± 0.0 to 8.1 ± 0.0 ; Fig. 3.8). For the next month, the pH remained around this value before decreasing rapidly to 7.2 ± 0.4 after 6 wk of maturing. For the remainder of this period, the pH remained around 7.8. The 70/30 waste mixture also showed an increase in pH during the initial composting period (approx 14 d), from 6.7 ± 0.0 to 7.8 ± 0.3 . The pH rose after 6 weeks of maturation before falling slightly back for the remainder of the maturation period. Significant differences between several of the individual 50/50 waste mixture samples was observed, whereas the only differences found in the 70/30 waste mixture samples were between the material analysed at $t = 0$ d and those analysed subsequently.

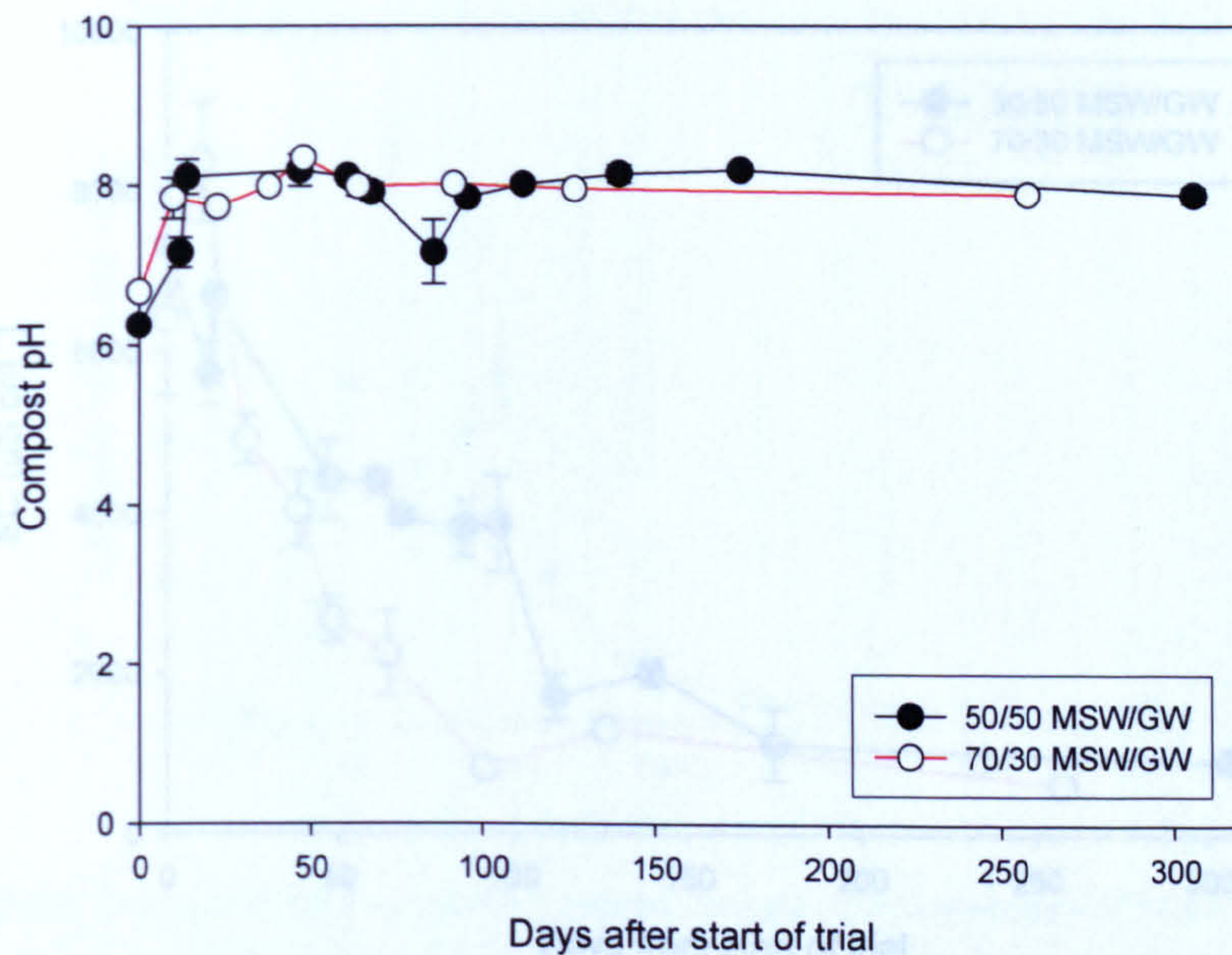


Figure 3.8: pH of two mixtures of MSW Fines and Green Waste during composting and subsequent maturation over an 11 month period. Values represent means \pm SEM.

3.4.4. Compost electrical conductivity (EC)

Generally, the EC of the compost decreased steadily over time in both waste mixtures both during the active composting and maturation phases (Fig. 3.9). Significant differences were identified between samples taken at the start and end of the composting process.

the NO_3^- content of the 70/30 waste mixture material remained low throughout the composting and maturation period. Significant differences were identified between some of the individual 50/50 samples but rarely in the 70/30 samples.

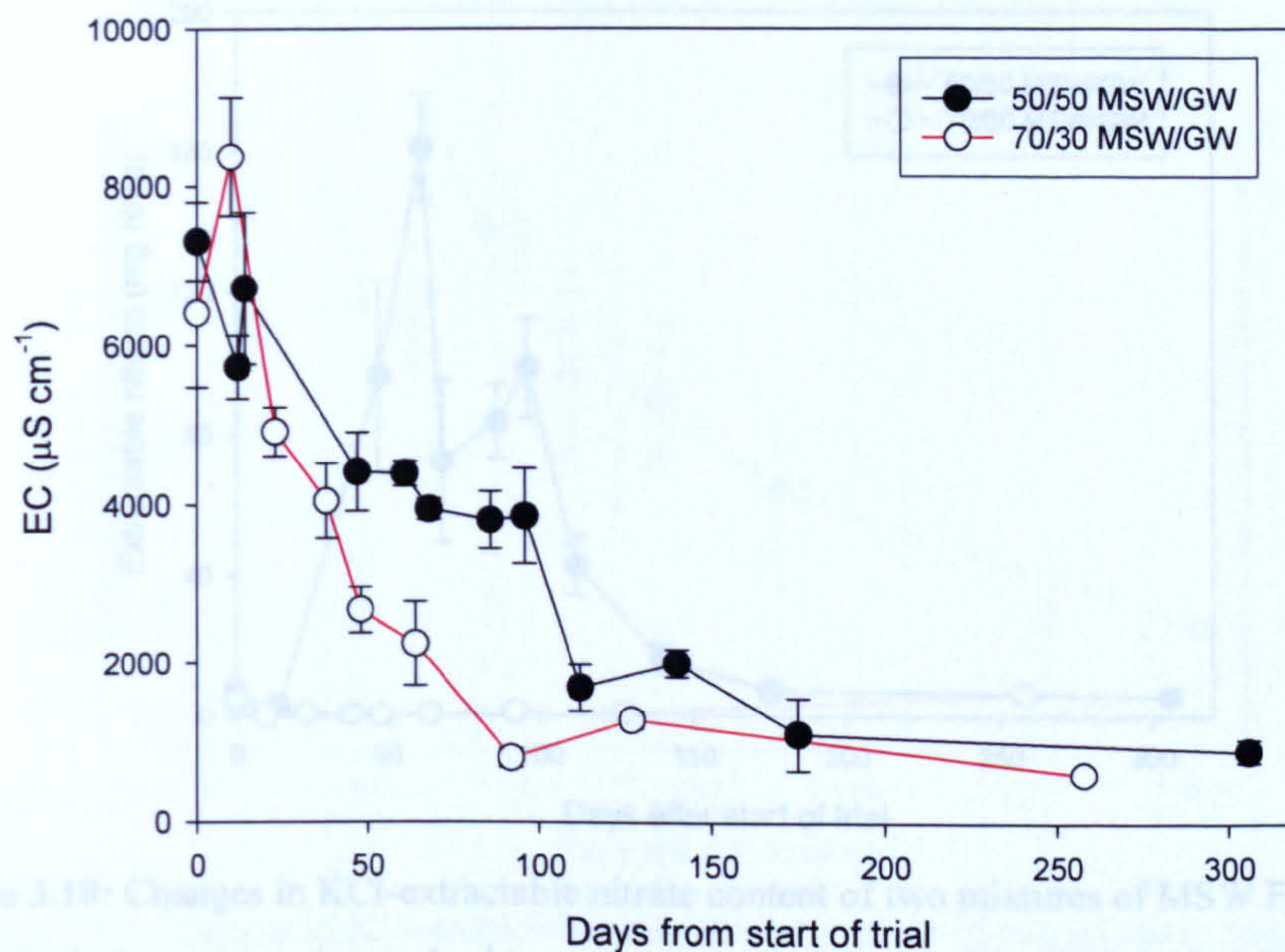


Figure 3.9: EC of two mixtures of MSW Fines and Green Waste during composting and subsequent maturation over an 11 month period. Values represent means \pm SEM.

3.4.5. Compost KCl-extractable NO_3^-

Initially, extractable NO_3^- in the 50/50 waste mixture rose reaching a peak 2 wk after unloading (Fig. 3.10). Towards the end of the trial, however, the NO_3^- content subsequently reduced, with the final levels reached being similar to those of the initial samples. In contrast, the NO_3^- content of the 70/30 waste mixture material remained low throughout the composting and maturation period. Significant differences were identified between some of the individual 50/50 samples but rarely in the 70/30 samples.

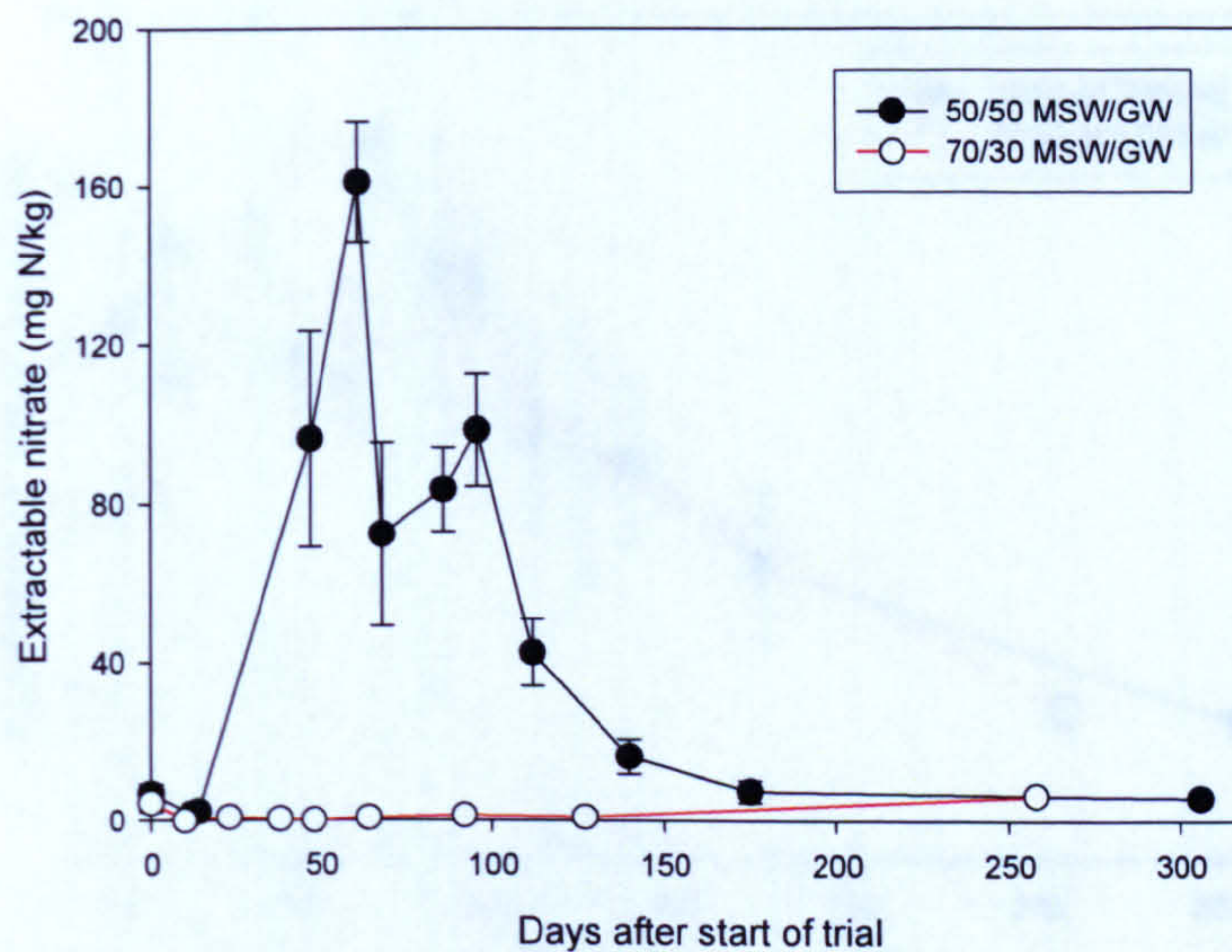


Figure 3.10: Changes in KCl-extractable nitrate content of two mixtures of MSW Fines and Green Waste during composting and subsequent maturation over an 11 month period. Values represent means \pm SEM.

3.4.6. Compost KCl-extractable NH_4^+

In both composts, NH_4^+ levels generally declined during the trial and especially during maturation (Fig. 3.11). The drop in NH_4^+ is most likely caused by losses of NH_3 by volatilization, however, this was not quantified here. In the 70/30 waste mixture, KCl extractable NH_4^+ increased for the initial 4 wk and then declined steadily for the remainder of the maturation period. Statistical analysis identified that the final sample of the 50/50 compost had significantly lower KCl extractable NH_4^+ than the samples taken prior to 2 months maturing, whilst the majority of individual samples showed no differences between one another. The final 70/30 sample contained significantly lower NH_4^+ than the majority of the other samples taken between the start and 2 months of maturation.

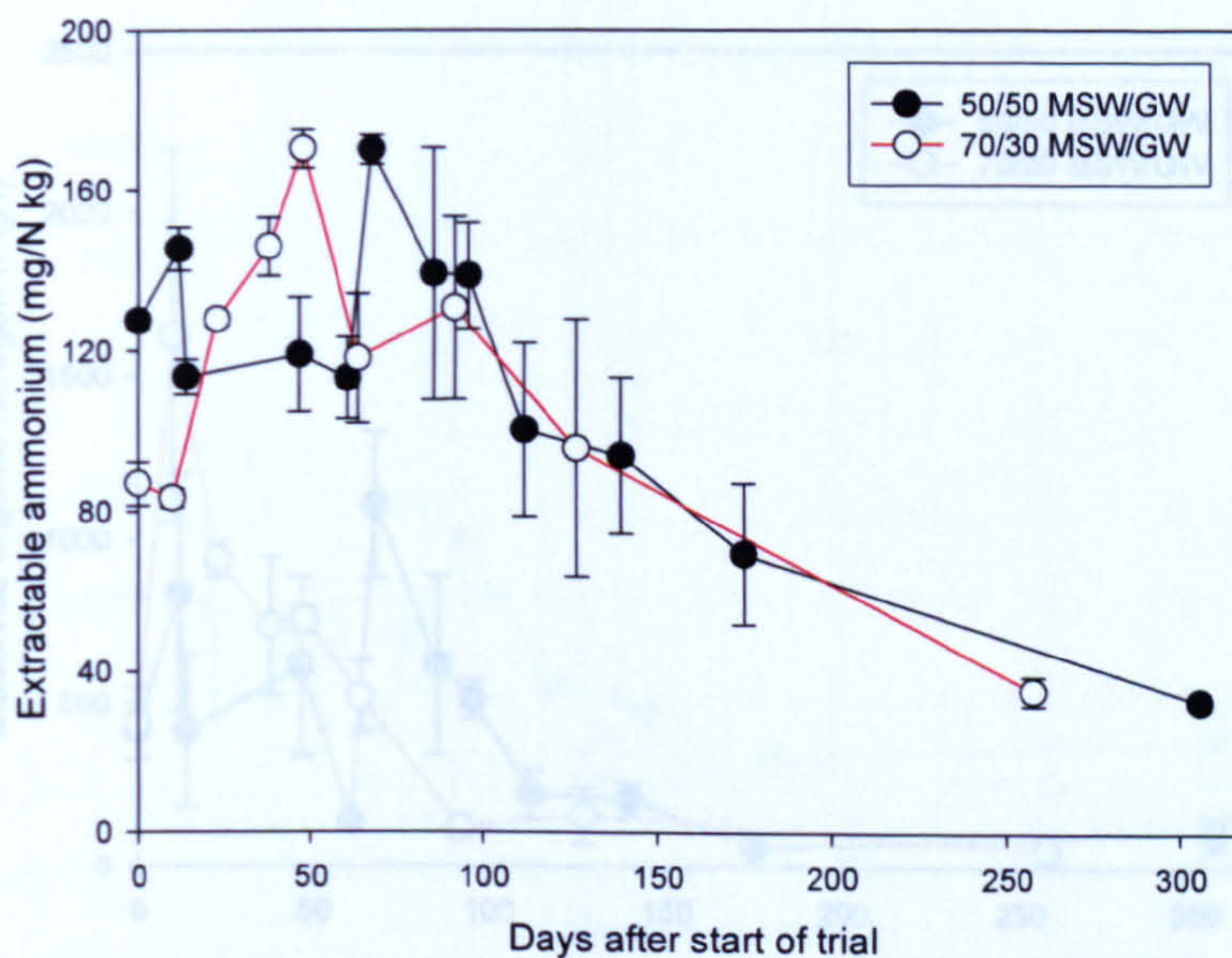


Figure 3.11: Changes in KCl-extractable ammonium content of two mixtures of MSW Fines and Green Waste during composting and subsequent maturation over an 11 month period. Values represent means \pm SEM.

3.4.7. Compost dissolved organic nitrogen (DON)

In the 50/50 waste mixture, DON constituted over 60% of the total dissolved N during the early stages of composting falling to 30% by the end of the composting process (Fig. 3.12). In comparison, the 70/30 waste mixture contained higher DON levels for the majority of the composting period, although this trend was reversed during the maturation phase. Significant differences were found between the individual samples of the 50/50 waste mixture and also between the samples of the 70/30 waste mixture.

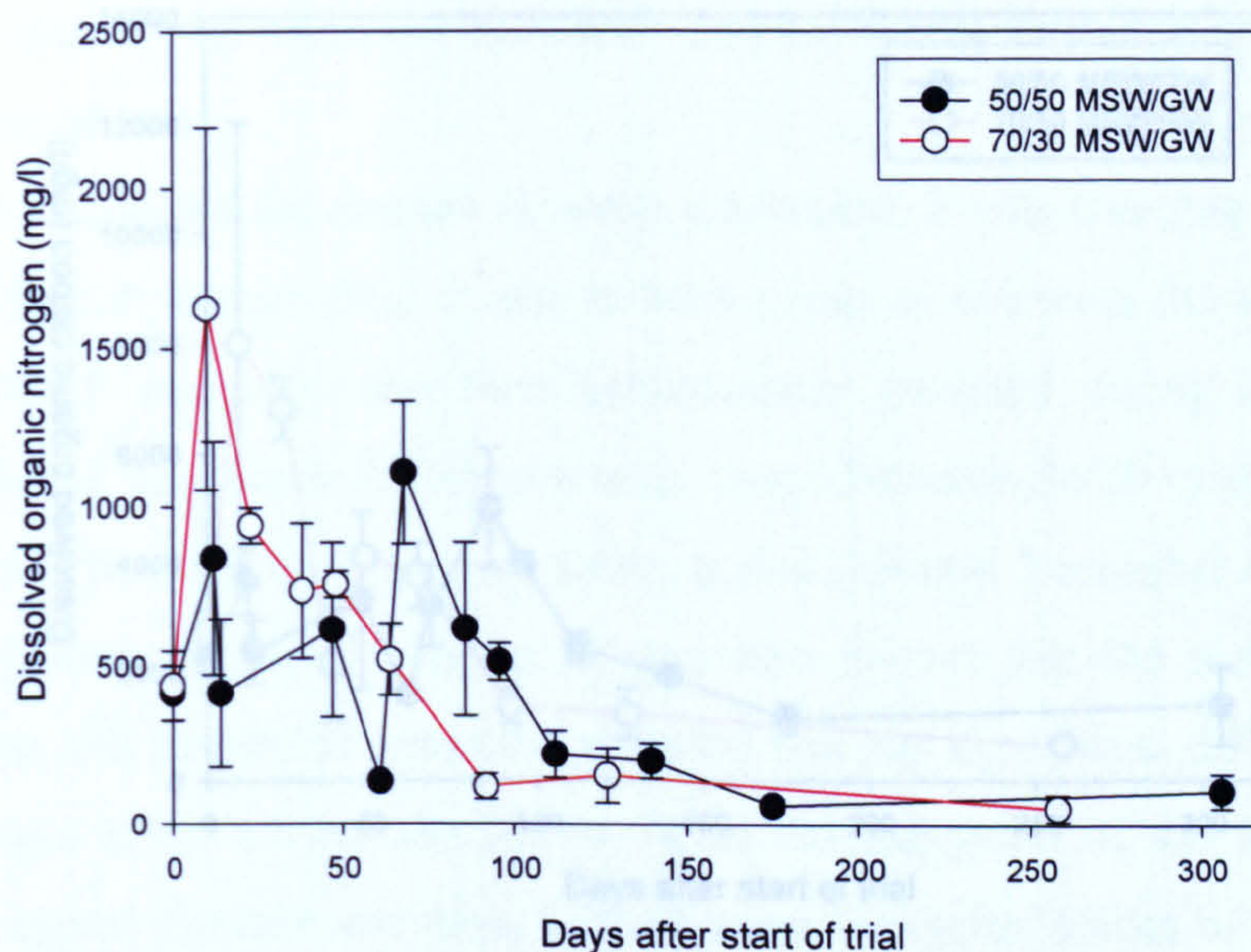


Figure 3.12: Changes in Dissolved organic nitrogen content of two mixtures of MSW Fines and Green Waste during composting and subsequent maturation over an 11 month period. Values represent means \pm SEM.

3.4.8. Compost dissolved carbon

Overall, there was an overall reduction in dissolved organic C (DOC) levels in both compost types over the duration of the composting period (Fig. 3.13). Significant differences were identified between some of the individual samples of the 50/50 and 70/30 waste mixtures. As can be seen from Figure 3.14, dissolved inorganic C (DIC) only accounts for a very small percentage of the total dissolved carbon (TDC) within a sample (between 0.7 and 3.3%). In this study, DIC reduced in the first 2 wk of composting before rising again to levels similar to those found in the initial sample after 3 months of maturation. There was very little overall change recorded in the amount of DIC present in the 70/30 compost, with levels reducing only slightly during the entire length of composting/maturation period. No significant differences were found between the DIC contents of any of the samples taken from either the 50/50 or the 70/30 materials.

3.4.9. Compost phosphorus

During the time spent within the composting vessel, the levels of Olsen extractable P in both waste mixtures declined, however, after this, levels remained relatively

constant (Fig. 3.15). Few significant differences were found between individual samples due to the variation between samples across both the composting and maturation periods.

A greater amount of water extractable P was less than that recovered in the Olsen P extract (Fig. 3.15). In both compost mixtures the levels of water extractable P increased during the composting period and then subsequently declined during the maturation period. Some significant differences were found between 50/50 mixture, especially those from the composting period (around 3 months) and those from the maturation period. The 70/30 mixture also showed that the material that had undergone the longest maturation period had no significant differences to the first sample taken 10 months before. With the exception of the final sample of 70/30 compost mixture and those taken during the earlier stages of maturation, no significant differences were found.

Figure 3.13: Changes in DOC content of two mixtures of MSW Fines and Green Waste during composting and subsequent maturation over an 11 month period. Values represent means \pm SEM.

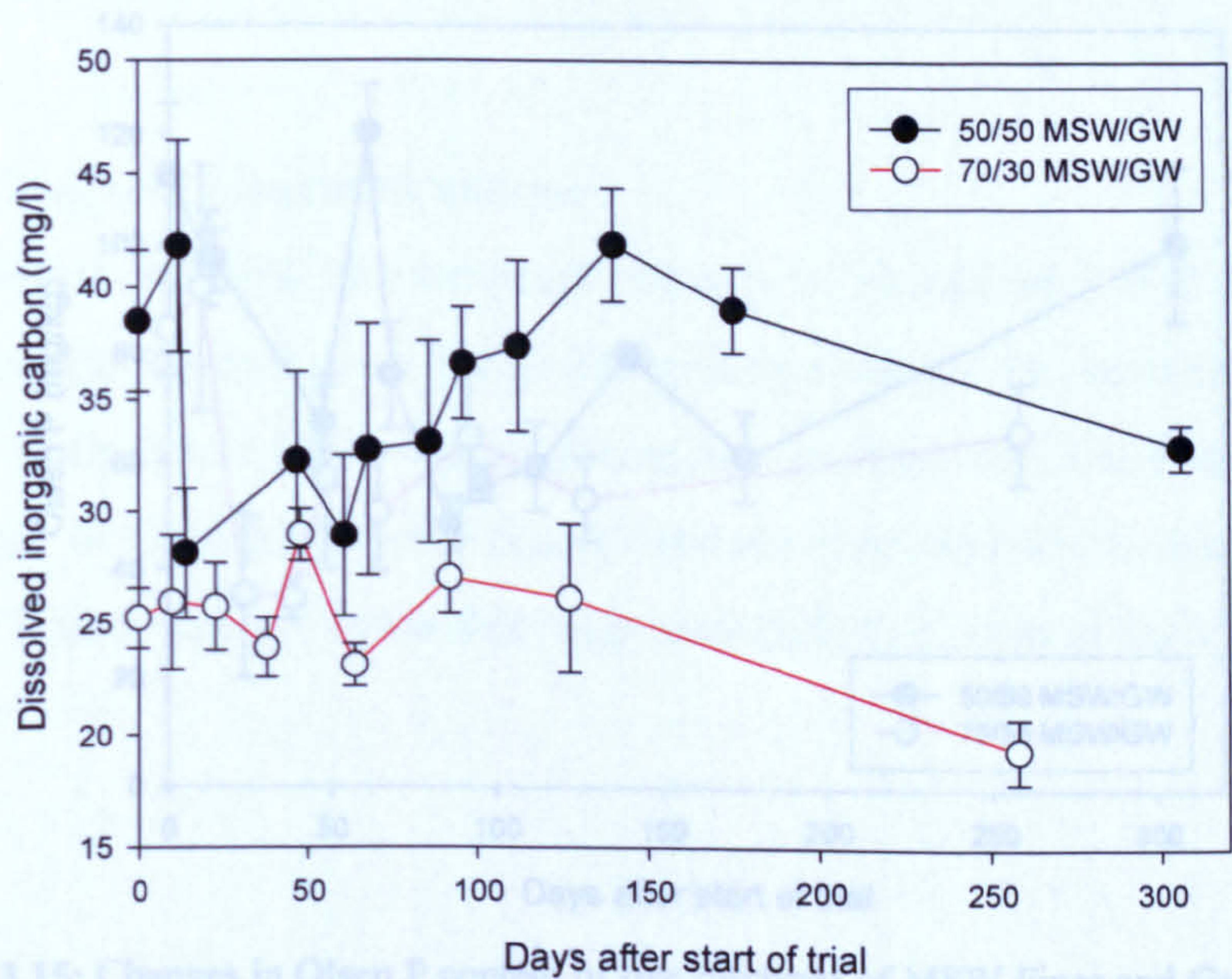


Figure 3.14: Changes in Olsen P content of two mixtures of MSW Fines and Green Waste during composting and subsequent maturation over an 11 month period. Values represent means \pm SEM.

Figure 3.15: Changes in DIC content of two mixtures of MSW Fines and Green Waste during composting and subsequent maturation over an 11 month period. Values represent means \pm SEM.

3.4.9. Compost phosphorous

During the time spent within the composting vessel, the levels of Olsen extractable P in both waste mixtures declined, however, after this, levels remained relatively

constant (Fig. 3.15). Few significant differences were found between individual samples due to the variation between samples across both the composting and maturation phases.

As expected the amount of water extractable P was less than that recovered in the Olsen P extract (Fig. 3.16). In both compost mixtures the levels of water extractable P increased and then subsequently declined during the maturation period. Some significant differences were found between 50/50 mixture, especially those from the middle of the maturation period (around 3 months) and those from the composting period. However, it was also shown that the material that had undergone the lengthiest maturation period had no significant differences to the first sample taken 10 months before. With the exception of the final sample of 70/30 compost mixture and those taken during the earlier stages of maturation, no significant differences were found.

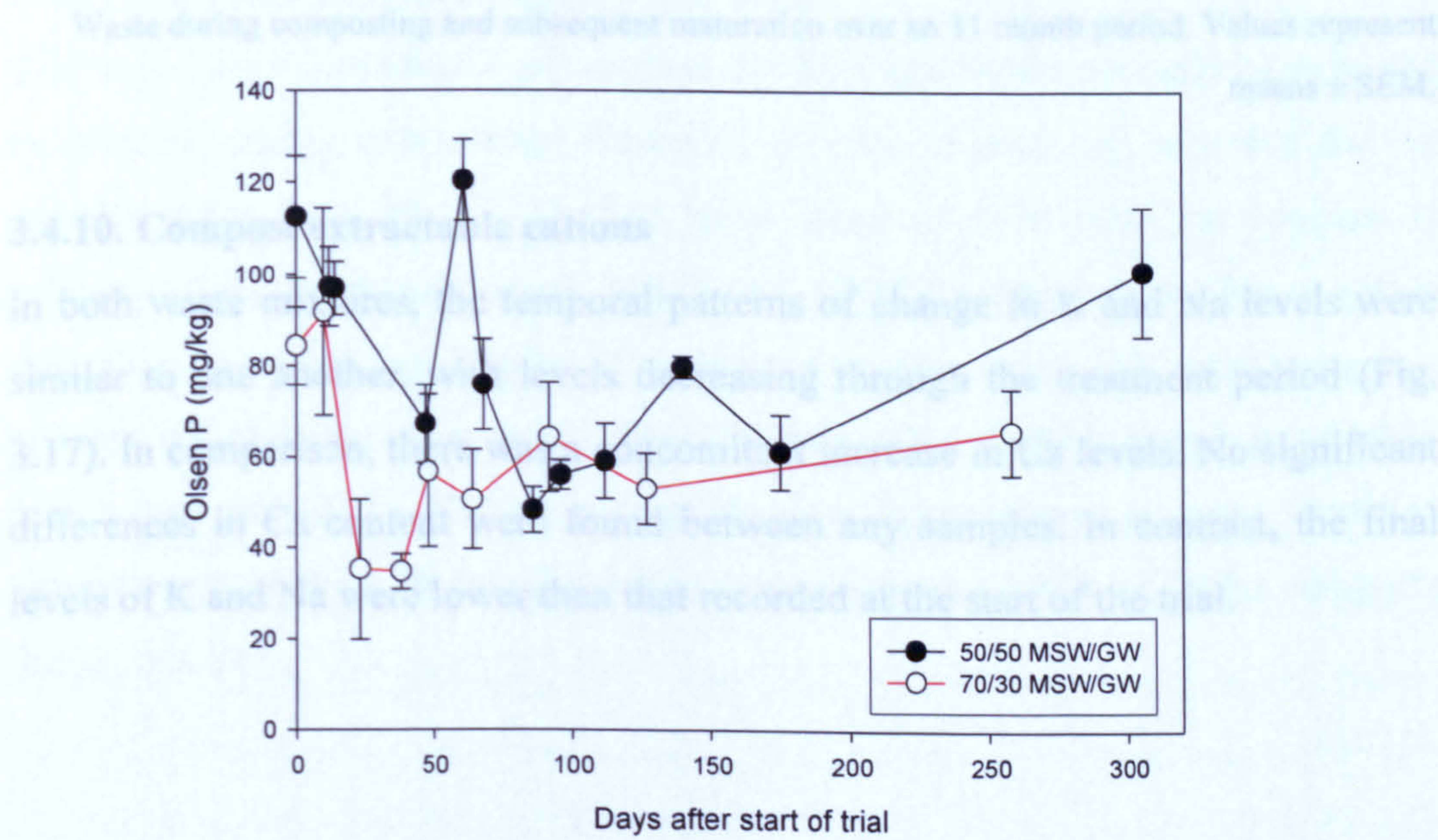


Figure 3.15: Changes in Olsen P content of two mixtures of MSW Fines and Green Waste during composting and subsequent maturation over an 11 month period. Values represent means \pm SEM.

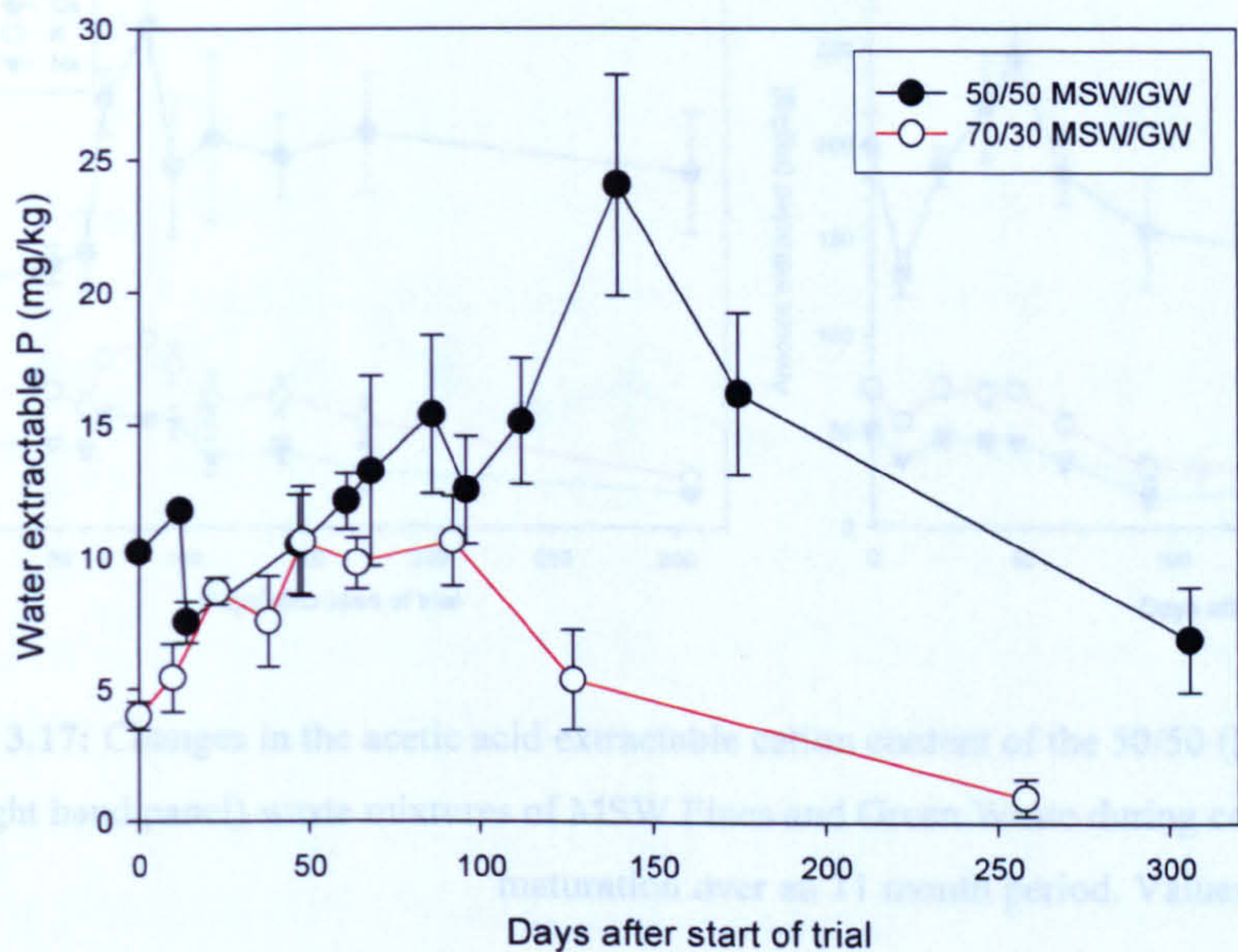


Figure 3.16: Changes in water extractable P content of two mixtures of MSW Fines and Green Waste during composting and subsequent maturation over an 11 month period. Values represent means \pm SEM.

3.4.10. Compost extractable cations

In both waste mixtures, the temporal patterns of change in K and Na levels were similar to one another, with levels decreasing through the treatment period (Fig. 3.17). In comparison, there was a concomitant increase in Ca levels. No significant differences in Ca content were found between any samples. In contrast, the final levels of K and Na were lower than that recorded at the start of the trial.

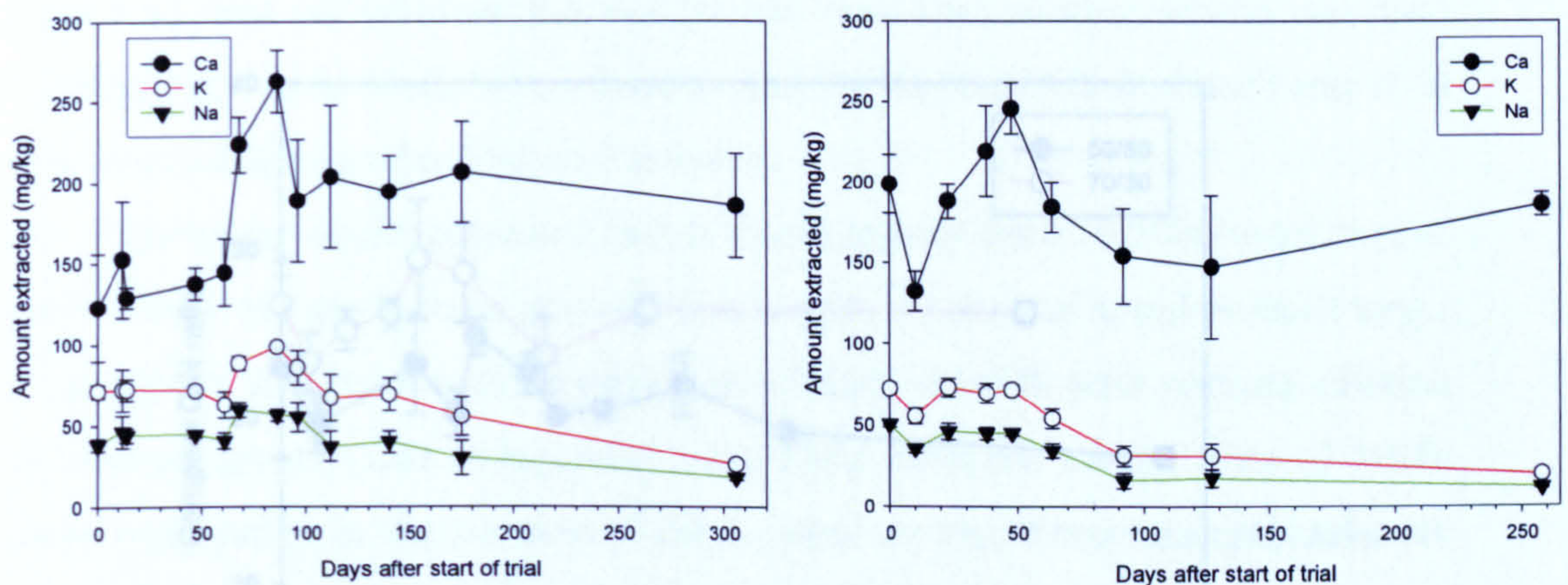


Figure 3.17: Changes in the acetic acid extractable cation content of the 50/50 (left hand panel) and 70/30 (right hand panel) waste mixtures of MSW Fines and Green Waste during composting and subsequent maturation over an 11 month period. Values represent means \pm SEM.

3.4.11. Compost carbon/nitrogen (C/N) ratio

C/N is an important indicator of compost maturity and would normally be expected to decrease during composting. However, no such change was recorded for the samples taken here (Fig. 3.18). The usual range of C/N ratios for compost is between 15-25 (De Bertoldi *et al.*, 1983), with all of the samples here falling within this range; therefore the material, if considered in terms of its C/N ratio alone, is a suitable feedstock for composting. The C/N ratio of the 70/30 material was slightly higher than that of the 50/50 compost. No significant differences were identified between any of the C/N samples taken from either the 50/50 or 70/30 composts during this trial.

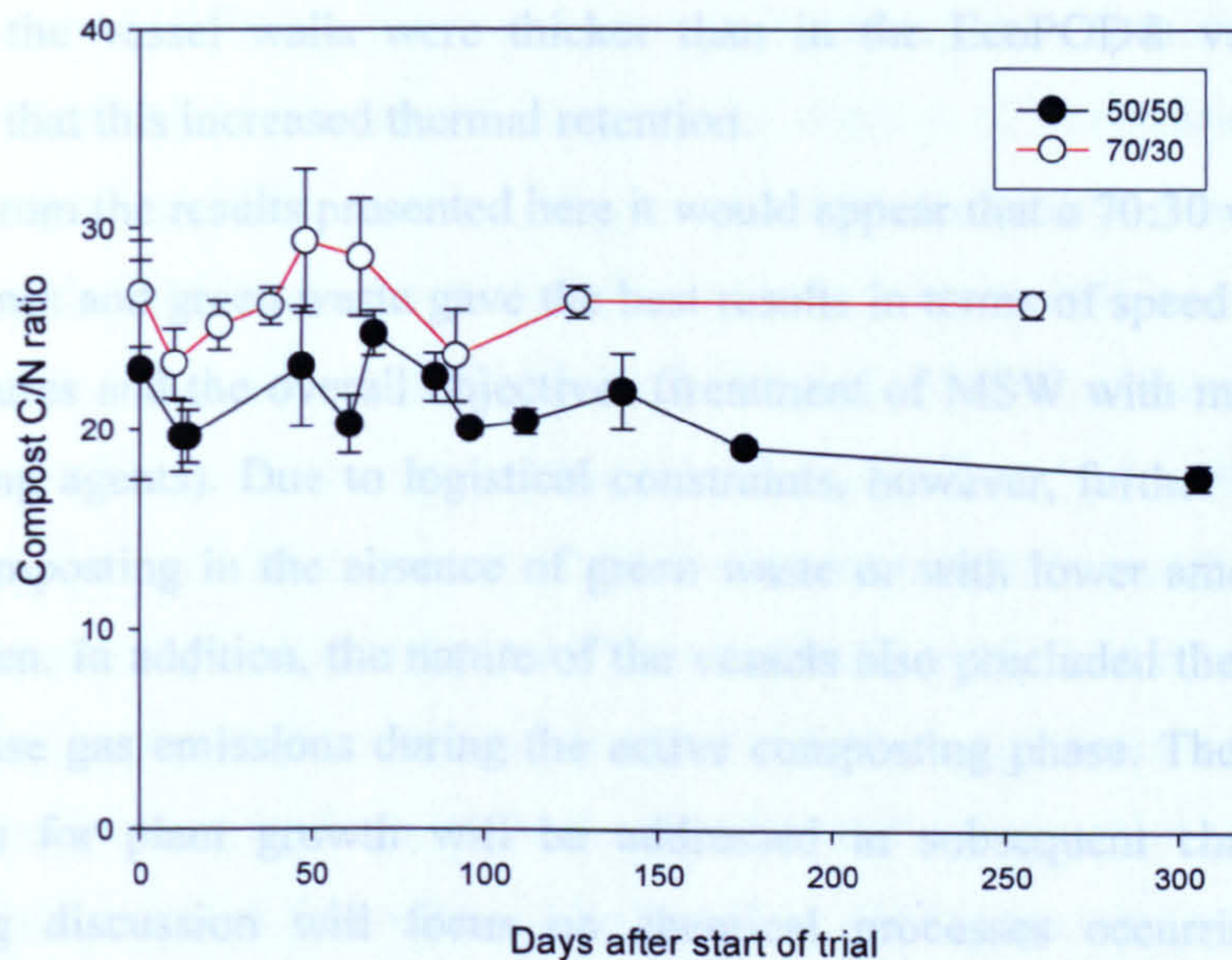


Figure 3.18: Changes in C/N Ratio (solid) of two mixtures of MSW Fines and Green Waste composted and matured over 11 months. Values represent means \pm SEM.

3.5. Discussion

3.5.1. Success of MSW Fines composting

The experiments conducted here confirm previous trials showing that MSW Fines can be successfully composted on a commercial scale. Unlike previous trials presented in this thesis, however, this trial also showed that MSW Fines could be composted in compliance with the 2005 ABPR regulations (i.e. capable of reaching high temperature for a prolonged period). This should effectively ensure sanitization of the material in the vessel and allow greater end-uses and market value for the resulting compost product. This sanitization not only reduced the risk to human health but also prevents the spread of weed propagules and plant pathogens (Robertson and Morgan, 1995). It should also allow for the production of a more aesthetically pleasing product (Bernal *et al.*, 1998b). The ability of the vessel to achieve ABPR compliance is ascribed to the larger volume of the vessel and the greater critical mass of compost contained within it (the capacity of compost to heat up is inversely correlated with its surface area-to-volume ratio). In addition, the air circulation regime was directly linked to the vessel's thermal

regime so that air addition did not induce excessive cooling of the compost. Thirdly, the vessel walls were thicker than in the EcoPOD® vessel and it is assumed that this increased thermal retention.

From the results presented here it would appear that a 70:30 w/w mixture of MSW Fines and green waste gave the best results in terms of speed to reach target temperatures and the overall objectives (treatment of MSW with minimal dilution by bulking agents). Due to logistical constraints, however, further trials of MSW Fines composting in the absence of green waste or with lower amounts were not undertaken. In addition, the nature of the vessels also precluded the assessment of greenhouse gas emissions during the active composting phase. The quality of the composts for plant growth will be addressed in subsequent chapters and the following discussion will focus on chemical processes occurring during the composting and maturation process.

3.5.2. pH

The pH of the MSW-derived materials tested here was found to be similar to previous reports stating that 6 month old MSW compost MSW tends to have a pH between 7 and 8 (Iglesías-Jiménez and Perez-Garcia, 1989; Keeling *et al.*, 1994). Overall, the changes in pH were small. The stabilization of pH at around 8.0 is most likely to be linked to the production of $\text{NH}_3/\text{NH}_4^+$ and buffering from inorganic carbonates. The transitory drop in pH in the 50/50 waste compost after 2 months directly correlates with a peak in NO_3^- accumulation, the production of which is known to be an acidifying process (i.e. nitrification; $\text{NH}_4^+ + \text{O}_2 \rightarrow \text{NO}_3^- + \text{H}^+$). The drop in pH of the 50/50 mixture could also be related to a reduction in microbial N mineralization (Schefferle, 1965) whereby less NH_4^+ is produced, lowering the pH.

3.5.3. Salt content (EC)

Using the same criteria as Villar *et al.* (1993), it can be concluded that during the composting phases and the early stages of maturation, both composts were highly saline ($\text{EC} > 8 \text{ mS cm}^{-1}$) and application of these materials to land at this stage of

the process could lead to crop damage. It is apparent from the results that the EC progressively fell during the composting and subsequent maturation phase. This can be ascribed to losses of ions by a variety of mechanisms including: volatilization (e.g. $\text{NH}_4^+_{(\text{aq})} \rightarrow \text{NH}_{3(\text{g})}$), microbial immobilization (e.g. $\text{NO}_3^- \rightarrow \text{microbial-N}$), leaching to the base of the vessel (e.g. Na^+ and K^+) and the formation of insoluble carbonates (e.g. $\text{Ca}^{2+} \rightarrow \text{CaCO}_3$). The relative importance of each mechanism, however, remains unknown. In the Villar *et al.* (1993) study, the finished MSW-based composts had EC values ranging between 7 to 12 mS cm^{-1} which is much higher than a typical UK agricultural soil (0.1 to 0.5 mS cm^{-1}) and if applied in large quantities to land it could induce a significant reduction in soil quality (i.e. increased salinity and associated loss of soil structure plus a disruption of microbial activity). In contrast, Keeling *et al.* (1994) recorded an EC of 0.9 mS cm^{-1} for a 6 month old MSW-derived compost which is similar to that reported here (0.9 \pm 0.1 mS cm^{-1} for the 50/50 mixture and 0.6 \pm 0.1 mS cm^{-1} for the 70/30 mixture). Therefore, it is concluded that it is possible to have these materials classed as non-saline, increasing their number of potential end uses.

3.5.4. Nitrogen

Bernal *et al.* (1998b) showed that maturation of composted organic wastes generally resulted in an increase in NO_3^- levels, which in turn would provide higher amounts of plant-available N. The overall pattern of the 70/30 mixture was one whereby levels increased, with the maturation phase showing a marked increase. Iglesias-Jiménez and Perez-Garcia (1989) state that the increase in NO_3^- occurs largely during the third and fourth month of composting (note, not maturation); none of the data recorded here appears to concur with this. García- Gómez *et al.* (2003) also state that material with a low starting NO_3^- content is unlikely to show much N loss by denitrification; therefore the material analysed here is more likely to have loss N by way of other pathways (e.g. volatilization or microbial immobilization).

NH_3 volatilization must be considered as a potential route of NH_4^+ loss during composting as the pH of both mixtures rose to over 7 within the early stages

of the composting process and the temperatures reached were high enough to facilitate removal gaseous N (Witter and Lopez-Real, 1988). This loss pathway would also have been enhanced by the forced aeration of the vessel which would further drive volatilization.

Microbial activity is a highly important facet of both N dynamics and composting. Therefore any slight inhibitory effects caused by changes in environmental conditions could lead to a reduced composting rate. Gosz and White (1986) state that the presence of certain organic compounds within samples result in inhibition of microbial action, reducing nitrification; something that is also attributable to high NH_4^+ levels and several other factors. In this trial, NH_4^+ was present in much greater concentrations than NO_3^- leading to the conclusion that nitrification was indeed retarded here either by: (1) the presence of elevated NH_4^+ levels, (2) the lack of a nitrifying bacteria inoculum (i.e. zero numbers), (3) out competition of nitrifying bacteria by other organisms present (i.e. low numbers), (4) inhibition of growth of nitrifying bacteria above 40°C (Bernal *et al.*, 1998b), or (5) inhibition of the nitrification process by some other chemical factor (i.e. sub-optimal conditions). It would be interesting to inoculate or seed mature compost with nitrifiers (e.g. *Nitrosomonas* and *Nitrobacter*) to try and elucidate which of the factors above is responsible for the low rates of nitrification.

The ratio of NH_4^+ to NO_3^- in compost is often used to assess compost “stability” or “maturity”. If the $\text{NH}_4^+:\text{NO}_3^-$ ratio is lower than 1 this is generally perceived to be good (Bernal *et al.*, 1998b). The ratios of the final matured compost in this study were 5.6 ± 0.3 for the 50/50 mixture and 5.7 ± 0.8 for the 70/30 mixture. High ratios indicate poor composting and unfavourable environmental conditions for nitrifying bacteria (Villar *et al.*, 1993). In addition to the points raised above, this may also indicate that anaerobic conditions may have prevailed in the vessels due to a uniform lack of aeration (Iglesías-Jiménez and Perez-Garcia, 1989; Britto and Kronzucker, 2002; Wong, 1985). The two compost mixtures here exhibit qualities that suggest they are not of good quality using this indicator.

3.5.5. Total dissolved C

Total dissolved C and dissolved organic C were both observed to fall during the maturation phases of both mixtures which is consistent with previous studies (Bernal *et al.*, 1996). The 50/50 mix experienced a greater level of variation between samples compared to the 70/30 mix; this experienced a rise during the first week of composting, before steadily declining for the remainder of the trial. Wu *et al.* (2000) showed that increased composting and maturation times resulted in a reduced DOC content, although the DOC contents of some materials have been known to plateau during the maturation (curing) phase. A decrease in DOC concentration indicates the occurrence of mineralization of organic matter by microbial activity (García-Gómez *et al.*, 2003). Whilst this is to be expected to some degree, the increase in TDN concentration caused by the speed of OM degradation exceeding the rate of NH_3 volatilization (Bernal *et al.*, 1998b), does not occur here; as such, the C/N ratio does not decrease as expected. It can therefore be concluded that N was lost via NH_3 volatilization may be faster than organic matter was being degraded.

3.5.6. Solid C/N ratio

Many studies have looked at the possibility of using solid C/N ratio as a way of assessing the maturity of composted organic materials, however, there is no real consensus on guideline values. Iglesias-Jiménez and Pérez-García (1992) state that a C/N ratio of <12 indicates that MSW-based composts are mature enough for land application without significant alteration of the soil microbial balance. However, Allison (1973) found that C/N ratios <15 do not deleteriously affect the soil microbial community. Alternatively, Zucconi and de Bertoldi (1987) recommended that a C/N ratio of <22 is sufficient to indicate maturity while Parker and Sommers (1983) recommend that the C/N ratio should be <20 to avoid N immobilization after application to soil. Conversely, those composts with a C/N value of <12 (often classed as very mature) can release large amounts of N into soils, yet the rate of mineralization decreases with time due to the switch from labile organic N to more recalcitrant forms such as that from microbial biomass (Iglesias-Jiménez and

Alvarez, 1993). Both the 50/50 and 70/30 mixtures do not have C/N ratios within these beneficial ranges, which would suggest that they were not to be considered completely mature after 6 months in static piles. One possible explanation for the reduction in decomposition (and hence the lack of a decreasing C/N ratio) could be due to the composting taking place during winter months when lower ambient temperatures reduce microbial activity (Schefferele, 1965). It should also be noted that contamination of samples by inert materials such as glass and plastic fragments may also bias the C/N ratio of the solid phase (Iglesias-Jiménez and Alvarez, 1993).

The majority of C/N ratios quoted in previous studies have been based on calculations made from data in the solid phase. Another school of thought is that calculation C/N ratio using water extractable C and N (i.e. C_w/N_w) would give a more realistic reading for the maturity of a material, as the majority of interactions between biochemically active compounds and roots/soil microbes occur in the aqueous phase. Such analysis is stated in several studies (Iglesias-Jiménez and Perez-Garcia, 1992; Senesi, 1989; Bernal *et al*, 1998b) as helping to reduce the geographical variation between samples of MSW-derived composts caused by differing consumption/production habits across the globe. The Iglesias-Jiménez and Perez-Garcia (1992) study references other trials (Chanyasak and Kubota, 1981; Chanyasak *et al.*, 1982) that had calculated this value and produced a figure of between 5 and 6 for stable MSW-based compost; this uses the ratio of Organic-C/Organic-N. The trial conducted here using the MSW Fines and Green Waste was found to have a variety of values for this ratio; it was also found that the C_w/N_w ratio actually increased during this trial. One problem identified with this method is that the concentration of organic-N is often low in water extracts from mature composts according to Bernal *et al.* (1998b).

3.5.7. Phosphorus

Comparing the water extractable-P to the Olsen extractable-P led to the conclusion that as the MSW Fines composts matured, P has become more available within the compost. This is likely to be due to the progressive degradation of organic matter,

in which the P is contained in organic forms (e.g. DNA, phospholipids etc). Degrading OM causes a release of tightly bound P and allows more to be removed by water, a weaker extractant than the NaHCO_3 used in the Olsen P method.

3.5.8. Maturity

Bernal *et al.* (1998b) recommend the use of a 2 month “Safety Period” between compost application to soil and the sowing of plants to allow immobilization of the available N provided by the compost. Many of the results obtained during this study showed significant differences between samples taken at the start of composting and those taken after 2 months of maturation. However, it is likely that a greater period of time could be used to ensure maturity. Wong (1985) does not give any chemical indicators of maturity, yet mentions a 4 month maturation period as being the minimum which would allow the composted material to be used as a soil conditioner; Keeling *et al.* (1994) mentions that stabilization of MSW-based compost takes anything up to 6 months or maybe longer, further showing a lack of consensus in the literature. In practical terms, it is clear that MSW Fines composting is a commercial operation and that longer maturation periods means greater costs, an increased need for storage space and the greater potential for environmental problems (e.g. leachate or odour generation due to greater volumes on site). Overall, waste operators want to maximise throughput. This goal may not be consistent with producing the best end-product and the economic tradeoffs involved in balancing these two aims requires serious consideration

The majority of indices used as indicators of maturity show that neither the 50/50 or 70/30 mixture composts could be truly considered stable/mature after 6 months of maturation. This could be due to the presence of certain compounds within the parent material, a lack of sufficient time within the composting vessel or a lack of suitable conditions for maturation. Further studies into this process with differing conditions of maturation would be recommended.

3.6. Conclusions

- The in-vessel composting temperatures proved sufficient to meet the conditions of the 2005 Animal By-Products Regulations (ABPR) for both feedstocks. From a sanitization perspective, the 70/30 MSW Fines-to-Green Waste appeared to be the best composting mixture based on the thermophilic temperatures reached.
- Compost pH increased significantly in both mixtures during both the composting and maturation phase. It is speculated that this could possibly leading to excessive loss of NH_3 in a forced aeration in-vessel composting system.
- The EC readings for both composts were found to be much lower than other studies that had composted MSW for similar time periods. There appears to be little issue with excessive salinity and the subsequent potential for using the matured compost as a growth medium or soil improver.
- Large losses of soluble N were seen during composting. It is possible that N lost by volatilization or microbial immobilization prevented the conversion of NH_4^+ to NO_3^- . This may negatively affect its use as a growth medium or soil enhancer.
- The C/N ratio of both composted materials fell within the accepted “beneficial” range for suitable end-use.
- The liquid state C_w/N_w ratio of both materials was observed to increase during composting, and not to decrease as anticipated. Again this could be due to excessive microbial N immobilization.
- Neither of the MSW-derived composts could be considered to be mature when compared to several widely accepted maturation standards.
- There appears to be a serious trade off between optimising the rapid throughput of compost that is ABPR compliant with one that also retains greatest economic value in the final product. Further economic analysis is required to resolve this.

3.7. References

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Evaluation of MSW-based composts as growth media in comparison to commercially available compost varieties.

Mr Richard W. Furniss and Prof David L. Jones

Evaluation of MSW-based composts as growth media in comparison to commercially available compost varieties.

4.1. Abstract

With the increasing move away from landfill as the primary method of waste disposal and the increase in methods of waste treatment such as composting, there is also an increase in compost produced. These materials have many potential uses but the quality of the compost is closely related to the suitability for horticultural use. Addition of organic materials to land has many beneficial effects (e.g. improved porosity/nutrient content). However, there are also many negative effects associated with this including the addition of phytotoxic substances (e.g. heavy metals/organic acids). Therefore, this study considered the use of composts produced using in-vessel composting methods from the biodegradable fraction of Municipal Solid Wastes (MSW) when mixed with Green Waste (GW). The aims of this study were firstly to establish the effect of increased maturation period of MSW based composts on yields of two common pasture land species (*Lolium perenne* and *Trifolium repens*). Secondly, the effect on yield of the two species by adding composted MSW Fines to agricultural soils in differing proportions was recorded. It was found that yields of both species testing increased with longer maturation periods; this is likely to have been caused by the reduced phytotoxicity of the composts. Mixing MSW composts with soils increased the yields of *Lolium perenne* compared to soil alone. Conversely, adding composts to soils resulted in a reduced yield of *Trifolium repens* compared to soil alone. As such, it is concluded that addition of compost to soil can have species specific effects; a result of this nature means that there is limited potential for using MSW composts in agriculture/horticulture. Therefore, it would be favourable to use these materials where there is reduced risk to human/animal health (e.g. short rotation coppicing) if these were found to be financially and practically viable.

4.2. Introduction

Addition of composted MSW and GW to agricultural soils has become a common practice in many regions of the world as a way of recycling organic nutrients to soil. This practice is more common in those regions where soil organic matter contents are critically low (e.g. around the Mediterranean Sea) or where the intrinsic soil quality is poor (e.g. highly weathered or acidic soils in Asia) (Feng *et al.*, 1999; García-Gil *et al.*, 2004; Koerner *et al.*, 2008). As the levels of organic wastes being produced by society are increasing (Pascual *et al.*, 1997), their use as organic fertilizers provides an alternative path to landfill. This is particularly relevant considering the economic pressures facing agriculture and the dwindling reserves of some fertilizers (e.g. P; Cordell *et al.*, 2009).

Adding organic materials such as MSW to soils can have many positive effects upon the biological, chemical and physical properties of the soil (e.g. porosity, structure, nutrient content) (Gallardo-Lara and Nogales, 1987; Pascual *et al.*, 1997). However, this is not a one-way street as there are many negative effects that can result from the practice of MSW disposal to land. This can include the presence of phytotoxic substances such as heavy metals and organic acids due to incomplete decomposition of the parent material (Chanyasak *et al.*, 1983; Iglesias-Jiménez and Pérez García, 1989; Manios *et al.*, 1989; Pascual *et al.*, 1997; Zheljazkov and Warman, 2003). In addition, physical and biological contaminants may also be present (e.g. glass and metal fragments, human pathogens). Processes such as composting can be used to effectively reduce the phytotoxicity of raw waste and stabilize the organic matter in wastes (Witter and Lopez-Real, 1988). Overall, the more mature a compost has become, the “safer” it is for use (Bernal *et al.*, 1998c). However, insufficient processing time (i.e. decomposition) can be equally detrimental to the material and reduce the amounts of end uses that are possible; this is often caused by the fact that some microorganisms produce phytotoxic substances in unstable/immature composts (e.g. excess NH_4^+ , volatile fatty acids, organic acids; Bernal *et al.*, 1998). Adding immature materials to soil can also cause the immobilization of nutrients from the soil and compost, exacerbating phytotoxicity (Goyal *et al.*, 2005). Biological blockage of soil-

available nitrogen is one such example of this process (Iglesias-Jiménez and Pérez García, 1992).

In a UK context, more MBT-derived waste will be produced for the foreseeable future and there is a need to critically assess the options for the disposal of this biologically treated fraction. While it could be used as a refuse-derived fuel this does not close the nutrient recycling loop and would be a poor use of the resource. More appealing is the disposal of MBT- and MSW-derived composts to land as this does help close the nutrient recycling loop to some extent (e.g. not for foodstuffs produced overseas where the site of production and disposal are not linked). There is therefore a perceived need to assess the potential negative and positive aspects of adding MBT- or MSW-derived composts to agricultural or post-industrial land in the UK.

Due to strict regulations governing composting practices within the UK and their subsequent use on land, there is limited scope for carrying out field-based trials with non-Animal By-Products Regulations (2005) (ABPR) compliant composts such as those derived from MSW. Consequently, it was decided that a glasshouse-based plant growth trials would be undertaken to assess compost quality. The greenhouse also provides a greater degree of atmospheric control, reducing the variables that may affect the way plant species react to being grown in composted MSW (e.g. precipitation and temperature).

For this trial, there were two main aims:

- To establish the effect of increased maturation period of municipal waste-derived composts upon the yields of two common plant species;
- To record any differences in yields of two common plant species grown in mixtures consisting of differing proportions of two agricultural soils and two MSW-derived composts.

The two plant species chosen for the study represented those commonly found in UK agricultural grasslands and those widely used in experiments of this type, namely Perennial Ryegrass (*Lolium perenne* L.) and White Clover (*Trifolium repens* L.).

4.3. Materials and methods

4.3.1. Source materials

The plants growth trials were split into two sections, firstly to assess the impact of maturation period upon yield and, secondly, to assess the changes in yields when soils and composts were mixed in differing proportions. For the purposes of the first section of this trial, the two MSW-based composts were derived from wastes containing either:

- 50% Municipal Solid Waste Fines and 50% Green Waste
- or
- 70% Municipal Solid Waste Fines and 30% Green Waste

as detailed in Chapter 3 of the thesis. Briefly, each of these mixtures had undergone a period of “composting” using a 2005 ABPR compliant in-vessel composting system employing forced aeration. After removal from the vessel, the materials were matured for short periods in static piles prior to the commencement of the trial. As these mixtures were deemed to be susceptible to changes in their chemical nature, the material used for the growth trial was re-sampled every three weeks for the analysis of pH and electrical conductivity (EC).

4.3.2. Plant growth trials with composts alone

The first growth trial consisted of sowing 4 replicates of each of the two compost types with either *Lolium perenne* or *Trifolium repens*. Thirty seeds of either species were sown into individual opaque plastic pots (1000 cm³ volume) and were arranged randomly within the glasshouse. Each pot was watered daily and received no inputs of chemicals other than spraying for pests during the period of the trial.

To provide a reference for the MSW-derived composts, 4 replicates of both a commercially available peat-based compost (John Innes Multipurpose Compost, Humax Ltd, Gretna) and a commercially available green waste compost (Vitalizer®, Vital Earth Ltd, Ashbourne, Derbyshire) were also planted as described above. It should be noted that the Vitalizer® compost is produced predominantly from green waste using a similar in-vessel composting system to

that used to produce the MSW-based composts. Other additives are present in the Vitalizer® compost, however, these are subject to a non-disclosure agreement.

Following completion of composting, a set of replicates as detailed above were sown and then at further 3 wk intervals during the maturation period, new sets of replicates were planted to evaluate the maturity of the materials and how this had changed during the previous 3 weeks. In total, this was repeated 5 times and again after a further 15 weeks to evaluate the changes undergone after a lengthier period.

Harvesting of the plants took place after 8 wk of growth by removing all the shoot material and drying at 80°C for 72 h. pH and EC of the composts were measured at harvesting and compared to the initial values. pH and EC were determined in a 1:1 (v/v) slurry, after mixing for 1 h, with either a Hanna Instruments Model 209 pH meter or a Jenway 4010 EC meter.

The yields of the MSW-based materials were compared to those of the commercial peat-based compost by way of calculating the yield of the former as a percentage of the latter. Direct comparison of the results from each sowing date could then be undertaken as both the MSW and commercial composts had both been grown under similar conditions. It was hypothesized that as the MSW-based materials matured, yields of both *L. perenne* and *T. repens* would increase as the materials became less phytotoxic and concentrations of useful nutrients increased.

4.3.3. Plant growth trials with compost and soil mixtures

Traditionally, composts are mixed with soils in an agricultural setting, however, the dose rate can vary widely (e.g. 1 to 100 t ha⁻¹). Consequently, for the trial involving the mixtures of soils and composts, the two MSW-based composts were added to two soils in differing ratios (50:50 and 75:25 v/v). The two soils used were both Brown Earths but of contrasting texture (clay loam versus a sandy brown earth) and were collected from the Henfaes Research Centre near Abergwyngregyn, Gwynedd, UK (53°14' N, 4°01' W). The mean annual air temperature is 11°C and the annual rainfall is 1250 mm and the site is classed as being a temperate oceanic agricultural grassland. The clay-loam textured 'Brown Earth Soil 1' was derived from glacial till of Ordovician origin while the sand-

textured ‘Brown Earth Soil 2’ was derived from the collapse of a wind-blown sand dune system. Both soils are classified as Eutric cambisols in the FAO Soil Classification System. These soils represent the dominant soils used for high intensity sheep grazing and cereal production (e.g. fodder maize, barley) in North Wales. The grasslands are typically reseeded with a mixture of *Lolium perenne* and *Trifolium repens* on a 5 to 10 yr cycle at which point it would be possible to incorporate composts. The properties of these soils are summarised in the following table:-

| | Brown Earth Soil 1 | Brown Earth Soil 2 |
|---|--------------------|--------------------|
| Texture | Clay loam | Sand |
| EC _{1:1} , $\mu\text{S cm}^{-1}$ | 80 \pm 4 | 76 \pm 19 |
| pH (1:1, H ₂ O) | 6.06 \pm 0.03 | 5.31 \pm 0.1 |
| CaCO ₃ , g kg ⁻¹ | 0.11 \pm 0.02 | <0.01 |
| Water holding capacity, g kg ⁻¹ | 520 \pm 20 | 251 \pm 10 |
| Moisture content, g kg ⁻¹ | 160 \pm 10 | 62 \pm 4 |
| Organic C, g kg ⁻¹ | 2.1 \pm 0.1 | 2.7 \pm 0.1 |
| Total N, g kg ⁻¹ | 0.16 \pm 0.01 | 0.20 \pm 0.01 |
| C-to-N ratio | 13.3 \pm 0.6 | 13.4 \pm 0.3 |
| Soil solution NO ₃ ⁻ , mg N l ⁻¹ | 13.7 \pm 1.3 | 2.7 \pm 0.1 |
| Soil solution NH ₄ ⁺ , mg N l ⁻¹ | 1.4 \pm 0.1 | 28.8 \pm 14.2 |
| <u>Exchangeable cations</u> | | |
| Na, mg kg ⁻¹ | 29 \pm 3 | 17 \pm 1 |
| K, mg kg ⁻¹ | 116 \pm 18 | 5 \pm 1 |
| Ca, mg kg ⁻¹ | 1595 \pm 217 | 48 \pm 1 |
| Mg, mg kg ⁻¹ | 89 \pm 19 | 35 \pm 5 |
| Al, mg kg ⁻¹ | 22 \pm 2 | ND |
| Extractable P, mg kg ⁻¹ | 9.9 \pm 0.3 | 1.8 \pm 0.2 |
| Root biomass, g m ⁻³ | 245 \pm 6 | 337 \pm 101 |
| Soil respiration, g CO ₂ m ⁻² h ⁻¹ | 0.60 \pm 0.02 | ND |

All values represent means \pm SEM (*n* = 3).

ND indicates not determined.

Table 4.1: Chemical and physical characteristics of the two soils used in this study.

The purpose of this trial was to test the effect of mixing MSW-based compost with different soils to evaluate their impact on the yields of *Lolium perenne* and *Trifolium repens*. Four replicates of each species in each of the soil-compost mixtures were sown as described previously, giving a total of 80 pots (Table 4.2). As with the previous trial, harvesting took place after 8 wk growth and the dry weight yield of each replicate was calculated.

| Soil | Compost type | Mixing ratio (Soil:Compost) (v/v) | Replicates per species |
|--------------------|-----------------|--------------------------------------|---------------------------|
| Brown earth soil 1 | None | 100-0 | 4 |
| | 50-50 | 50-50 | 4 |
| | 50-50 | 75-25 | 4 |
| | 70-30 | 50-50 | 4 |
| | 70-30 | 75-25 | 4 |
| Brown earth soil 2 | None | 100-0 | 4 |
| | 50-50 | 50-50 | 4 |
| | 50-50 | 75-25 | 4 |
| | 70-30 | 50-50 | 4 |
| | 70-30 | 75-25 | 4 |

Table 4.2: Combinations of soils and composts used for growth trial.

4.3.4. Statistical analysis

Means and standard errors were computed using Microsoft Excel 2003 (Microsoft Corp., Redmond, WA). All data were inputted into SPSS v12.0 (SPSS Inc., Chicago, IL) and an ANOVA performed with Tukey pair-wise comparison with the *p* value threshold set to *p* < 0.05 for significance. All graphical figures were produced in SigmaPlot v8.0 (Systat Software Inc., San Jose, CA). All yields are presented on a dry weight basis

4.4. Results

4.4.1. Maturity based growth trial: *Lolium perenne*

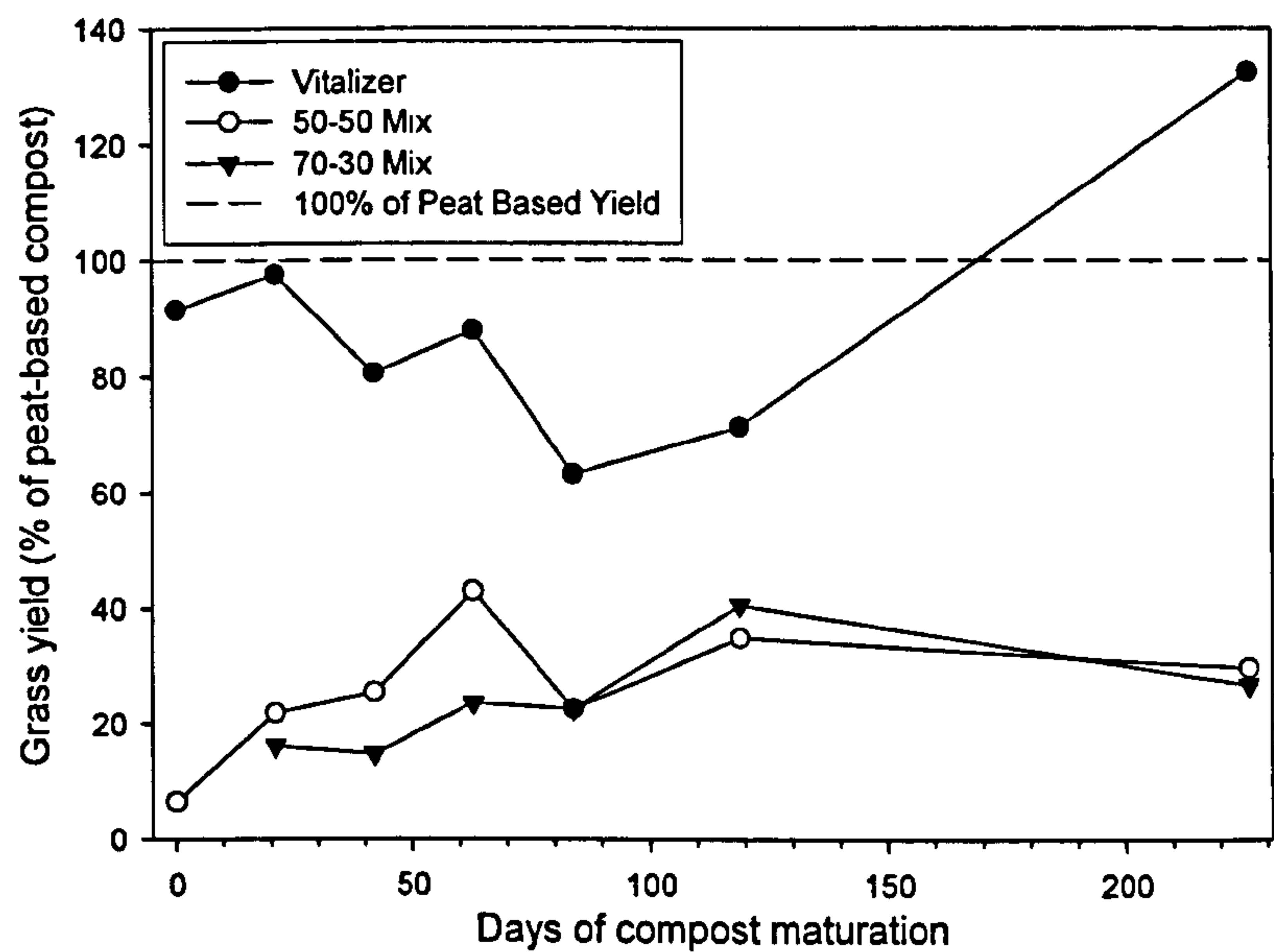


Figure 4.1: Dry weight yields of *Lolium perenne* planted in three different composts (green waste compost, a composted mixture of 50% MSW Fines and 50% Green Waste and a 70% and 30% mixture of MSW Fines and Green Waste) as a percentage of yields from a commercial peat-based compost.

Following harvesting, the change in the above-ground grass yield was calculated and directly compared with the yields from the commercially available composts. When planted in the two commercial composts (peat-based and non-peat-based), *Lolium perenne* yields appeared to decrease over the first 6 wk period (Fig. 4.1); this result is most likely to have been caused by changes in the greenhouse conditions rather than a change in the compost. However, it is possible that the compost sacks used here were not homogeneous and the materials used in each trial contained differing levels of nutrients. Subsequently, yields rose again towards the end of the trial. In contrast, in the 50/50 MSW-derived compost, *Lolium perenne* yields gradually rose during the 4.5 month maturation period. Growth in the 70/30 MSW-derived compost was similar to that of the 50/50 compost.

Statistical analysis of the growth data indicated significant differences in yields when grown in the different compost types for the duration of the trial ($p <$

0.05). Specifically, there were significant differences between grass yields at the start and end of the trial for the MSW-waste derived composts ($p < 0.05$). However, there were no differences in growth response between either the 50/50 or 70/30 MSW-waste derived compost ($p > 0.05$). Significant differences were identified between the two commercial composts and both MSW-based composts ($p < 0.05$ in both cases) with the commercial composts having higher yields. The commercial peat-based compost was used to provide an indication of maximal growth potential under the glasshouse growth conditions. Comparison of the 50/50 MSW-derived compost results showed an overall increase in productivity with time particularly during the first 2 months of the trial (Fig. 4.1). No statistical differences were observed between the two MSW-derived composts ($p > 0.05$), however, differences did exist when both were compared to the peat-based compost ($p < 0.001$).

4.4.2. Maturity based growth trial: *Trifolium repens*

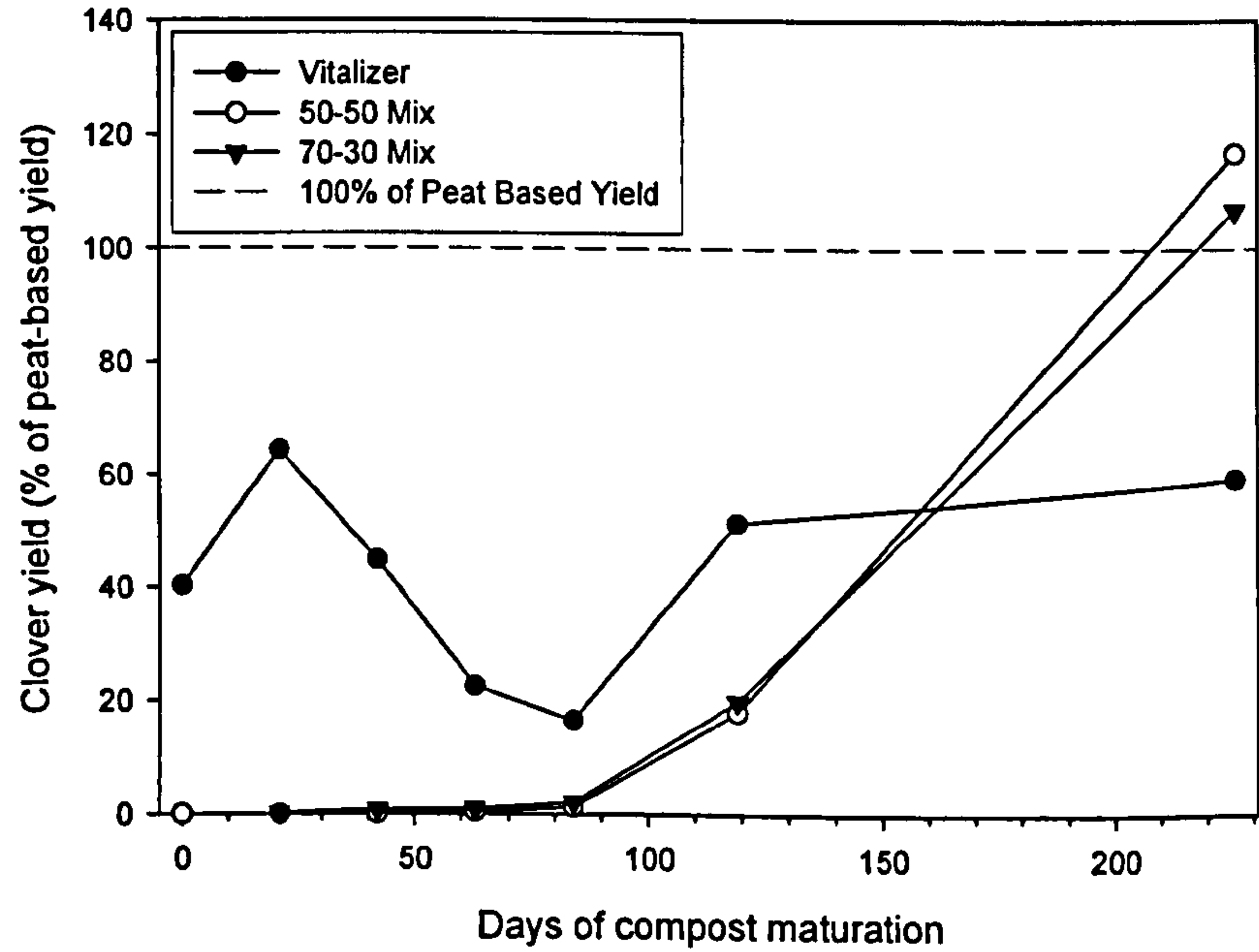


Figure 4.2: Dry weight yields of *Trifolium repens* planted in three different composts (green waste compost, a composted mixture of 50% MSW Fines and 50% Green Waste and a 70% and 30% mixture of MSW Fines and Green Waste) as a percentage of yields from a commercial peat-based compost.

The peat-based compost failed to provide consistent results during the trial period, with two distinct growth response periods observed (data not presented). Prior to 12 wk of maturation, clover yields remained relatively consistent, however, after this date yields were seen to decrease. As the peat-based compost was judged to be homogeneous, it is likely that a change in environmental factors such as increased daylight hours or amount of watering undertaken could be responsible for this.

An overall decrease in yields was recorded for *Trifolium repens* sown in the commercial green waste compost. It is worth noting that those plants sown after 9 wk (63 d) of maturation had noticeable chlorosis of the leaves and were subject to stunted growth compared to previous plantings. It is suggested that the compost batches used for the later plantings contained phytotoxic substances.

Initially, no clover growth was observed in either the 50/50 or 70/30 MSW-derived composts, however, after around 10 wk of maturation, some growth became apparent. After 6 months of maturation, yields were comparable to, if not greater than, both of the commercial composts evaluated here (see Fig 4.2). Statistical analysis identified significant differences between the two MSW-based composts and the commercial composts at the start of the trial. No significant differences between any of the composts were found in the last 6 months of maturation.

Again, the commercial peat-based compost was used to provide an indication of maximal growth potential under the glasshouse growth conditions. Overall, the commercial Vitalizer® compost performed consistently poorly in comparison to the commercial peat-based compost (Fig. 4.2). Evaluation of the 50/50 and 70/30 MSW composts with the peat-based compost showed that 3.5 months was pivotal with little or no growth before this time.

4.4.3 Changes in pH and Electrical conductivity (EC)

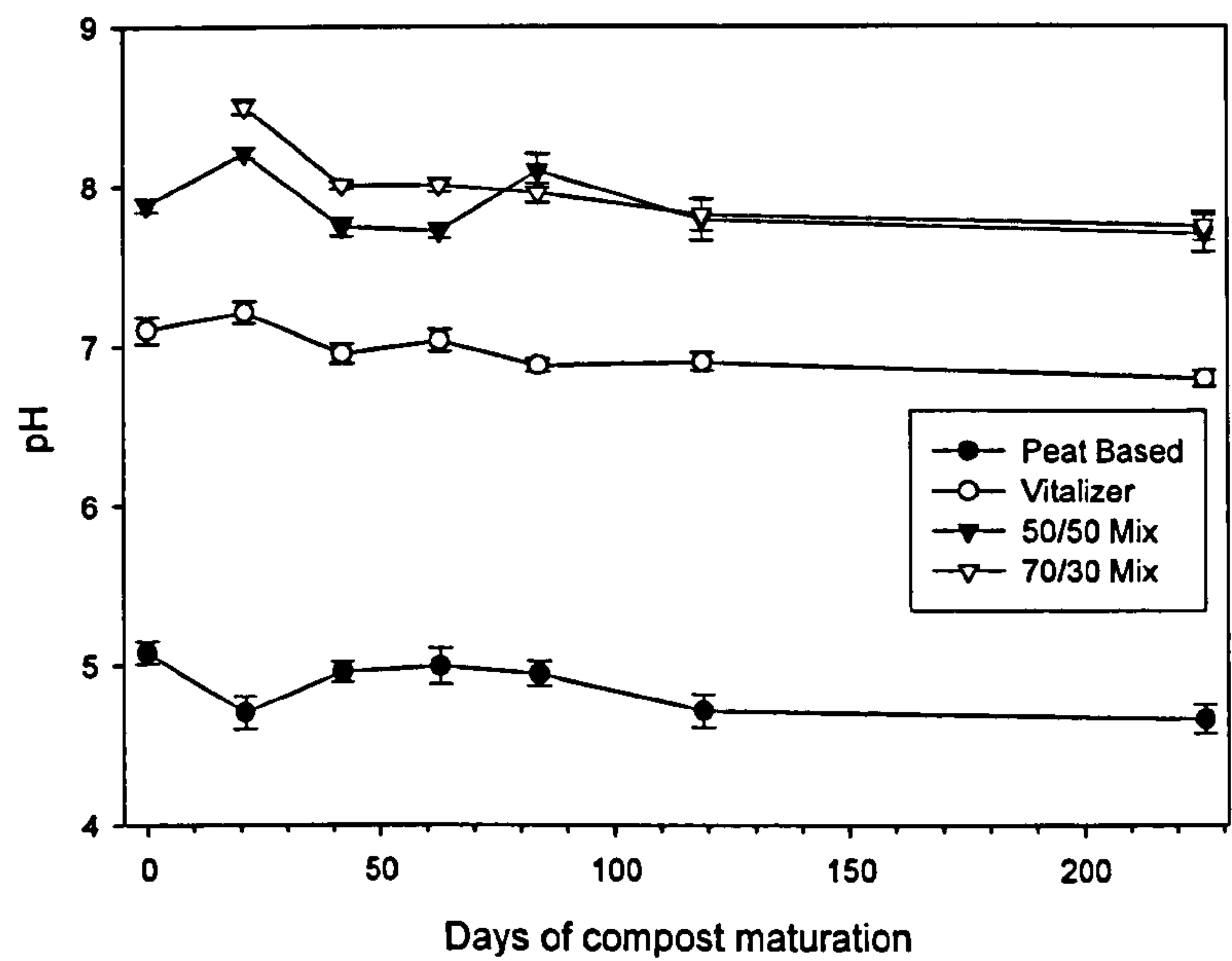


Figure 4.3: Changes in pH of the four composts of different maturities following different length growing periods of *Lolium perenne*.

In the *Lolium perenne* trial, the two MSW-based composts possessed significantly higher pH values than both the peat-based and the Vitalizer® composts ($p > 0.05$). The Vitalizer® compost had a pH consistently around 7.0, whilst the peat-based compost had a much lower pH (around 5.0); this was significantly lower than the Vitalizer® for the duration of the trial. All composts showed slight decreases in pH over the trial, but none of these trends were statistically significant.

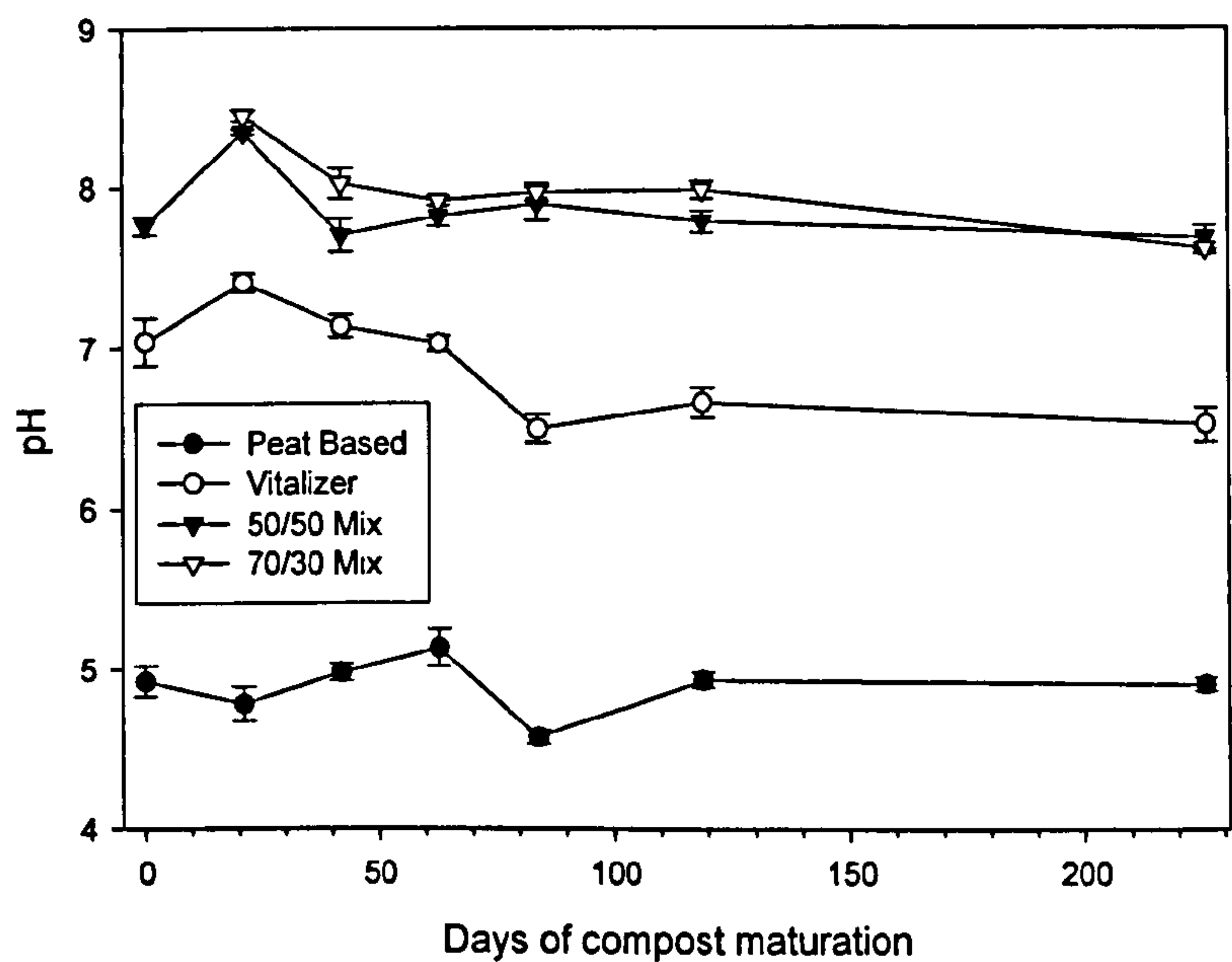


Figure 4.4: Changes in pH of the four composts of different maturities following different length growing periods of *Trifolium repens*.

As with the *L. perenne* trial, the two MSW-based composts in the clover trials possessed significantly higher pH values than the Peat-based and Vitalizer® materials ($p > 0.05$); again the Vitalizer® also had a significantly higher pH than the peat-based compost. Once more, the pH of each compost changed very little for the whole trial, with the exception of the 70/30 MSW compost, where pH dropped from 8.4 ± 0.03 to 7.7 ± 0.06 .

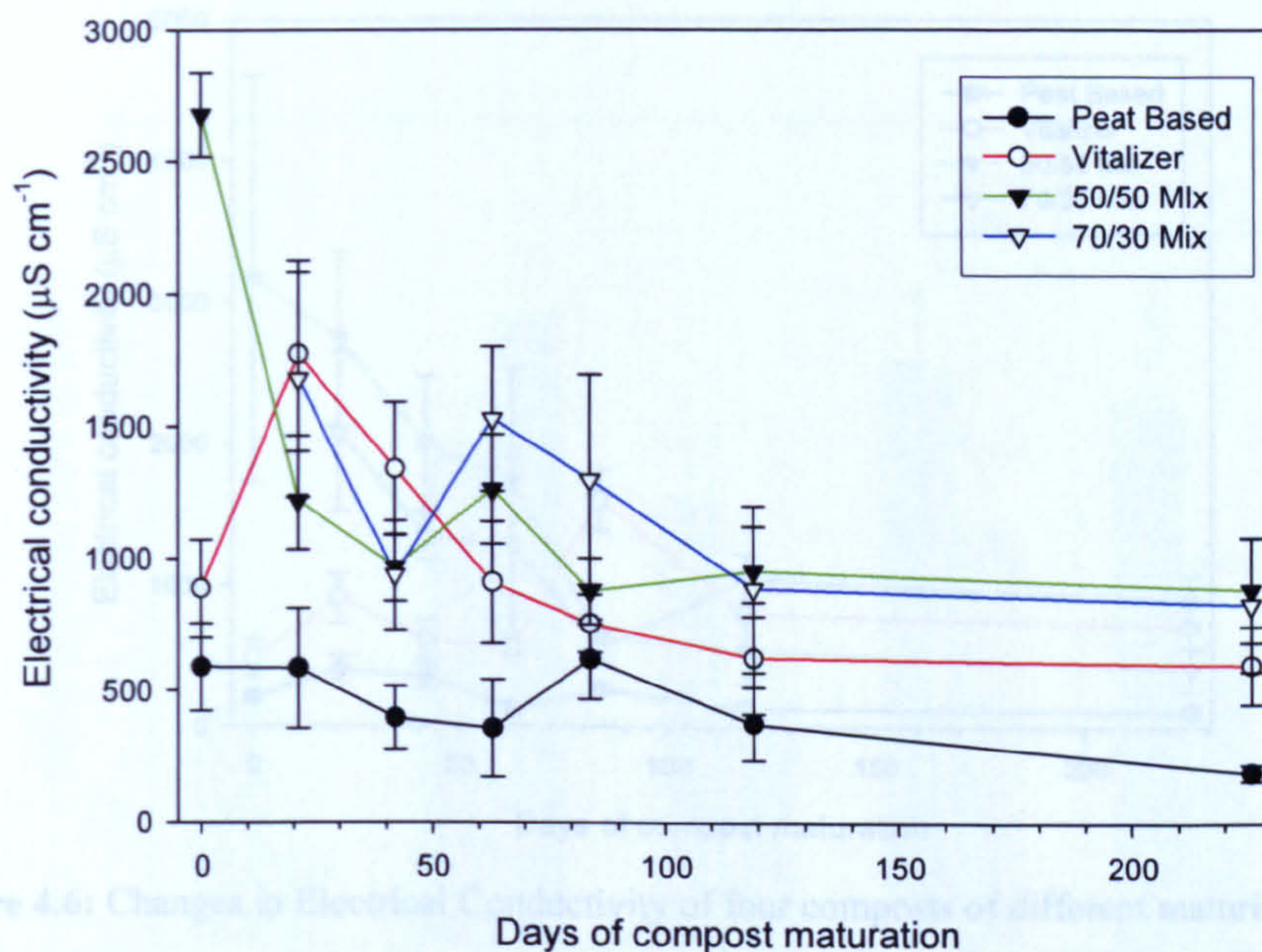


Figure 4.5: Changes in Electrical Conductivity of the four composts of different maturities following different length growing periods of *Lolium perenne*.

The EC of all four composts decreased over time, with the 50/50 MSW compost showing the largest drop; the peat based compost, by contrast, only changed by a relatively small amount. The peat-based compost had significantly lower EC than the other three composts ($p < 0.05$).

4.4.4 Soil and compost mixture based trial: *Lolium perenne*

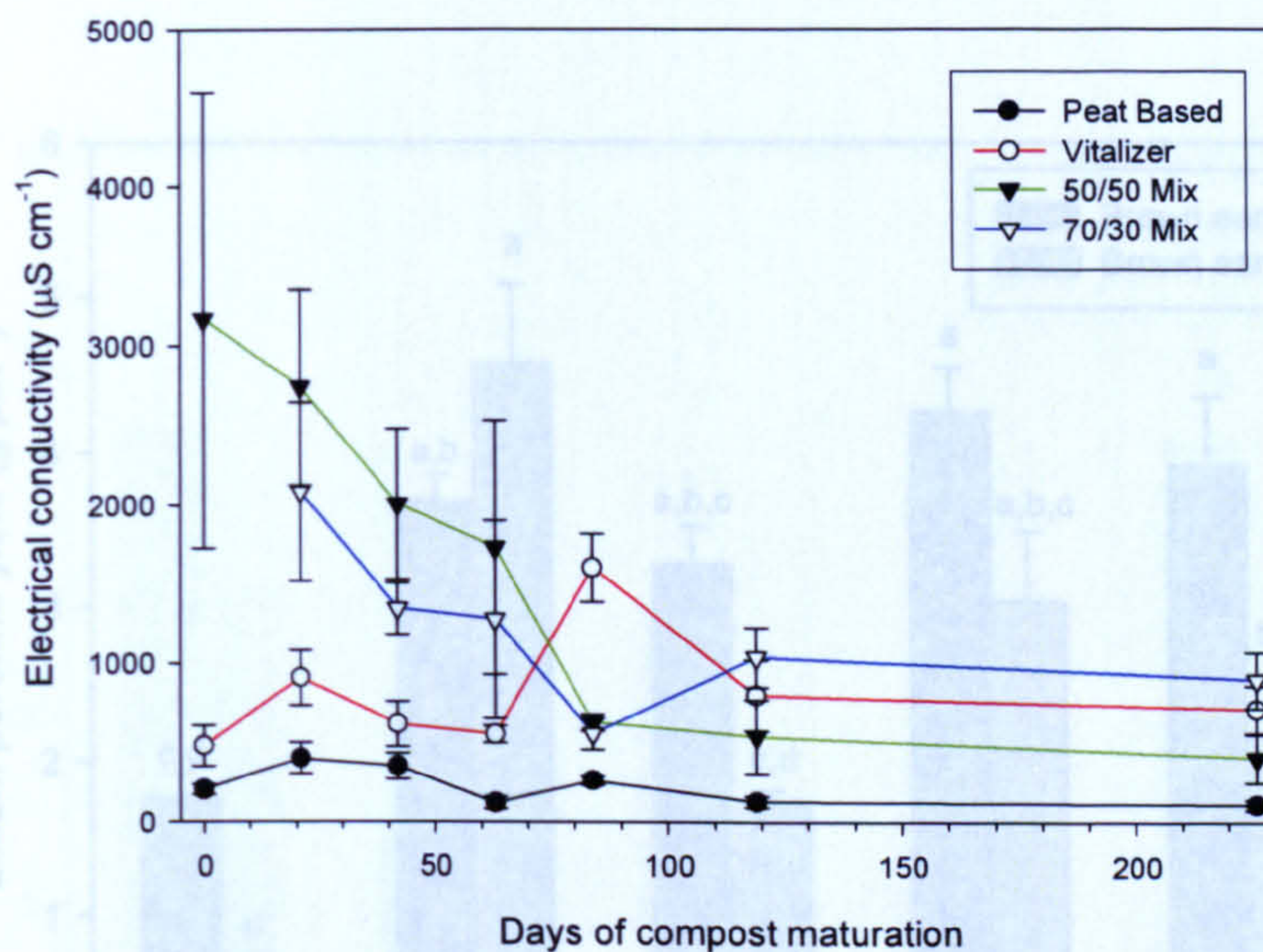


Figure 4.6: Changes in Electrical Conductivity of four composts of different maturities following different length growing periods of *Trifolium repens*.

As with the *L. perenne* trial, the two MSW-based composts were recorded as having the highest EC readings for the majority of the trial, with the exception that towards the final weeks the Vitalizer® material has the higher reading. Once again, the peat-based compost had significantly lower EC than the three other composts ($p < 0.05$). No other significant differences were identified.

In Brown Earth Soil 1, the addition of MSW-based composts had a significant ($p < 0.05$) increase in grass yield in all cases with the exception of the 75/25 mix of soil and 50/50 compost. For Brown Earth Soil 2, adding composts in a 50/50 (v/v) mixture with soil resulted in a significant increase in yields ($p < 0.05$). A soil type response was also observed with a more consistent positive growth response in Brown Earth Soil 1 in comparison to Brown Earth Soil 2. From these results, it can be concluded that adding compost to soil in a 50/50 (v/v) mixture appears to have a greater benefit than a 75/25 (v/v) mixture ($p < 0.01$).

4.4.4 Soil and compost mixture based trial: *Lolium perenne*

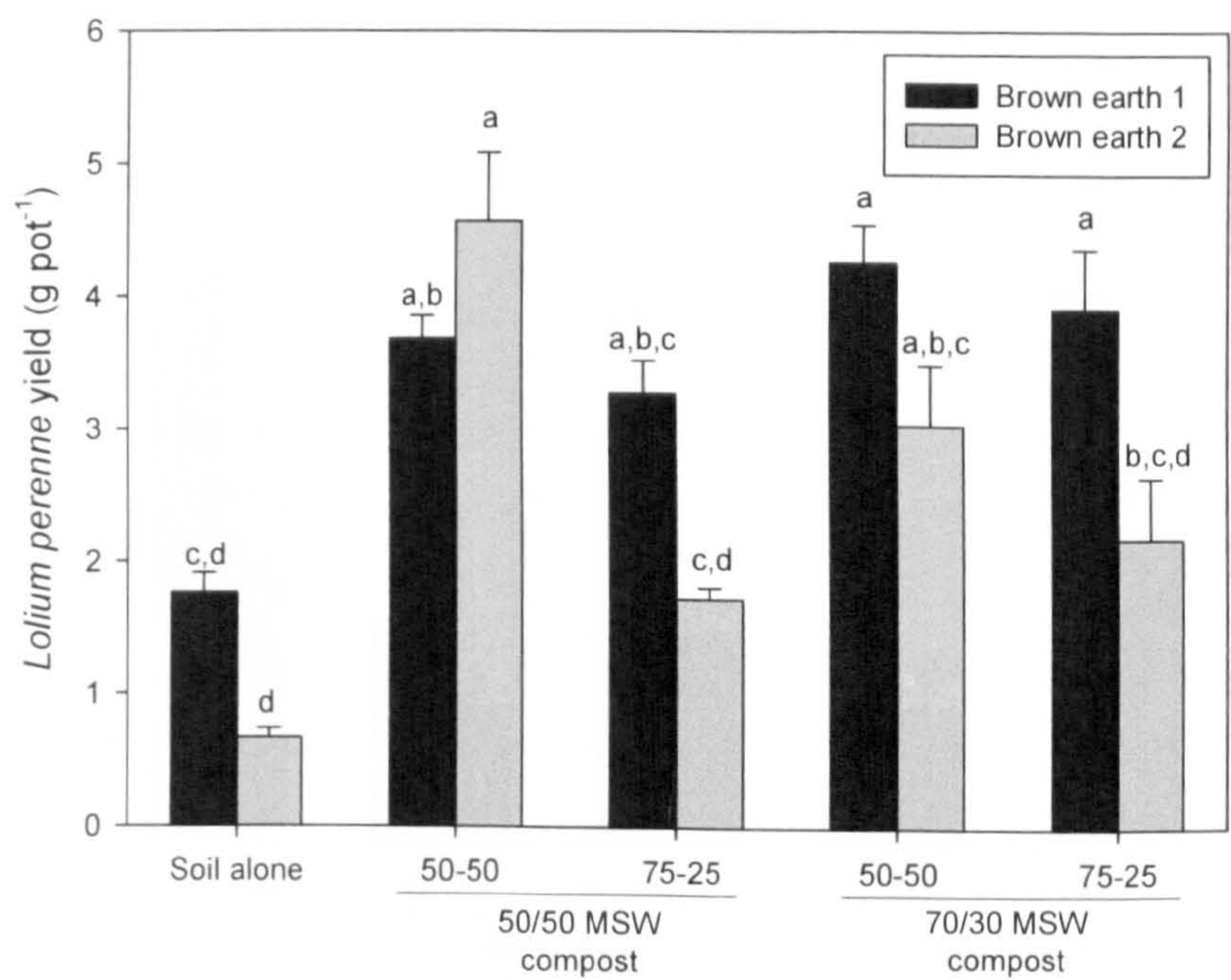


Figure 4.7: Dry weight yields of *Lolium perenne* planted in different mixes of a Brown Earth Soil and composted mixtures of MSW Fines and Green Waste and mixtures of a Sandy Brown Earth Soil and composted mixtures of MSW Fines and Green Waste. Values represent means \pm SEM. Different letters indicate statistical differences ($p < 0.05$).

In Brown Earth Soil 1, the addition of MSW-based composts had a significant ($p < 0.05$) increase in grass yield in all cases with the exception of the 75/25 mix of soil and 50/50 compost. For Brown Earth Soil 2, adding composts in a 50/50 (v/v) mixture with soil resulted in a significant increase in yields ($p < 0.05$). A soil type response was also observed with a more consistent positive growth response in Brown Earth Soil 1 in comparison to Brown Earth Soil 2. From these results, it can be concluded that adding compost to soil in a 50/50 (v/v) mixture appears to have a greater benefit than a 75/25 (v/v) mixture ($p < 0.01$).

4.4.5. Soil and compost mixture based trial: *Trifolium repens*

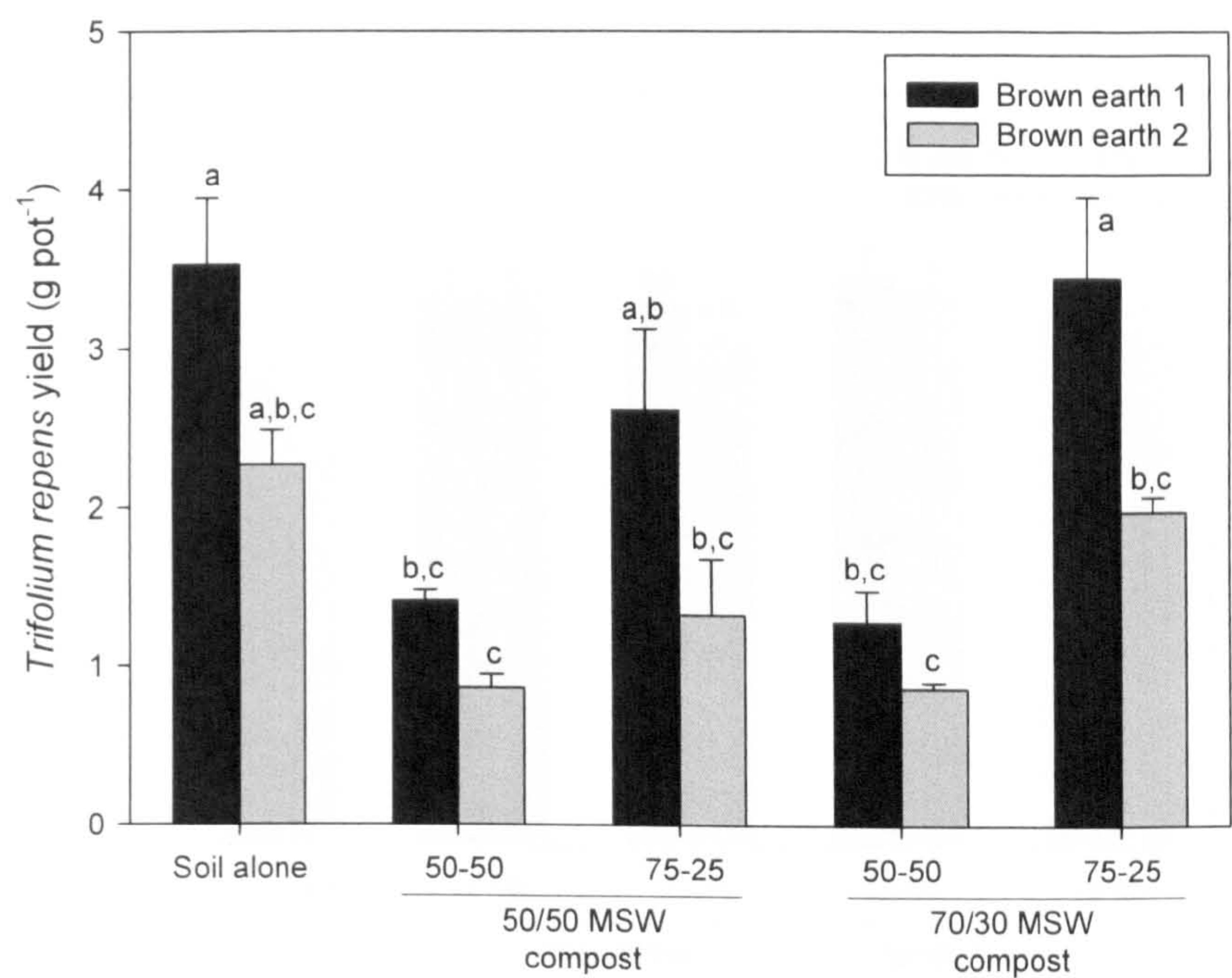


Figure 4.8: Dry weight yields of *Trifolium repens* planted in different mixes of a Brown Earth Soil and composted mixtures of MSW Fines and Green Waste and mixtures of a Sandy Brown Earth Soil and composted mixtures of MSW Fines and Green Waste. Values represent means \pm SEM. Different letters indicate statistical differences ($p < 0.05$).

When the MSW-based composts were added to the Brown Earth 1 soil in a 50/50 (v/v) soil/compost mixture, *Trifolium repens* yields decreased significantly ($p < 0.05$). Using the Brown Earth 2 soil, all treatments showed a reduced yield. However, none of the four treatments showed significantly lower *Trifolium repens* growth when compared to the soil alone ($p > 0.05$). *T. repens* yields were consistently, and significantly, lower when Brown Earth 2 was used compared to Brown Earth 1 ($p < 0.05$).

4.4.6 Changes in pH and electrical conductivity (EC)

4.4.6.1 pH

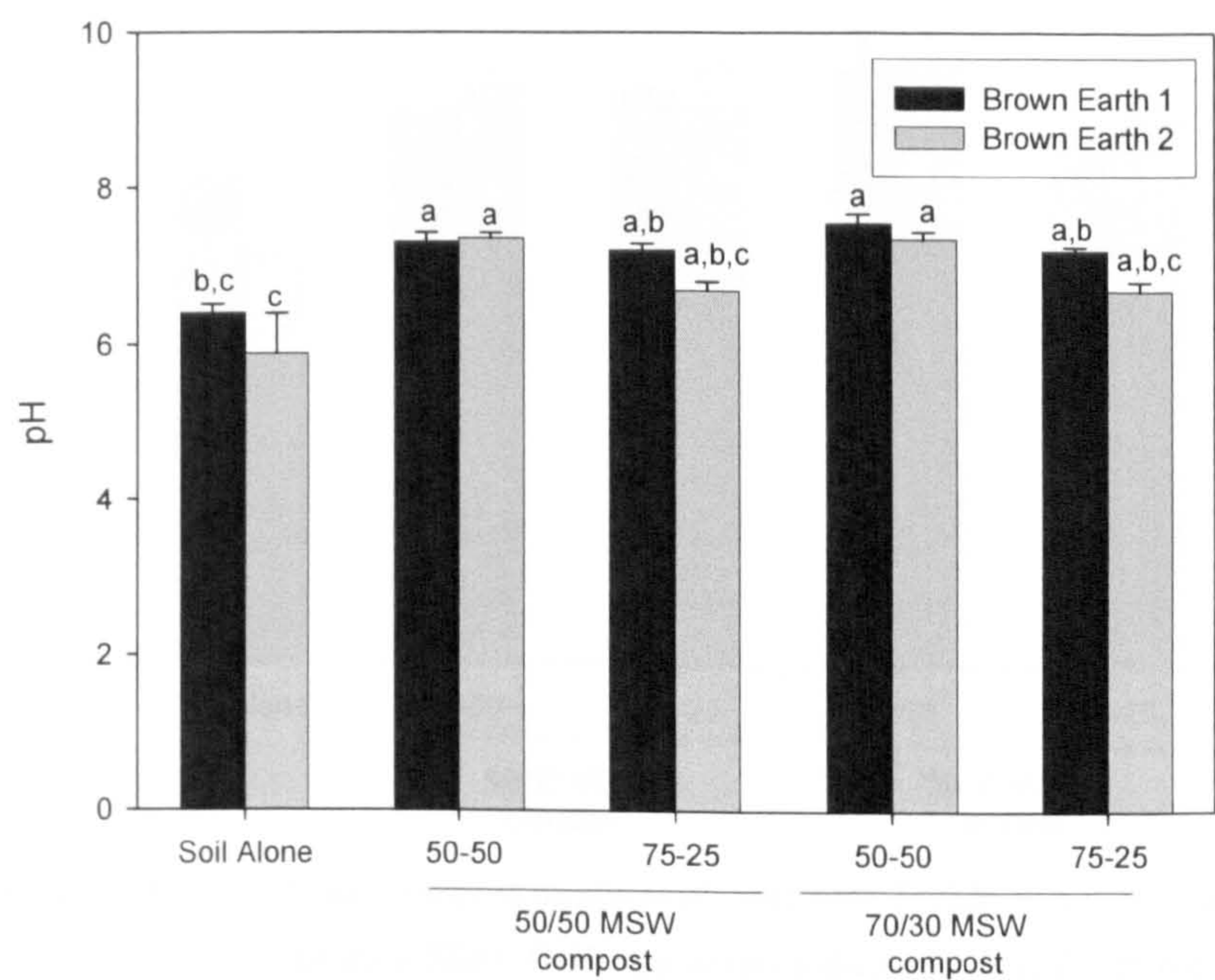


Figure 4.9: pH values of soil/compost mixtures planted with *Lolium perenne*. Values represent means \pm SEM. Different letters indicate statistical differences ($p < 0.05$).

The above graph shows that the addition of MSW-based composts, in the majority of ratios, to the Brown Earth 1 soil significantly increases pH ($p < 0.05$); the only exception being the 70/30 Compost added in a ratio of 75/25 (v/v) soil/compost ratio. With the Brown Earth 2 soil, only adding compost in a 50/50 (v/v) soil/compost ratio caused a significant increase in pH ($p < 0.05$).

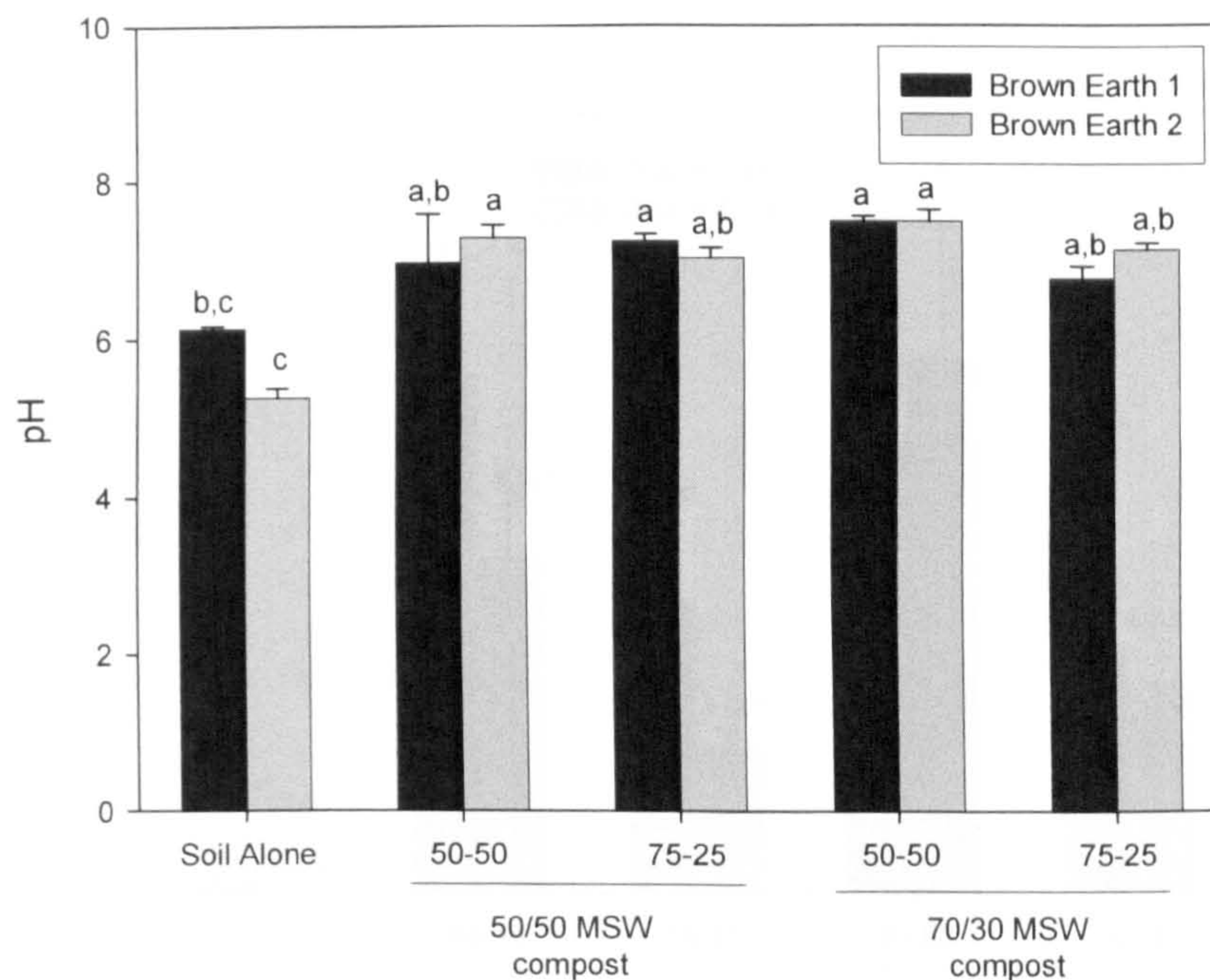


Figure 4.10: pH values of soil/compost mixtures planted with *Trifolium repens*. Values represent means \pm SEM. Different letters indicate statistical differences ($p < 0.05$).

No clear pattern was found for the addition of compost to the Brown Earth 1 soil; only the 50/50 compost added in a 75/25 (v/v) ratio and the 70/30 compost added in a 50/50 (v/v) mixture had a pH that was significantly higher than the soil alone ($p < 0.05$). Conversely, all of the treatments using Brown Earth 2 soil showed significantly increased pH compared to the soil alone ($p < 0.05$).

4.4.6.2 Electrical Conductivity

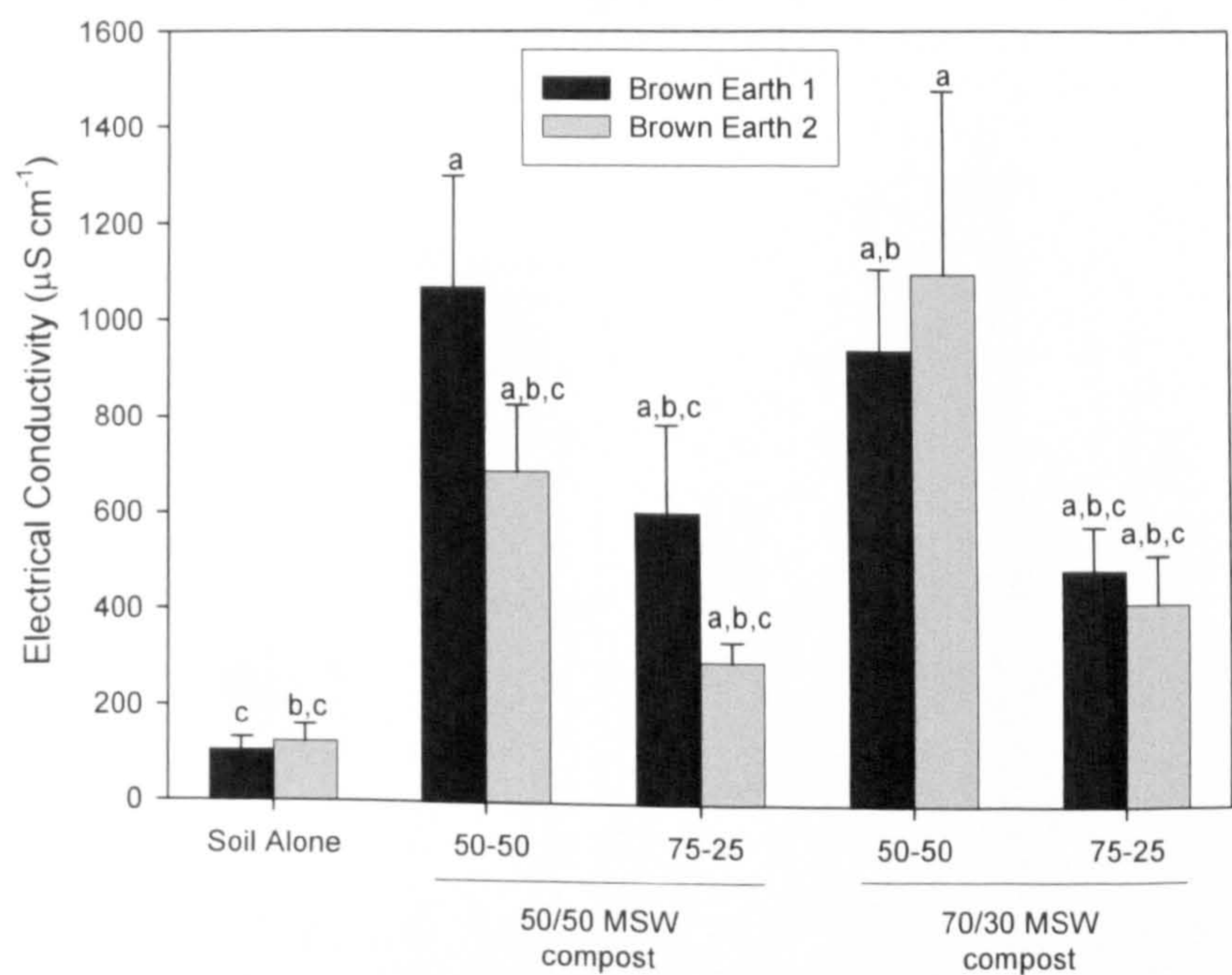


Figure 4.11: Electrical Conductivity values of soil/compost mixtures planted with *Lolium perenne*. Values represent means \pm SEM. Different letters indicate statistical differences ($p < 0.05$).

Addition of MSW-based compost, as with pH, appeared to significantly increase the electrical conductivity of the Brown Earth 1 samples containing 50% compost ($p < 0.05$). For the Brown Earth 2 soil, three out of the four samples increased but not significantly when compared to the original soil ($p > 0.1$). In comparison, the sample containing the 70/30 compost in a 50/50 (v/v) ratio showed significantly increased EC compared to the Brown Earth 2 soil alone ($p < 0.05$).

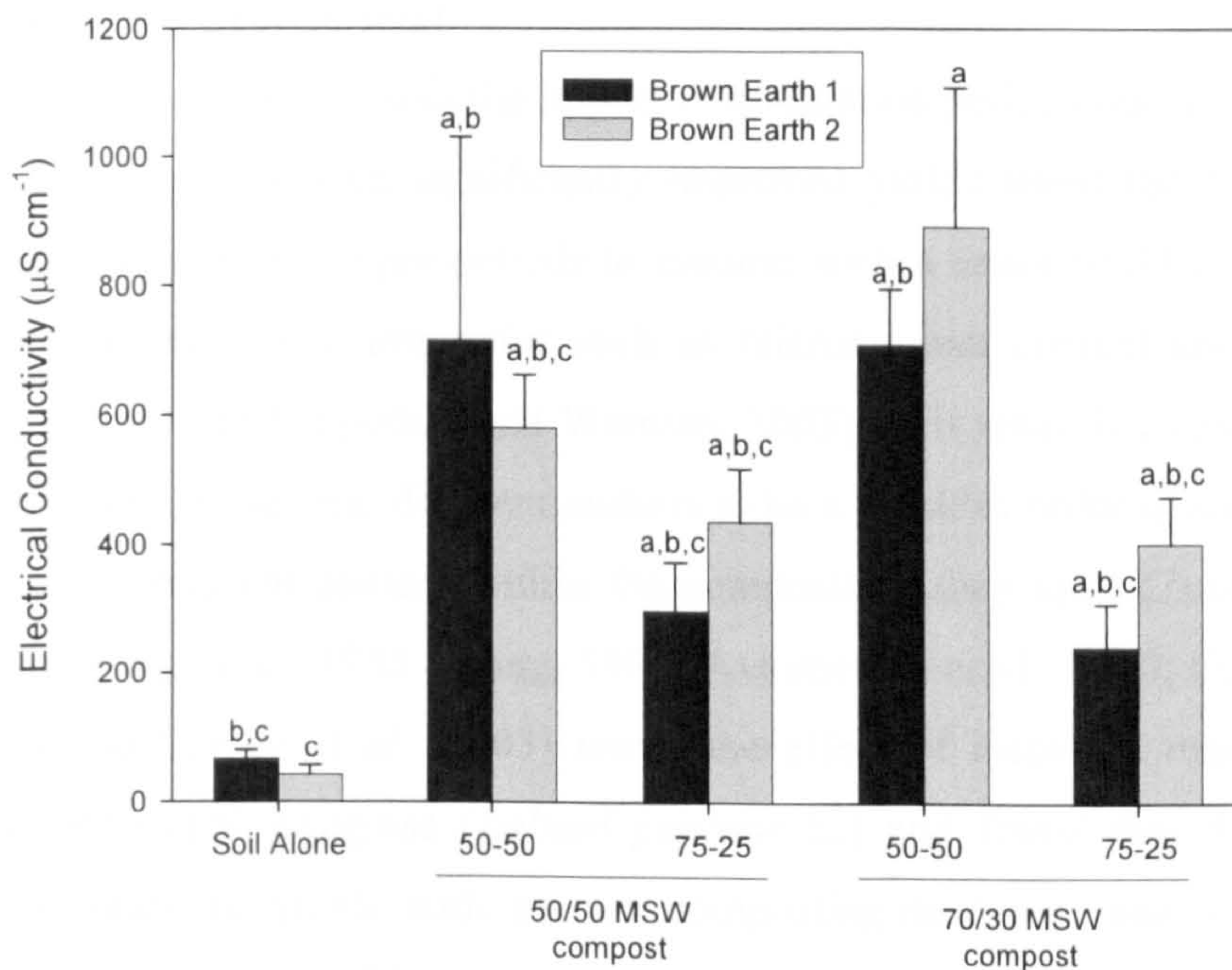


Figure 4.12: Electrical Conductivity values of soil/compost mixtures planted with *Trifolium repens*. Values represent means \pm SEM. Different letters indicate statistical differences ($p < 0.05$).

The addition of compost to the Brown Earth 1 soil increased the electrical conductivity readings in all samples, but none increased significantly ($p < 0.05$ in all cases). For the Brown Earth 2 soil, only two treatments using a 50/50 (v/v) mixture of soil and 70/30 compost had significantly higher conductivity than the unamended soil ($p < 0.05$).

4.5. Discussion

4.5.1. Maturity growth trial

The results of the trial to assess the impact of maturation period clearly showed that both plant species showed significantly improved yields when the MSW-based composts were left for longer periods to mature; such a result could be caused by the improvement of soil properties such as micronutrient content and improved physical properties (Zheljazkov and Warman, 2003). This result is as expected and has been shown by several different authors to be a result of reduced phytotoxicity and increased nutrient content within the composts as they aged (Zucconi *et al.*, 1981 a,b; Wong *et al.*, 1983; Wong, 1985; Avnimelech *et al.*, 1990; García *et al.*, 1991). García-Gómez *et al.* (2003) tested the effect of increased maturity upon yields of Perennial Ryegrass (*Lolium perenne* L.) and found that there was a distinct increase in yields with greater composting/maturing time. In terms of proportional increase, *Trifolium repens* underwent a greater percentage increase in yield than *Lolium perenne* (in this experiment), again this is probably related to lower levels of phytotoxicity within the growth media.

Increasing the length of time that a compost is stored for prior to use has significant implications for the composting industry. As material will continue to be processed, an ever increasing area would be needed to accommodate the maturing compost, resulting in a maximum volume that any one site can process (dependent upon the area available). There are also implications in terms of the amount of management a site may require; i.e. greater input of time and money to produce a suitable end product. However, these costs may be offset by the greater economic value of the compost due to increased management and maturation period. At present, compost prices are very low (£0 to £10 t⁻¹) and average about £5 t⁻¹ if supplied loose and £7 t⁻¹ if bagged. In some cases the compost has a negative value if there is no market and the local authority has to transport it away for disposal. Based on the results presented here, immature MSW-derived compost would be virtually worthless as a soil conditioner and its release on to the marketplace would soon result in dissatisfaction by customers if they were hoping to revegetate land. As consistent market sales are based on branding, dependable

product quality and customer care amongst other things, it would appear that immature MSW-derived compost would fail to realise a niche in the compost market, particularly when there are also large amounts of organic waste-derived composts attempting to occupy the same market niche. In opposition to this view, however, is that longer storage time might result in greater losses of nutrients either in a gaseous form (e.g. NH_3) or by leaching (e.g. K^+ , PO_4^{3-} , NO_3^-) or cause re-infection by human and plant pathogens (Finstein *et al.*, 1987; Sidhu *et al.*, 2001; Christensen *et al.*, 2002). New strategies need to be devised to minimise these risks (e.g. by covering maturing piles of compost or storage indoors) and further work is required in this area.

The results of this study are similar to those shown by many studies (Zucconi *et al.*, 1981 a,b; Wong, 1985; Senesi, 1989) that identify a decline in phytotoxicity as maturation proceeds. This decrease in toxicity is often related to the amount of humification that a material has undergone prior to use (Senesi, 1989). Germination inhibitors (e.g. NH_3 or C_2H_4) could also play a role in the reduced yields of *Lolium perenne* in the immature compost (Wollan *et al.*, 1978).

4.5.2 Effects upon pH and electrical conductivity

The pH of the MSW-derived compost in both plant growth trials was found to decrease slightly during maturation, whilst none of the commercial growth media showed any appreciable change in pH. However, it was evident that the two MSW-based composts had much higher pH values than both the peat-based and green waste-derived (Vitalizer®) composts.

The electrical conductivity (EC) of the *L. perenne* trial was again observed to decrease with time in all four composts. In the 50/50 MSW-based material, this value fell faster in the first 80 d of the trial than in the following period; in contrast, the values of the peat-based compost fell by a much smaller amount.

Again, the *T. repens* trial had a large reduction in EC during the first 80 d for both the MSW composts; during this time, the EC of the Vitalizer® material was found to nearly double, a result of this nature was not expected. The timing of the decrease in EC relates to the increase in *T. repens* yields, which would suggest

that *T. repens* growth is limited by the presence of larger amounts of ionic substances such as soluble nutrients.

4.5.3. Growth trial mixing compost with soils

Mixing soils and composts in differing proportions showed that there were both beneficial and detrimental effects upon the yields of plants, although this appeared to be species related. For example, a compost plus soil mix was found to give an increased yield of *Lolium perenne* when compared to soils alone; this is a similar result to that recorded by García-Gómez *et al.* (2003). However, this effect was reversed when the yields of *Trifolium repens* are considered. These results showed that the addition of refuse-based composts to agricultural soils caused a reduced yield of clover compared to soils alone, something that is attributable to the inability of *Trifolium repens* to cope with the presence of phytotoxic substances such as NH₃, organic acids or plant-available heavy metals (Iglesias-Jiménez and Pérez García, 1989; Serra-Wittling *et al.*, 1995).

4.5.4 Effects upon soil pH and electrical conductivity

Under *L. perenne* growth, the pHs of the soil/compost mixtures were higher than those of the soils alone, yet significant increases were only recorded when the composts were added to the Brown Earth 2 soil. The Brown Earth 2 soil had a lower pH than the Brown Earth 1 soil initially and the difference between the two soils may have exacerbated the final results. Similar results were recorded for *T. repens* in that the pHs increased in all cases when compost was added to either soil. Also, the addition of compost to the Brown Earth 2 soil resulted in a significant rise in pH.

Adding MSW-based compost to either soil caused the EC to increase when either species was grown. However, with *Lolium perenne* grown in a 50/50 mixture of soil/compost, the increases were significant in the Brown Earth 1 soil mixed with 50/50 compost and the Brown Earth 2 soil mixed with 70/30 compost. The ECs recorded for the 75/25 soil/compost mixes were lower than the 50/50 mixtures; this is attributable to the greater amount of compost present in the latter.

4.6. Conclusions

The major conclusions arising from this study are:

- As the composts aged and became more mature, the yields of both *L. perenne* and *T. repens* increased. This clearly demonstrates the need for a long compost maturation period to enhance the economic and environmental quality of any MSW-derived compost.
- *T. repens* yields underwent a greater proportional increase in growth than *L. perenne* during this trial suggesting that some plant species are more sensitive to immature composts than others. Further tests are required to determine the mechanistic basis for these observed species differences.
- The addition of MSW-based composts to soils gave higher yields of *L. perenne* when added to soil compared to soil alone. This demonstrates that agricultural land disposal could be a viable option for ABPR (2005) compliant MSW-derived composts if there is minimal risk from physical contaminants. Alternatively, the composts would be suitable for land restoration and construction schemes (e.g. landfill capping, road embankments, mine restoration etc).
- The addition of MSW-based composts to soils gave lower yields of *T. repens* when added to soil compared to soil alone. This suggests that care is needed about where the composts should be added in the environment. Further tests on a greater number of plant species are required, particularly woody plants.
- Addition of compost to soil in any of the ratios used here gave rise to a change in pH that is dependant upon soil type and not the presence of any vegetation.
- Further tests should be carried out to assess the role of MSW composts in biofuel production (e.g. short rotation coppice, *Miscanthus*) where the presence of physical contaminants is of minimal risk.
- An economic and market analysis study is required to evaluate the cost-benefit of compost maturation.

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**Economic and environmental assessment of the
influence of screen size on the contaminant load of
the biodegradable MSW fraction derived from an
MRF.**

Mr Richard W. Furniss and Prof David L. Jones

Economic and environmental assessment of the influence of screen size on the contaminant load of the biodegradable MSW fraction derived from an Materials Recovery Facility (MRF)

5.1. Abstract

In response to a range of legislative drivers, there has been an increase in the treatment of Municipal Solid Waste (MSW) by Materials Recovery Facilities (MRF). The biodegradable fraction produced in MRFs (MSW Fines) is often contaminated by non-biodegradable components (e.g. glass, metal, plastic) which reduce its market value and compromise its potential end use. The aim of this study was firstly to evaluate two contrasting methods for characterising contamination levels in MSW Fines; the first being a simple, rapid visual-based method and the second being a time-consuming weight-based physical fractionation method. The second aim was to assess whether decreasing the size of the aperture on the trommel screen that separates out the MSW Fines could help reduce contamination and materials recovery in the MRF. Due to the different densities, shape and moisture contents of the different waste fractions (biodegradable, glass, metal, plastic etc) contained in the MSW Fines, each method gave contrasting results. We conclude that the visual area-based method combined with a colour/spectral analysis provided the best assessment for physical contamination whilst the weight-based assessment is only suitable for determining losses of MRF segregation efficiency. Screening the MSW Fines through different trommel screens (10 to 50 mm) failed to reduce contamination but did improve material recovery of glass and metals. Economic analysis indicated that improved diversion from glass and metals from entering the MSW fines far outweighed the benefits in improving the quality of the biodegradable fraction or compost resulting from it.

5.2. Introduction

Municipal Solid Waste (MSW) is produced in high volumes and possesses low intrinsic value and its production is increasingly seen as a major problem of modern society. It contains a large percentage of organic waste alongside some hazardous

items and materials (e.g. household cleaners, discarded syringe needles, bacterial pathogens; Maltbaek, 1999). Due to the potentially hazardous nature of MSW, landfilling became the easiest and cheapest disposal route used in the majority of nations; however, due to the unsustainable nature of landfill, new and effective waste treatment methods capable of wide scale adoption are required. Within the MSW stream there is often a large proportion of material which can be reused, creating a need to separate the fractions of highest economic value for recycling and reuse (Marchettini *et al.*, 2007). Material reclamation facility (MRF) was a name originally created in the USA to describe the variety of sorting facilities developed in an attempt to cope with the vast increase in municipal rubbish from the 1970s onwards (Maltbaek, 1999). Based on agricultural and mining sorting and grading equipment, some of these facilities have evolved into sophisticated mechanised systems requiring elaborate design and considerable investment from the responsible municipality or its contractor. Installing MRFs in a solid waste management system could be a feasible alternative to achieve sustainable development goals in urban areas if current household and kerbside recycling cannot prove successful in the long run (Chang *et al.*, 2005). Ultimately, MRFs however, must be considered a temporary measure to allow some material recovery while the whole nature of material flows in society is readjusted to ensure that nothing of recycling value ends up as waste. MRFs encompass several different technologies to progressively remove useful materials from the waste stream (e.g. using eddy current and magnetic separators, hand picking lines and trommel screens; Lunde, 1995; Schlett *et al.*, 2002). MRFs are now seen as a vital arm in meeting waste management targets in many developed countries (Kang and Schoenung, 2006). The number of MRFs are also likely to increase particularly in European nations where the majority of recyclates are not source segregated (e.g. UK and Portugal; Magrinho *et al.*, 2006).

While MRFs are being deployed in many regions of the UK, they are relatively costly to run and it is clear that significant research is needed to maximise throughput and reduce the cost and environmental footprint of MRF operations (Smith and Linton, 2001; Alexiou and Moore, 2005; Chang *et al.*, 2005).

One key area associated with this is the treatment of the biodegradable fraction of MSW generated in the MRF when treating mixed wastes. Under current legislation (i.e. the 2005 Animal By-Products Regulations) this material must be treated post-separation to eliminate potential pathogens (e.g. TSEs, Salmonella etc). Typically, this is achieved by in-vessel composting. The product resulting from this, however, can contain a range of contaminants that will not be removed by in-vessel composting (e.g. plastics, glass shards etc; Xie *et al.*, 2006) and therefore effective separation techniques are required to remove them prior to post-treatment.

Estimates of the amount of biodegradable material present within MSW in Wales typically ranges from 60 and 65% (Burnley *et al.*, 2007; Welsh Assembly Government, 2002) although figures of 60-75% have also been reported (ENDS, 2001; García *et al.*, 2005). A recent report suggests that 36% of MSW in Wales consists of recyclables, 28% is compostable and that packaging accounts for 17% of the total MSW (Burnley *et al.*, 2007). The actual allocation to different categories, however, is likely to be dependent upon local conditions, consumption habits and seasonality (Abu Qdais *et al.*, 1997). The major cause of the variability of this figure is the heterogeneity of MSW and the number of different materials present within a sample, which can in turn vary greatly in proportion between one sample and another. Optimization of the collection, separation, treatment and reuse of this biodegradable fraction from MSW is imperative if we are to meet national waste diversion targets and promote sustainable societies. However, there are some environmental issues associated with the management of the organic fraction contained within MSW. This may reflect such issues as odours (Sironi *et al.*, 2006), airborne particles released during sample processing in an MRF and their possible effect on human health (e.g. increased incidence of respiratory and gastrointestinal symptoms; Gladding *et al.*, 2003) through to potential contaminants in the biodegradable fraction that may interfere with its subsequent treatment (e.g. heavy metals during composting) or in its end use (e.g. plastic and glass contaminants that prevents its sale or spreading to land; Kanat *et al.*, 2006). Some evidence exists to suggest that trommel screen size may be an effective management tool for optimizing the recovery of waste fractions in MSW (Lau *et al.*, 2005).

The aims of this study were:

- To evaluate the impact of different trommel screen sizes on the recovery of the biodegradable fraction of MSW at an MRF;
- To evaluate the economic and environmental influence of trommel screen size on the diversion of recyclates from the biodegradable fraction;
- To evaluate weight versus area based measurements and colour-based measurements of waste separation and contamination.

5.3. Methods

5.3.1. MSW Fines Sampling

A commercial MRF located at LAS Waste Ltd, Lampeter was used for this study. The MRF processes around 275 tonnes of waste per week (on average), with approximately 40-50 tonnes per week (on average) being recovered as Fines. For the purposes of this project, Fines are defined as “the material which passes through the 50 mm trommel screen out of the MSW stream.” In the present study, the material passing the <50 mm screen is currently used for landfill cover but is suitable for composting (see earlier chapters) whereas the larger material is recovered (metals, papers etc.) and the reject oversize material sent directly to landfill. Adani *et al.* (2004) conducted a study which also used an aperture size of 50 mm to screen out the finer material from the MSW stream with the residual material (>50 mm) transferred to an “Energy from Waste” (EfW) plant for incineration, due to its low moisture content and high calorific value.

The first section of the MRF into which the MSW bags are loaded is a bag shredder which allows the contents of the rubbish sacks to be spilled onto a conveyor belt; from here it is carried to the trommel screen. This consists of a large rotating drum perforated with 50 mm diameter holes. Any material which is smaller than 50 mm in diameter (in any one plane) passes through the trommel screen and onto a conveyor belt from where it is transported to collecting hoppers. This material consists of biodegradable matter along with small fragments and shards of plastic, metal and glass. Any MSW which does not pass through the 50 mm aperture exits the trommel and is conveyed into further sorting sections including a

magnetic metallic separator and a hand picking line. That which is left over at the end of the MRF is known as “Residual MSW” and is transported direct to landfill.

This project is concerned with utilising the MSW Fines and finding a sustainable end use for them. Therefore, it was deemed necessary to classify the material into its constituent fractions, such as organic, metal, glass and plastic. By doing this, it would become clearer as to which materials are most prominent within the MSW Fines and so a suitable process for removal of these inert materials from the biodegradable fraction could be identified. This would also give important information regarding the suitability for different end uses, and any processes by which this could be achieved. For example, if the MSW Fines were found to be high in organic/biodegradable material, composting could be used as a further treatment to reduce the volume of the waste. This is dependant upon the majority of the inert material being removed as it will not compost and has a degrading effect upon the aesthetics of the final “product”. Also, by calculating the proportions of each constituent material, potential increases in recovery rates could be identified.

5.3.2. Methods for Physical Characterisation of MSW Fines

Two methods were used to assess the proportions of each constituent material within the sample of MSW Fines. The first was a visually based method whilst the second was based upon the weights of different fractions within the sample of MSW Fines.

5.3.2.1. Visual (Area) Assessment Method

The Fines were laid onto a large plastic sheet and levelled to a depth of approximately 50 mm and a 1 m² gridded quadrat placed over the waste. The type of material directly underneath each 10 cm intersection of the horizontal and vertical lines was recorded and the relative proportions of each were calculated.

5.3.2.2. Weight-Based Method

This method was designed to assess the weight of each material fraction that would pass through the trommel screen. In addition to the standard 50 mm trommel

screen, a set of screens at 10, 20, 30 and 40 mm were also used to size-grade the MSW to determine whether trommel screen size could be manipulated to better eliminate non-biodegradable contaminants. Briefly, a sample of MSW Fines was initially passed through the 40 mm screen and the material which did not pass through (known as the “>40 mm” fraction) was then divided into its constituent fractions before being dried overnight at 105°C and the relative proportions (by weight) of each fraction calculated. The residual material (that which passed through the 40 mm screen) was then passed through a 30 mm diameter screen. This process was repeated using 20 mm and 10 mm screens, each time the residual being divided into fractions, dried and the proportions by weight being calculated. Material which passed through the 10 mm screen was known as the “<10 mm” fraction and was not screened any further for practical reasons, although the different materials within this sample were segregated and weighed. At each stage of this method, glass and plastic were also divided on the basis of colour.

5.4. Results

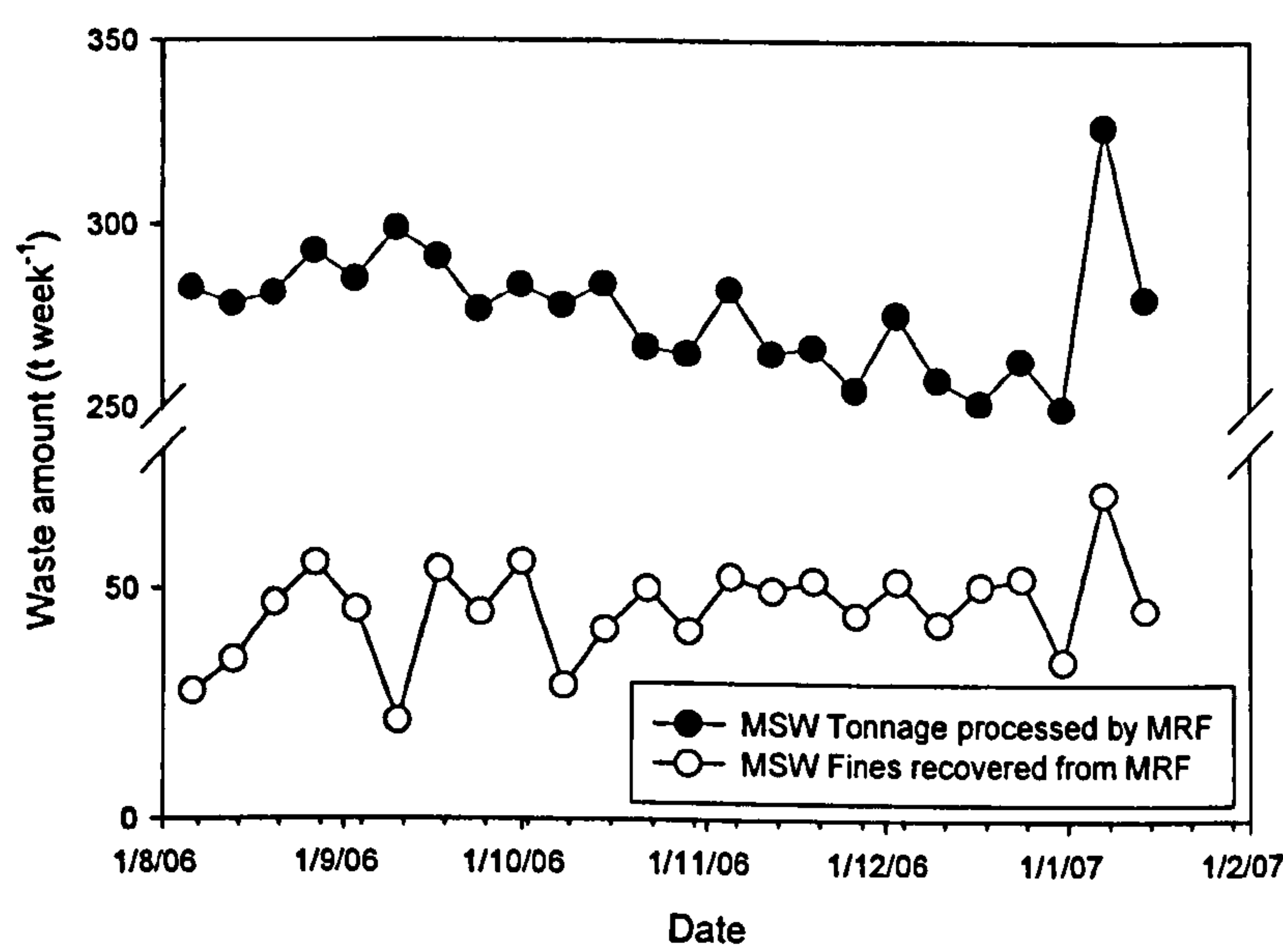


Figure 5.1: Trends in tonnage of material processed by MRF at LAS Waste Ltd and amount recovered as MSW Fines, on a weekly basis.

Figure 5.1 shows the trends in both tonnages of material processed and the tonnages of MSW Fines recovered during the same period. The sharp peak during the week ending 7/1/07 is most likely related to the disposal of materials from the Christmas/New Year holiday period, such as bottles and cans. Tonnage of MSW Fines recovered during this time also rose by around 25-30 tonnes above the normal average. There is a clear relationship between the two sets of data, which indicates that the proportion of the overall MSW stream which is removed as MSW Fines tends to remain reasonably constant over time ($16.5\% \pm 0.7\%$ of the overall MSW processed) ($r^2 = 0.239$; $p = 0.261$). It is also clear that the tonnage processed by the MRF appears to decrease with time ($r^2 = -0.309$; $p = 0.142$), whilst the amount of MSW Fines recovered does not undergo the same change ($r^2 = 0.382$; $p = 0.065$). This could signify that there is a maximum amount of MSW Fines recoverable at any one time, although further data is needed to confirm/deny this.

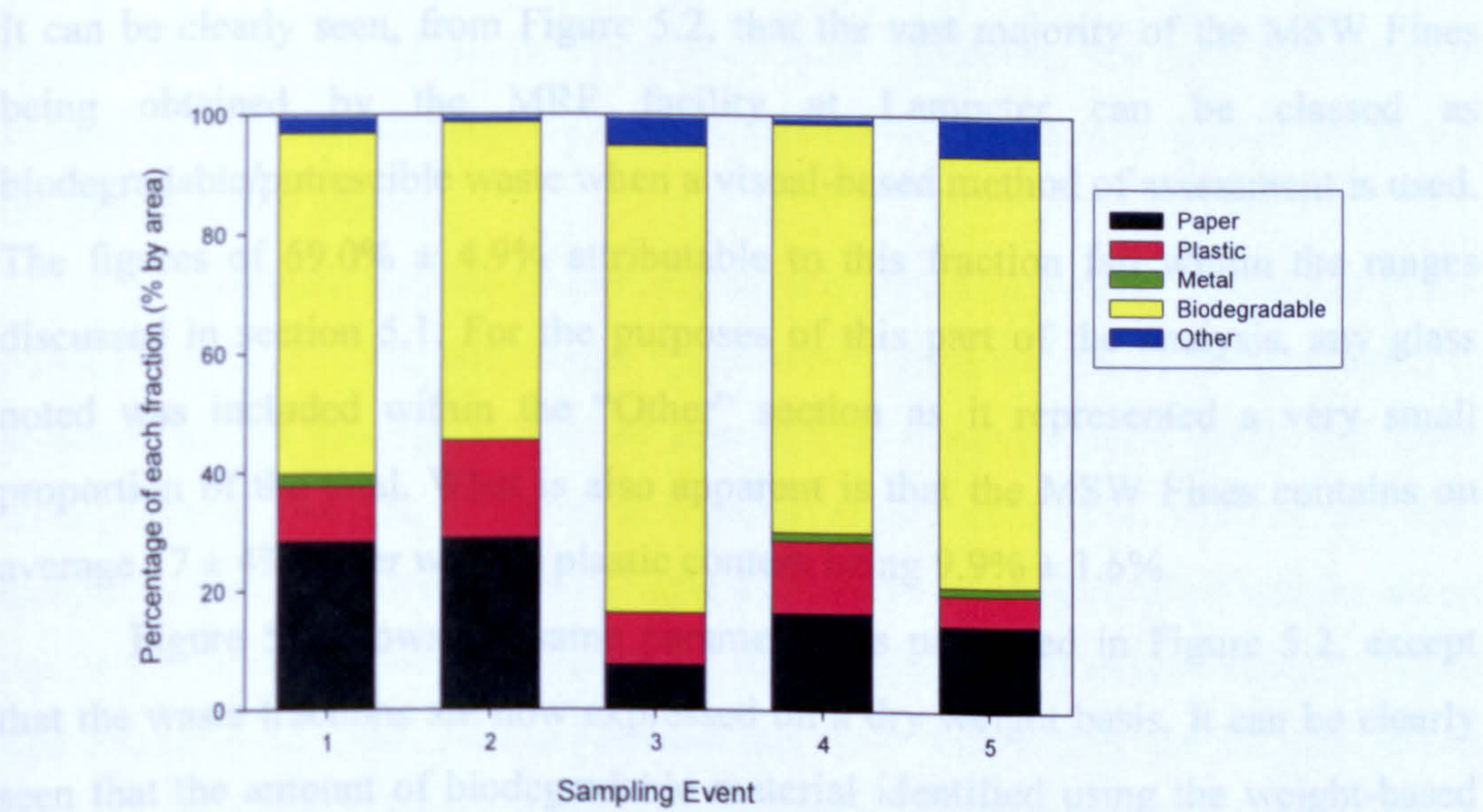


Figure 5.2: Data for the quadrat area-based visual assessment of proportions of different fractions within a sample of MSW Fines sampled on 5 independent occasions.

When the quadrat-based method was used, paper and plastic made up much greater proportions (as discussed earlier). Paper and plastic are lightweight materials and found to have low dry bulk densities (i.e. ca. 0.099 and 0.033 g cm⁻³).

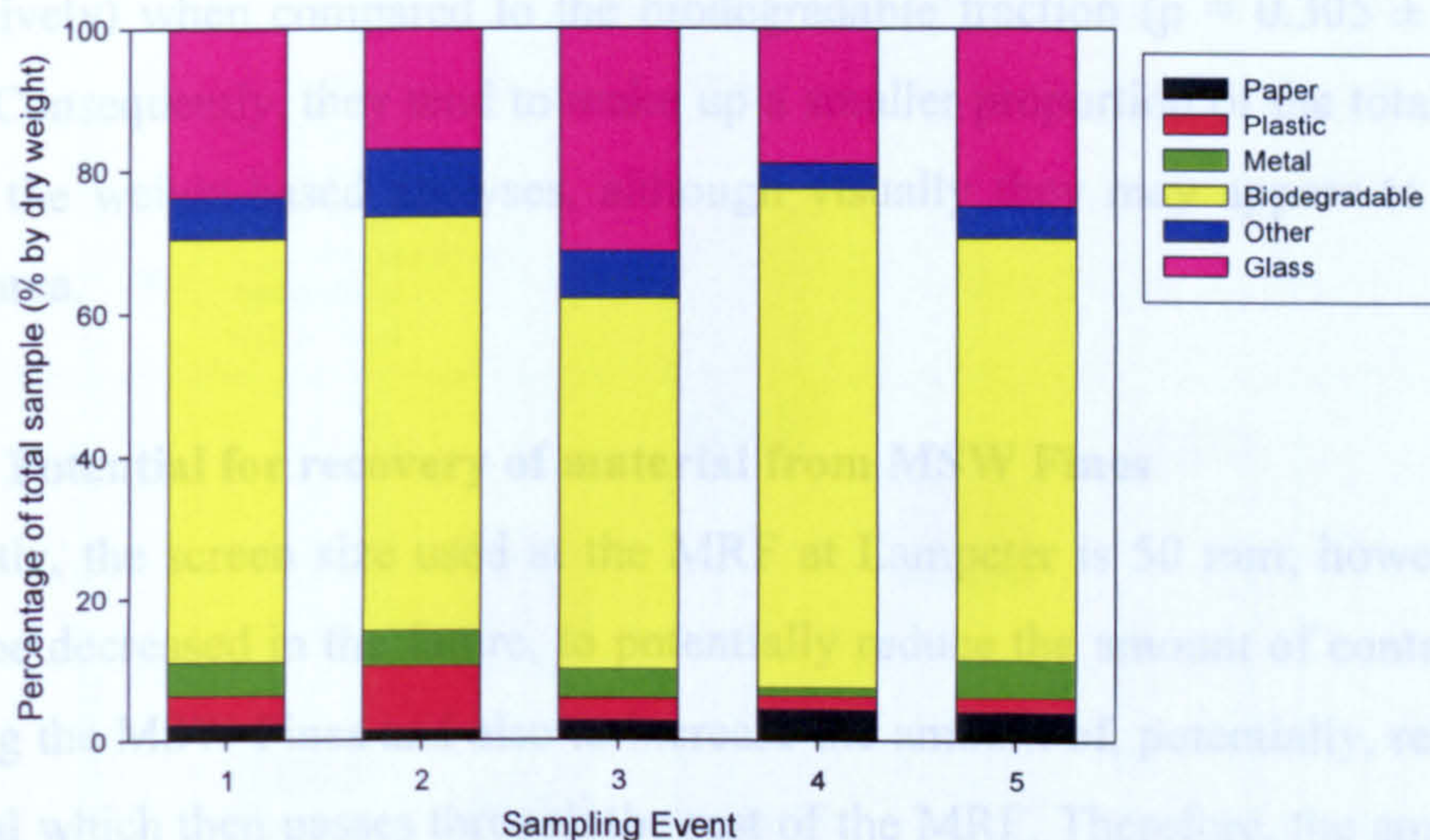


Figure 5.3: Changes in proportions of different fractions in MSW Fines, measured by weight based analysis sampled on 5 independent occasions.

It can be clearly seen, from Figure 5.2, that the vast majority of the MSW Fines being obtained by the MRF facility at Lampeter can be classed as biodegradable/putrescible waste when a visual-based method of assessment is used. The figures of $69.0\% \pm 4.9\%$ attributable to this fraction fall within the ranges discussed in section 5.1. For the purposes of this part of the analysis, any glass noted was included within the “Other” section as it represented a very small proportion of the total. What is also apparent is that the MSW Fines contains on average $17 \pm 4\%$ paper with its plastic content being $9.9\% \pm 1.6\%$.

Figure 5.3 shows the same parameters as presented in Figure 5.2, except that the waste fractions are now expressed on a dry weight basis. It can be clearly seen that the amount of biodegradable material identified using the weight-based method is roughly the same as the visual method, $59.7 \pm 3.9\%$. However, the differences between the two assessment methods are clearly highlighted when the data for both plastic and paper are considered. Data from the weight-based method shows that plastic accounted for $4.0 \pm 1.3\%$ and paper for $4.0\% \pm 0.7\%$ of the total whereas when the quadrat-based method was used, paper and plastic made up much greater proportions (as discussed earlier). Paper and plastic are lightweight materials and found to have low dry bulk densities (ρ ca. 0.099 and 0.033 g cm^{-3}

respectively) when compared to the biodegradable fraction ($\rho = 0.305 \pm 0.058 \text{ g cm}^{-3}$). Consequently, they tend to make up a smaller proportion of the total sample within the weight-based analyses, although visually they may appear to cover a larger area.

5.4.1. Potential for recovery of material from MSW Fines

Currently, the screen size used at the MRF at Lampeter is 50 mm; however, this could be decreased in the future, to potentially reduce the amount of contaminants entering the MSW Fines and also to increase the amount of, potentially, recyclable material which then passes through the rest of the MRF. Therefore, the amounts of recoverable materials have been calculated using figures obtained from the MRF site after passage through a range of screen sizes. Based on the results presented in Figures 5.4 and 5.5, decreasing the trommel screen size from 50 mm significantly reduces the amount of MSW Fines recovered. However, the data presented in Figure 5.4 does indicate that the waste fractions are not equally distributed across the different size grades.

It was found that the 40 mm screen only removed a small percentage of the overall sample, $6 \pm 1\%$, with larger fragments of glass and metal being commonly screened out at this stage. The 30 mm screen removed more glass and some biodegradable material, resulting in a removal rate of $17.5 \pm 3.9\%$ of the total sample weight. Glass removal was highest (in terms of proportion by weight) using the 20 mm screen, where an average of $10.4 \pm 0.8\%$ of the overall sample weight was found to be glass. Further calculations have shown that the 20 mm screen removes around 45% of all of the glass present within a sample (by weight); again furthering the theory that a 20 mm screen appears to have the most potential for producing a less contaminated end product whilst not removing a large proportion of the biodegradable material.

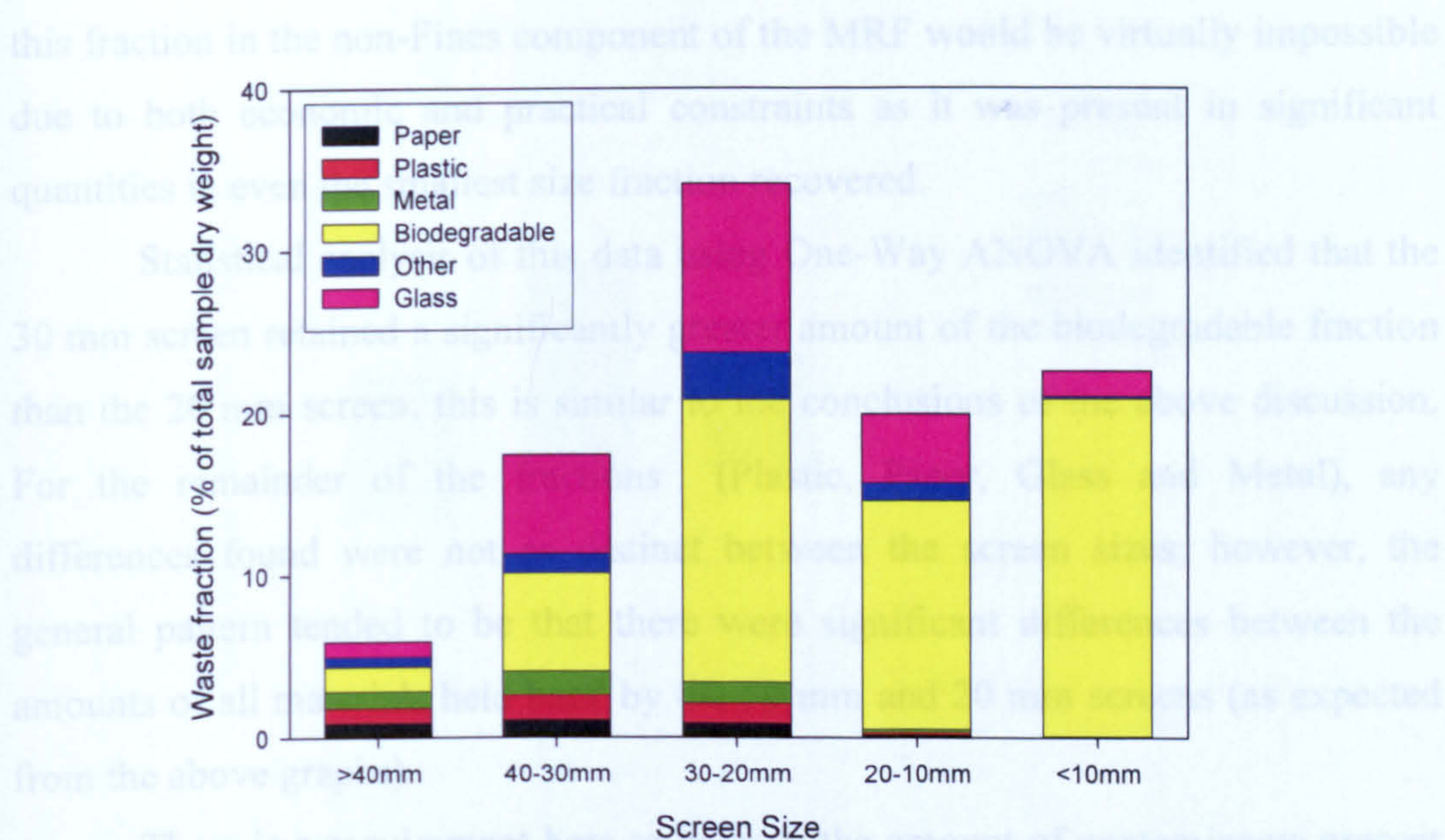


Figure 5.4: Average proportions of each set of screened material passing through different screen sizes. Values represent means of 5 independent sampling events.

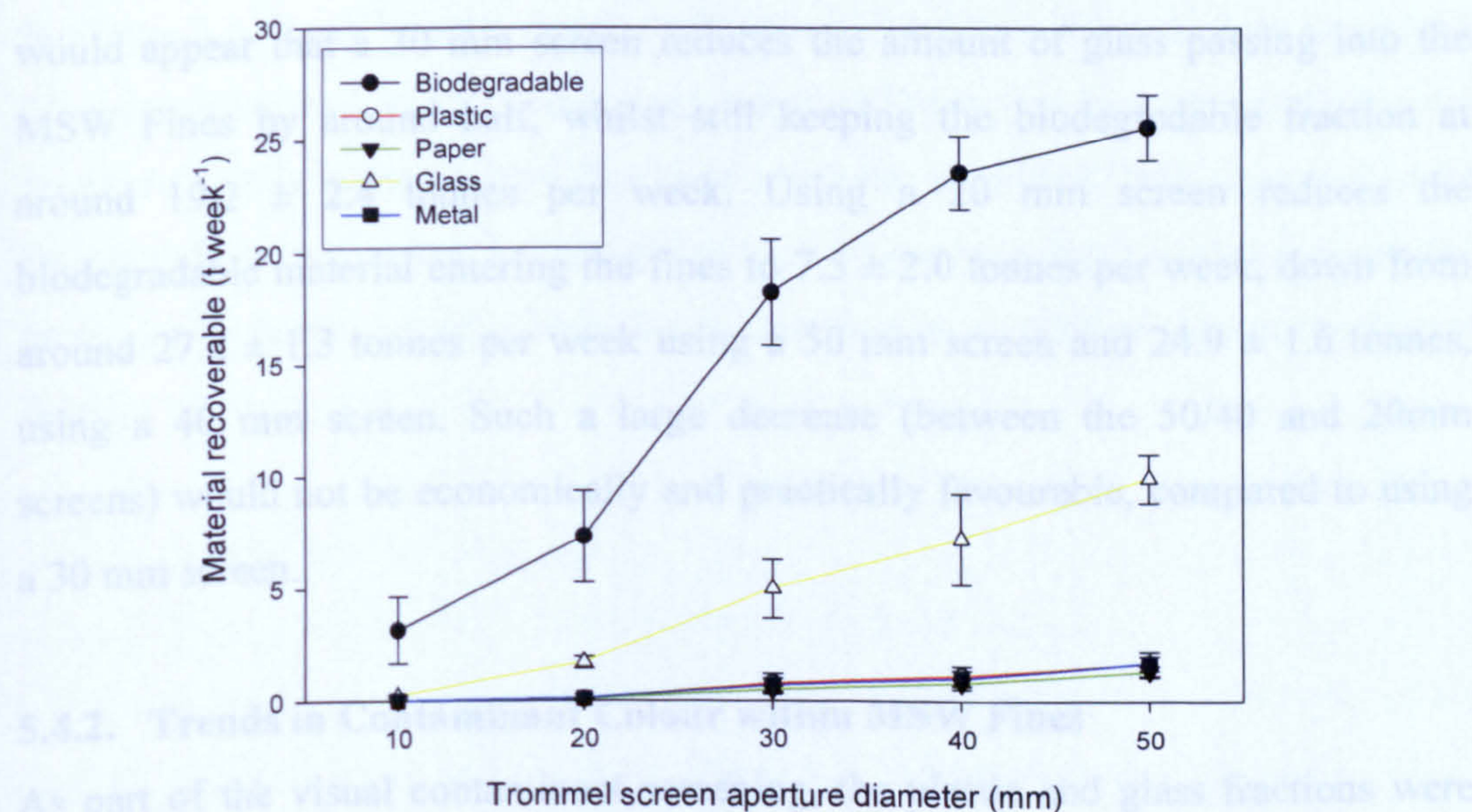


Figure 5.5: Theoretically recoverable tonnages from the MRF at Lampeter. Values are based upon the means of 5 independent sampling events.

On average, around 10 tonnes of glass passes through the trommels 50 mm screen each week into the MSW Fines category. However, in reality complete recovery of

this fraction in the non-Fines component of the MRF would be virtually impossible due to both economic and practical constraints as it was present in significant quantities in even the smallest size fraction recovered.

Statistical analysis of this data using One-Way ANOVA identified that the 30 mm screen retained a significantly greater amount of the biodegradable fraction than the 20 mm screen; this is similar to the conclusions of the above discussion. For the remainder of the fractions (Plastic, Paper, Glass and Metal), any differences found were not as distinct between the screen sizes; however, the general pattern tended to be that there were significant differences between the amounts of all materials held back by the 50 mm and 20 mm screens (as expected from the above graphs).

There is a requirement here to balance the amount of contaminants present within the MSW Fines with diverting as much biodegradable material as possible away from landfill, in accordance with the various pieces of legislation governing waste disposal (the main one being the 1999 Landfill Directive (EU, 1999)). It would appear that a 30 mm screen reduces the amount of glass passing into the MSW Fines by around half, whilst still keeping the biodegradable fraction at around 19.2 ± 2.4 tonnes per week. Using a 20 mm screen reduces the biodegradable material entering the fines to 7.5 ± 2.0 tonnes per week, down from around 27.2 ± 1.3 tonnes per week using a 50 mm screen and 24.9 ± 1.6 tonnes, using a 40 mm screen. Such a large decrease (between the 50/40 and 20mm screens) would not be economically and practically favourable, compared to using a 30 mm screen.

5.4.2. Trends in Contaminant Colour within MSW Fines

As part of the visual contaminant screening, the plastic and glass fractions were divided into different coloured fractions (e.g. clear glass, brown glass, etc.) and the dry weights of these coloured fractions recorded. The following figures show the percentages of each colour of material (plastic and glass) as a percentage of the total for that material (e.g. plastic as a percentage of total plastic in a sample).

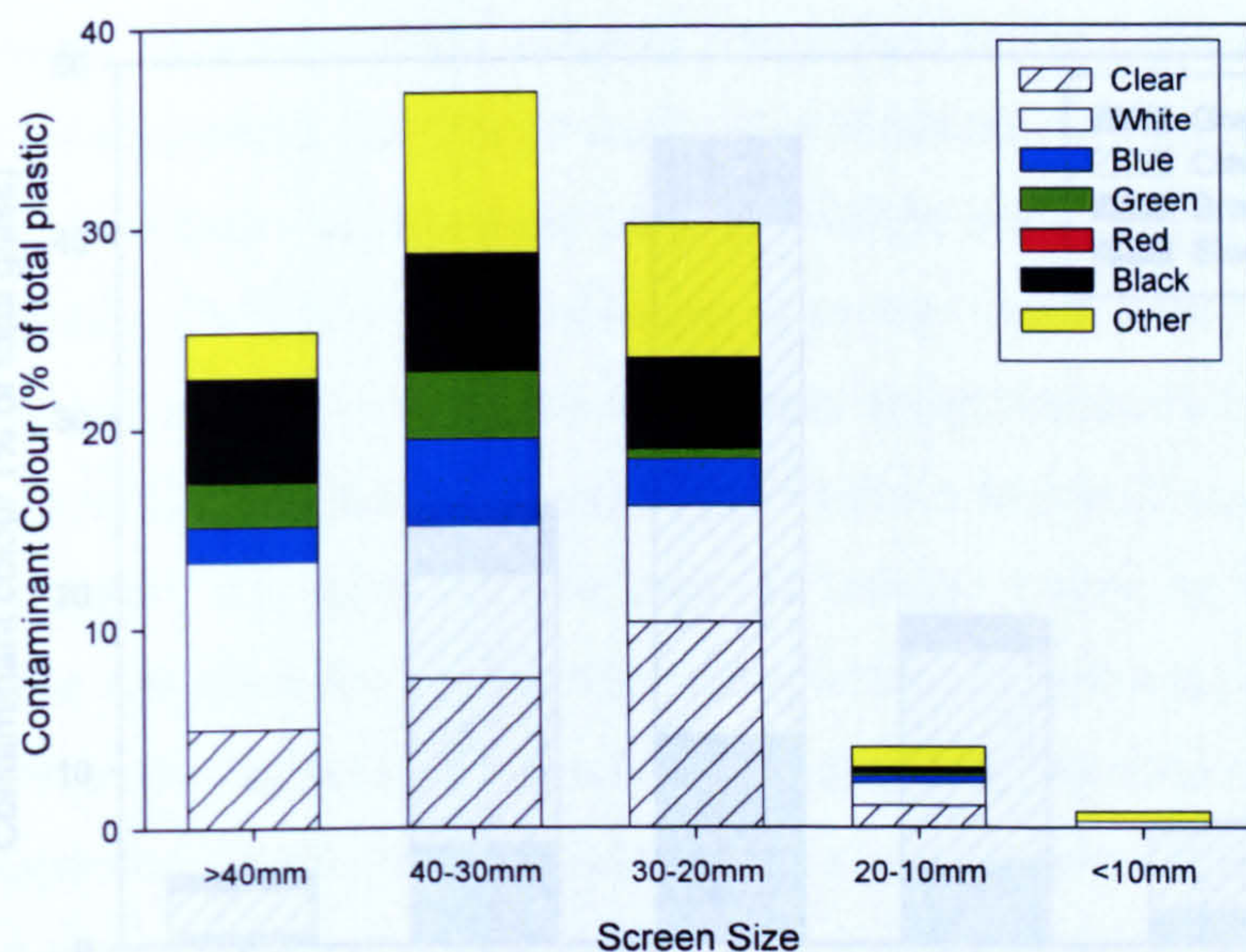


Figure 5.6: Percentages of each colour of plastic as a percentage of the total plastic within a sample. Values are based upon the means of 5 independent sampling events.

In Figure 5.6, it can be seen that the majority of the plastic fraction does not pass a 20 mm screen, with the 30 mm screen stopping the largest proportion of the total fraction (>30%). Of the different colours of plastic identified, both clear and white appear to have the largest percentages of each screen size, but it is also evident that there is a large amount of plastic that is classified as an “Other” colour than those listed.

5.5. Discussion

5.5.1. Comparison of two physical assessment methods

Finding reliable ways to physically characterise waste are important for waste audit purposes, the development of waste standards, and for determining the impact of different separation technologies on material recovery. Here we evaluated two physical separation techniques, namely one suited to an assessment of visual contamination (area-based assessment) with one suited to waste auditing purposes (weight-based assessment). From this study it is clear that there are significant differences between the results produced from the two methods. The results from

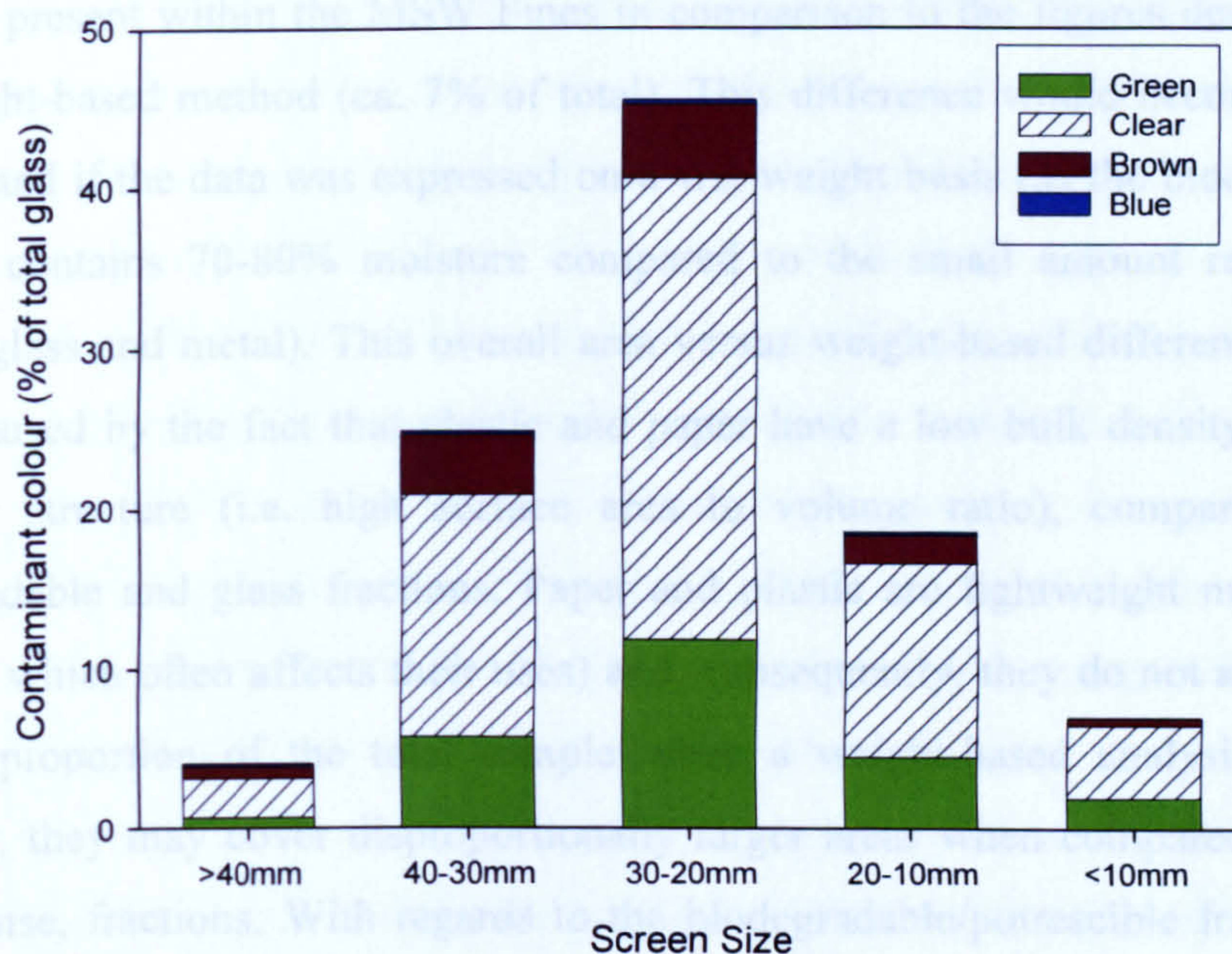


Figure 5.7: Percentages of each colour of glass as a percentage of the total glass within a sample. Values are based upon the means of 5 independent sampling events.

Clear glass was recorded as making up the greatest proportion of the total glass recovered from a sample of MSW Fines (Fig. 5.7). It is interesting to note that unlike the plastic fraction, there is both very little glass in the >40 mm fraction and in the <10 mm fraction; the majority of the glass found here is stopped by a 20 mm screen, although the 30 mm screen does retain over 20%.

5.5. Discussion

5.5.1. Comparison of two physical assessment methods

Finding reliable ways to physically characterise waste are important for waste audit purposes, the development of waste standards, and for determining the impact of different separation technologies on material recovery. Here we evaluated two physical separation techniques, namely one suited to an assessment of visual contamination (area-based assessment) with one suited to waste auditing purposes (weight-based assessment). From this study it is clear that there are significant differences between the results produced from the two methods. The results from

the visual/area method show a higher proportion of both paper and plastic (ca. 20% of total) present within the MSW Fines in comparison to the figures derived from the weight-based method (ca. 7% of total). This difference would become further accentuated if the data was expressed on a wet weight basis (as the biodegradable fraction contains 70-80% moisture compared to the small amount retained by plastic, glass and metal). This overall area versus weight-based difference is most likely caused by the fact that plastic and paper have a low bulk density and very laminate structure (i.e. high surface area to volume ratio), compared to the biodegradable and glass fractions. Paper and plastic are lightweight materials (a property which often affects their uses) and, consequently, they do not account for a large proportion of the total sample when a weight-based analysis is used, however, they may cover disproportionately larger areas when compared to other, more dense, fractions. With regards to the biodegradable/putrescible fraction, the results obtained from both analyses were similar, this fraction made up around 60% of the MSW Fines, this is also consistent with the figures quoted by the Environment Agency and the Welsh Assembly Government (Welsh Assembly Government, 2002).

The visual impact of more recalcitrant materials in the MSW Fines (e.g. plastic) can have a major knock-on effect for processes such as biological treatment (either composting or anaerobic digestion) that may be undertaken following separation from the main MSW stream. Based on the results presented here, any end-product produced from the treatment of MSW Fines would inevitably contain some physical contamination, most likely be in the form of glass or plastic. The presence of glass shards would present a significant problem in the use of any product that may come into contact with the general public. Plastic contamination would reduce the aesthetic appeal of a material made from biologically treated MSW Fines, again reducing the amount of possible end-uses further. The measurements made here indicate that this visual contamination is made more apparent by the spectral properties of the plastic contamination present. The biodegradable fraction is largely dark brown in colouration whilst the bright colours of the plastics (e.g. white, red and blue) disproportionately accentuate the

contamination. Therefore, future assessment of waste contamination should employ the area-based method (to accurately reflect the aesthetic qualities of waste) rather than a weight-based assessment (as per the current UK PAS-100 compost standards). However, even if this approach is adopted, further consideration needs to be given to the development of quality standards and the formulation of contaminant tolerance limits. These standards need to reflect contamination scoring on both a human/environmental health basis and on an aesthetic/visual amenity basis. To develop standards for glass contamination, however, it may be better to use the weight-based method as the tolerance limits will be much lower than those for plastics. This assessment should also include a qualitative physical assessment of the likelihood of the glass fragments to cause physical harm.

It is apparent that there is a significant amount of coloured contamination in the MSW Fines. It is possible that a visual recognition system could be used to eliminate coloured contaminants if the MSW Fines material was passes along a conveyer belt. Although this technology has been used successfully for contaminant removal in the textile industry (Abbott, 1995; Zhang *et al.*, 2005) to our knowledge it has not been used in the waste management sector for this purpose.

Many plants which treat MSW utilise a second stage of screening, with a smaller aperture being employed. Zach *et al.* (2000) studied a Mechanical Biological Pre-treatment plant at Oberpullendorf in Austria and found a 25 mm screen produced material high in biodegradable matter and low in physical contaminants. The material screened out using the 65 mm screen and then the 25 mm screen was subsequently recycled, landfilled or sent for incineration. No data for the relative proportions of each constituent was produced from the study by Zach *et al.* (2000).

The removal of contaminants could influence the subsequent processing of the MSW Fines particularly for composting. The large contaminants could enhance the aeration of the MSW Fines promoting their biodegradation, however, they could also dilute the self-heat generating capacity of the material (as they are

essentially non-biodegradable). A composting trial should therefore be undertaken to test the effect of screening upon the effectiveness of subsequent composting.

5.5.2. Economic Analysis of MSW Fines

Based on 2005-2009 UK market prices for waste materials (published weekly in *Recycling & Waste World*), uncontaminated MSW fines possess a very low intrinsic economic value in comparison to other segregated MRF-derived products. If contamination is present, then this value falls close to zero. If composted, however, its value will rise to €1-3 t⁻¹ somewhat lower than that for source-segregated green waste-derived compost at €7-10 t⁻¹. Assuming that the MSW Fines are composted and that a 50% loss of weight occurs, then the potential economic value of this to LAS Waste is ca. €1.5K y⁻¹. In comparison, recovered mixed glass waste has a much higher economic value of €17-22 t⁻¹. Based on the findings here the loss of glass into the MSW Fines waste stream equates to €11.0K y⁻¹ and therefore finding ways to enhance recovery of this fraction would be more desirable than optimizing the treatment of the biodegradable fraction. Similarly, the amount of metals lost into the MSW Fines stream has a market value of €8.9K y⁻¹ whilst that of mixed plastics is €0.1K y⁻¹. We conclude that further ways should be sought to determine ways to enhance recovery of metals and glass from the MSW prior to or post passage through the trommel screen. The results presented here employing different trommel screen sizes, however, suggest that this cannot be effectively achieved by altering the aperture of the trommel screen alone

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Investigation into the feasibility of using MSW-based feedstocks in small scale in-vessel composting.

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Investigation into the feasibility of using MSW-based feedstocks in small scale in-vessel composting.

6.1. Abstract

In-vessel composting facilities are becoming more commonplace, yet such installations are often highly capital intensive (ca. £1-10 million) to set up and run. Therefore, there is a need to test in-vessel composting systems on a smaller scale to establish the feasibility for using small scale plants in treating wastes. The main issue inherent with small scale plants is the temperature generated by the material present; larger vessels tend to have the potential to generate higher temperatures. Additionally, the presence of inert material in the compost feedstock can increase the aeration during composting but can also reduce the end quality of the compost produced and the heating potential of the feedstock. A trial was conducted utilising vessels with a 3 litre capacity connected to a vacuum pump system. The initial trial consisted of screening MSW Fines to remove particles of certain size to produce several materials with differing levels of physical contamination. The second set of trials tested the effect of adding nutrient solutions (organic/inorganic) to unscreened MSW Fines. The aims of the study were to firstly to evaluate the suitability of small scale benchtop composting vessels in multifactorial composting trials. Secondly, the effect of screening of MSW Fines upon the heating potential of the material was assessed. Thirdly, the effect of adding extra nutrients (e.g. labile carbon/nitrate) to the MSW Fines feedstock was recorded. The results of the screening trial did not, on the whole, follow those recorded from trials involving larger scale systems (e.g. increased electrical conductivity). Temperatures did not reach the same levels associated with larger trials and hence did not meet the requirements of legislation (e.g. ABPR 2005). Finer materials were recorded as having higher temperatures than coarser matter; this related to the increased proportion of biodegradable material in the sample. Addition of labile carbon (e.g. as sucrose) was seen to cause a secondary peak in temperature but no other major effects upon the feedstock material were recorded. Inorganic nutrient solutions produced no significant effects upon the composition of the MSW Fines being

composted. Disappointingly, the use of such small scale vessels failed to accurately replicate the changes documented in larger scale operations.

6.2. Introduction

Composting has been identified as a viable method with the potential to treat large amounts of mixed household waste prior to landfilling. The advantages can be many fold including a reduction in the volume of the material destined to landfill (Bari *et al.*, 2000; Norbu *et al.*, 2005), or the creation of products with other potential end uses (so long as the material meets with the requirements of various pieces of legislation; e.g. 1999 Landfill Directive; EU, 1999). Biological treatment of waste prior to landfilling also has scope for reducing the bioactivity of this material, a process that has several possible benefits including lower levels of greenhouse gas emission (e.g. CH₄) if the treated waste is subsequently landfilled (Hoeks, 1983; Mor *et al.*, 2006). Large scale in-vessel composting can often involve significant capital investment (ca. £1-10 million), making it potentially unattractive when compared to more common practices such as recycling or landfill, at least in the short term. Smaller scale ventures would therefore seem more attractive due to the reduced cost, although this could severely limit the amount of waste which could be processed at any one time.

One major problem with composting research is that the process needs to be undertaken on a large scale to generate temperatures sufficient to make the process happen in a realistic way. If replication is added to these experiments they have a large spatial footprint making comparison of many treatments difficult, extremely costly and time intensive. Critical to compost experimentation is heat retention at the edge of the compost heap or vessel as otherwise the compost cools too rapidly and the rate of decomposition falls dramatically (Hogan *et al.*, 1989; Lau *et al.*, 1992; Bari *et al.*, 2000). Vessels that are insulated from the surrounding atmosphere are advantageous in this respect; replicating these heat retaining vessels at the bench-reactor scale remains difficult (Vinneras *et al.*, 2003; Sundberg *et al.*, 2004). Consequently, it is often difficult to experimentally test and compare many different feedstock mixes to determine which mixture composts quickest and which

produces the best end-product. This lack of laboratory technology is severely limiting the adoption of best practice by the industry. In addition, it also prevents them from trialling what are perceived to be ‘difficult to treat’ waste streams (e.g. citrus pulp from food processing factories which has a high acidity; Van Heerden *et al.*, 2002). This thesis chapter aims to evaluate a new way to experimentally test multiple compost treatments simultaneously.

The objectives of this study were as follows:

- To evaluate the suitability of using very small scale compost vessels for multifactorial composting trials, and,
- To evaluate the effect that screening waste to different size fractions has upon the composting process.

6.3. Materials and methods

6.3.1. Screening trial

MSW Fines were collected from a Materials Recovery Facility (MRF) at Lampeter, Ceredigion and passed through a set of trommel screens to remove various constituent materials (see Chapter 5 for further details). The screens used had holes which were 10, 20, 30, 40 and 50 mm in diameter. In theory, removal of large pieces of inert matter would create a denser material with less void space, which in turn could have a three-fold effect. Firstly, it could result in reduced aeration within the composting vessel. This may increase or decrease the temperature profile in the vessel depending upon the thermal heat capacity of the material and its heat generating potential. Secondly, it could be beneficial by removing potential physical hazards (e.g. large pieces of glass and metal). Thirdly, it may remove inert material that has little or no heat generating potential and which limits compost self-heating. This trial was therefore set up to test whether screening prior to heat treatment would have a significant effect upon the temperatures that can be reached by the MSW Fines.

The composting vessels used here were commercially available “air-pots” usually utilised for the storage and dispensing of hot liquids via a simple pump-type

mechanism (vacuum flasks); the ones used here were manufactured by Olympia Airpots (Model Number K636) and possessed a 3 l capacity.

These containers were connected to a vacuum pump system, allowing air to be drawn through the vessel at a standard rate of $0.5 \text{ litres min}^{-1}$ as shown below:

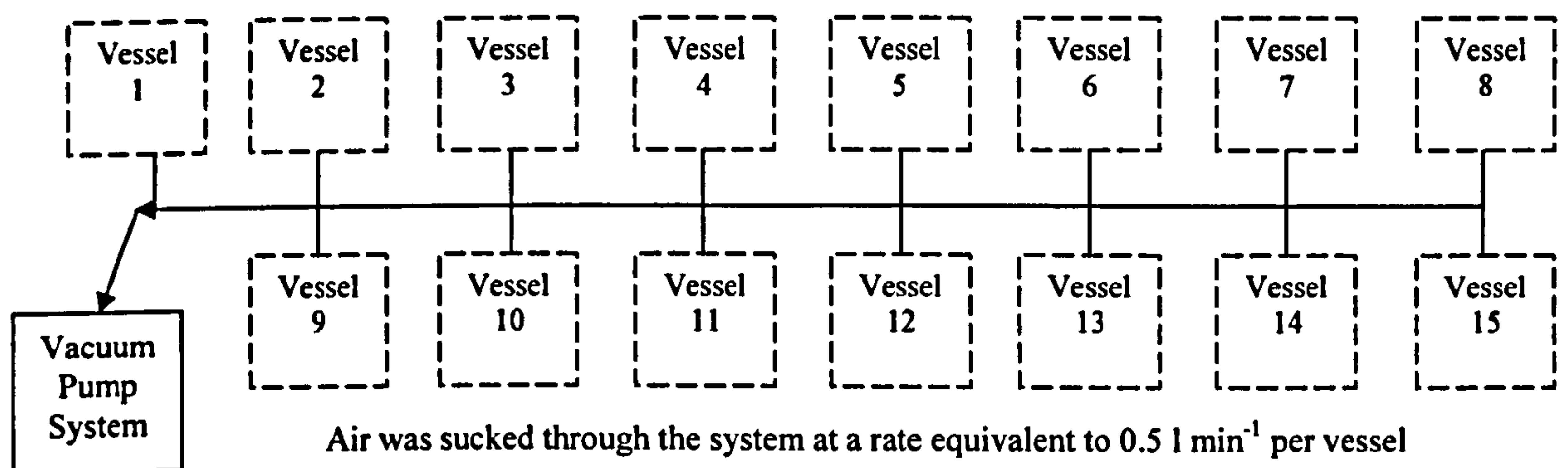


Figure 6.1: Schematic diagram of heating vessel set up connected to vacuum pump system to draw air through at a rate of 0.5 l min^{-1} per vessel as used in the experiments detailed in this section.

Each flask was filled with between approximately 700 and 900g of MSW Fines (the amount varied with different screen size but equated to a 3 l volume), screened through different mesh sizes and connected to a vacuum system, set to a flow rate of $30 \text{ l vessel}^{-1} \text{ h}^{-1}$ (0.5 l min^{-1}). The weights of the loaded flasks were recorded at the start and end of the experiment, to allow mass loss from the system to be calculated as both a weight and a percentage. As the main purpose of the trial was to evaluate thermophilic heat generating capacity of the material, most attention was paid to the first week of composting. Temperatures in the middle of the vessel were recorded using a 15 cm long Checktemp-1 digital thermometer (Hanna Instruments Ltd, Leighton Buzzard, UK) at regular intervals of 0, 0.5, 1, 2, 4, 8, 12, 18, 24 h and then every 24 hours until the 8th day (192 h). The ambient temperature was also recorded during the trial ($n = 3$ at each time interval).

6.3.2. Nutrient limitation trials

Following the initial screening-based trials, it was decided that the next step would be to test the effects of organic and inorganic nutrient limitation during the composting process and the use of additives as potential “kickstarters”. The MSW

Fines used in the trial were taken directly from the MRF at Lampeter (i.e. screened <50 mm). The weights of the loaded flasks were recorded at the start and end of the experiment, to allow mass loss from the system to be calculated as both a weight and a percentage. For the organic nutrient limitation section of the trial, 50 ml of one of the following solutions was added to replicate vessels.

1. Control (distilled water)
2. Labile source of C (100 mM sucrose)
3. Labile source of C and N (100 mM sucrose + 10 mM glutamate)

Five replicates of each solution were prepared, giving 15 flasks in total for a 6 day trial; with the solutions being mixed into the MSW Fines thoroughly before the material was placed into the vessels.

Potential inorganic nutrient limitation within the vessels was evaluated using the same method as described above but with the following solutions:

- Control (distilled water).
- Inorganic N and P (50 mM KNO_3 + 10 mM K_2HPO_4).
- Commercial nutrient additive (Garotta®; William Sinclair Horticulture Ltd, Lincoln, UK) prepared according to the manufacturer instructions.

The active ingredients in Garotta® are listed by the manufacturers as nitrogen and ground limestone. According to the manufacturers, “Garotta® is made from natural materials selected to help improve the structure of home-produced garden compost, maintain a neutral pH and speed up the process of decomposition. It provides food to promote bacterial growth which in turn encourages the temperature in the compost heap to rise, kill off weed seeds and hasten the conversion of garden and kitchen waste into a rich compost ideal for enriching garden soil”.

6.3.3. Physical and chemical analysis

Samples of the MSW Fines were taken for chemical analysis prior to filling of the vessel, and once the experiments were completed. pH (Hanna Instruments pH 209 pH meter) and electrical conductivity (EC) (Jenway 4010 EC meter) were determined in a 1:1 (v/v) deionized water:waste slurry, after mixing for 1 h. Moisture content was determined by drying samples at 80°C for 24 h and organic matter content was determined as loss-on-ignition at 450°C (16 h). Total C and N were determined using a CHN2000 analyser (LECO Corp., St Joseph, MI); prior to CHN analysis, samples were air-dried (25°C, 14 d) before being ground in a T1-100 vibrating sample mill (Heiko Co Ltd., Fukushima, Japan) equipped with tungsten grinding vessels. Nitrate and ammonium were extracted with 1 M KCl and deionised water at a 1:5 (w/v) fresh sample:extractant ratio for 1 h on a reciprocating shaker at 250 rev min⁻¹ (Zhong and Makeschin, 2003). The supernatant was removed by centrifugation of material at 4000 rev min⁻¹ for 30 min and the collected solutions filtered through a Whatman 42 filter paper. Nitrate and ammonium were determined using a San⁺⁺ segmented flow analyser (Skalar Inc., Norcross, GA). Extractable K, Na, Ca and P were extracted using 1 M ammonium acetate, buffered to pH 7.0 using 0.5 M acetic acid, and deionised water using 1:5 (w/v) fresh sample:extractant ratio (Helmke and Sparks (1996). K, Na and Ca were analysed using a Sherwood 410 flame photometer (Sherwood Scientific, Cambridge, UK). P was obtained by extraction, firstly with NaHCO₃ buffered to pH 8.5, and also with deionised water using a 1:5 (w/v) fresh sample:extractant ratio for 1 h on a reciprocating shaker at 250 rev min⁻¹. The supernatant was removed by centrifugation of material at 4000 rev min⁻¹ for 30 min and the collected solutions filtered through a Whatman 42 filter paper. Phosphate was determined colorimetrically using the method of Murphy and Riley (1962). Water extractable dissolved organic C (DOC), dissolved organic N (DON) and total dissolved N (DN) were measured using a Shimadzu TOC-V-TN analyser (Shimadzu Corp., Kyoto, Japan).

6.3.4. Statistical analysis

All data were inputted into SPSS v12.0 (SPSS Inc., Chicago, IL), and an ANOVA performed. Tukey’s Honestly Significant Difference (HSD) post hoc test was used to differentiate between the compost treatments. Means and standard errors were computed using Microsoft Excel 2003 (Microsoft Corp., Redmond, WA), and all graphical figures were produced in SigmaPlot v8.0 (Systat Software Inc., San Jose, CA). All nutrient values are expressed on a dry weight basis.

6.4. Results

6.4.1. Experiment 1: Size screening trial

6.4.1.1. Composting vessel temperature

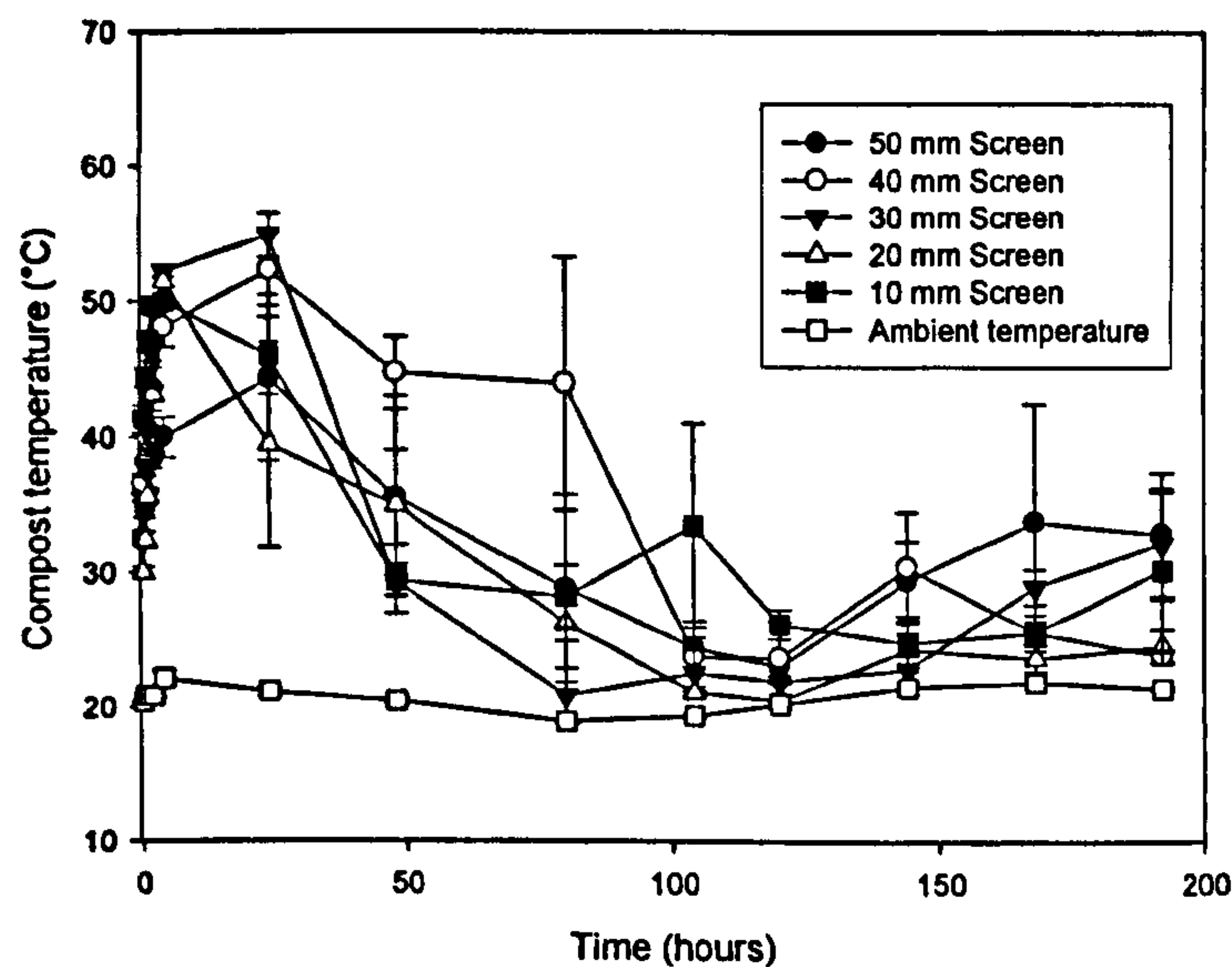


Figure 6.2: Changes in compost temperature in different sized MSW Fines screenings over a period of 192 h during a benchtop scale trial. Symbols represent means ± SEM (n = 5)

Temperature was observed to increase rapidly during the initial 24 h of the trial, with the majority of flasks undergoing cooling in the subsequent days. The material that had been passed through the 30 mm screen recorded the highest average temperature ($54.9 \pm 1.6^{\circ}\text{C}$) after 24 h, yet 56 h after this measurement was taken, this material had cooled by over 30°C . The majority of the samples cooled to under 30°C after 100 h, only the 10 mm screen fraction being higher ($33.5 \pm 7.5^{\circ}\text{C}$). For the remainder of the trial, a slight increase was recorded in several of the size

fractions. Ambient temperatures remained between $18.9 \pm 0.1^{\circ}\text{C}$ and $22.2 \pm 0.6^{\circ}\text{C}$ for the duration of the trial, with little diurnal variation being recorded.

Significant differences were present between the different sized materials for the first 4 h of heating, with the coarsest screenings (50 mm) having highly significantly lower temperatures than the finest screenings (10 mm). Once the 4 h mark had passed, no further differences were identified between any of the replicates. T-tests also identified some differences between some of the different sized screenings but no overall conclusions could be drawn.

6.4.1.2. Compost pH

| Screen Size | Start | End | Significance |
|-------------|-----------------|-----------------|--------------|
| 50 mm | 7.55 ± 0.03 | 7.86 ± 0.29 | $p = 0.356$ |
| 40 mm | 7.20 ± 0.22 | 7.39 ± 0.62 | $p = 0.732$ |
| 30 mm | 7.55 ± 0.02 | 8.29 ± 0.18 | $p = 0.066$ |
| 20 mm | 7.40 ± 0.10 | 8.20 ± 0.11 | $p = 0.0001$ |
| 10 mm | 7.81 ± 0.14 | 8.16 ± 0.08 | $p = 0.232$ |

Table 6.1: pH values recorded in different sized MSW screenings prior to and following 192h residence within a bench top heating vessel. All values represent means \pm SEM. Significance is for each screen size from start to end (e.g. 50mm Start to 50mm End)

Compost pH increased in every treatment following 8 d heating ($p < 0.05$). Generally, the finer screened material was observed as having a lower pH value than the larger screened material, with the largest overall change occurring in the material screened to 20 mm (increased by 0.8 ± 0.01). At the beginning of the trial, the 40 and 10 mm screenings had significantly different pH readings; by the end of the heating period, however, no differences were present.

6.4.1.3. Compost electrical conductivity (EC)

| Screen Size | Start | End | Significance |
|-------------|-----------|------------|------------------|
| 50 mm | 6.8 ± 0.7 | 8.0 ± 0.9 | <i>p</i> = 0.451 |
| 40 mm | 5.2 ± 0.4 | 7.8 ± 1.1 | <i>p</i> = 0.270 |
| 30 mm | 7.8 ± 0.1 | 7.0 ± 1.0 | <i>p</i> = 0.823 |
| 20 mm | 6.8 ± 0.2 | 9.4 ± 2.4 | <i>p</i> = 0.552 |
| 10 mm | 8.0 ± 0.9 | 10.1 ± 1.5 | <i>p</i> = 0.051 |

Table 6.2: Electrical Conductivity values (mS cm⁻¹) recorded in different sized MSW screenings prior to and following 192 h residence within a bench top heating vessel. All values represent means ± SEM. Significance is for each screen size from start to end (e.g. 50mm Start to 50mm End)

Electrical conductivity followed a similar pattern to pH change, with most replicates undergoing an increase with time (only the 30 mm screenings showed a lower EC after heating). ANOVA results showed that screening has no significant effect upon EC before or after heating; T-tests also identified no effect of heating upon the EC (*p* > 0.05).

6.4.1.4. KCl-extractable nitrate

| Screen Size | Start (mg N/kg) | End (mg N/kg) | Significance |
|-------------|-----------------|---------------|------------------|
| 50 mm | 0.71 ± 0.71 | <0.01 | <i>p</i> = 0.423 |
| 40 mm | 0.10 ± 0.10 | 0.25 ± 0.15 | <i>p</i> = 0.185 |
| 30 mm | 1.84 ± 0.67 | 0.89 ± 0.11 | <i>p</i> = 0.340 |
| 20 mm | <0.01 | 0.34 ± 0.34 | <i>p</i> = 0.423 |
| 10 mm | 0.83 ± 0.23 | <0.01 | <i>p</i> = 0.070 |

Table 6.3: KCl Extractable NO₃⁻ values recorded in different sized MSW screenings prior to and following 192 h residence within a bench top heating vessel. All values represent means ± SEM. Significance is for each screen size from start to end (e.g. 50mm Start to 50mm End)

There was a large degree of variation in the results obtained for KCl extractable NO₃⁻; both the material screened to 10 mm and that from the trommel screen (50 mm) were recorded as having no detectable NO₃⁻ present post-heating. The 20 mm screenings showed the opposite trend, with negligible NO₃⁻ present to begin, but 0.34 ± 0.34 mg/kg present 8 d later. Due to the size of the standard errors recorded, no definitive conclusion can be drawn from these results.

6.4.1.5. KCl-extractable ammonium

| Screen Size | Start (mg N/kg) | End (mg N/kg) | Significance |
|-------------|-----------------|---------------|--------------|
| 50 mm | 244 ± 10 | 200 ± 17 | $p = 0.254$ |
| 40 mm | 199 ± 36 | 186 ± 25 | $p = 0.844$ |
| 30 mm | 230 ± 29 | 182 ± 20 | $p = 0.414$ |
| 20 mm | 210 ± 21 | 190 ± 4 | $p = 0.432$ |
| 10 mm | 213 ± 18 | 198 ± 4 | $p = 0.550$ |

Table 6.4: KCl Extractable NH_4^+ values recorded in different sized MSW screenings prior to and following 192 h residence within a bench top heating vessel. All values represent means ± SEM. Significance is for each screen size from start to end (e.g. 50mm Start to 50mm End)

All treatments were found to show reduced NH_4^+ content after heating. When this data is compared to that of the KCl extractable NO_3^- , it can be said that that the amount of N present within the material decreases overall; this is likely to be a result of gaseous losses of N. Analysis showed that there is no significant effect of screen size upon the level of extractable NH_4^+ . However, the material that had been heated contained significantly lower levels of NH_4^+ than the parent MSW Fines ($p < 0.05$) when all screen sizes were considered.

6.4.1.6. Extractable P

6.4.1.6.1. Olsen P extraction

| Screen Size | Start (mg/kg) | End (mg/kg) | Significance |
|-------------|---------------|-------------|--------------|
| 50 mm | 243 ± 74 | 401 ± 46 | $p = 0.778$ |
| 40 mm | 227 ± 131 | 472 ± 40 | $p = 0.902$ |
| 30 mm | 189 ± 70 | 434 ± 100 | $p = 0.967$ |
| 20 mm | 256 ± 49 | 395 ± 38 | $p = 0.199$ |
| 10 mm | 324 ± 66 | 368 ± 68 | $p = 0.528$ |

Table 6.5: Olsen P values recorded in different sized MSW screenings prior to and following 192 h residence within a bench top heating vessel. All values represent means ± SEM. Significance is for each screen size from start to end (e.g. 50mm Start to 50mm End)

In all 5 treatments the amount of Olsen extractable P increased following heating. The largest increases occurred within the 30 mm and 40 mm fractions; material screened to 10 mm underwent little change by comparison. The results from

ANOVA analysis show that changing screen size had no significant effect upon Olsen P levels. T-tests showed the end values had significantly higher amounts of Olsen P than the parent MSW Fines ($p < 0.05$) when all samples were considered.

6.4.1.6.2. Water-extractable P

| Screen Size | Start (mg/kg) | End (mg/kg) | Significance |
|-------------|---------------|-------------|--------------|
| 50 mm | 244 ± 31 | 159 ± 31 | $p = 0.085$ |
| 40 mm | 202 ± 67 | 135 ± 17 | $p = 0.902$ |
| 30 mm | 95 ± 43 | 167 ± 51 | $p = 0.215$ |
| 20 mm | 187 ± 29 | 76 ± 27 | $p = 0.204$ |
| 10 mm | 144 ± 26 | 46 ± 26 | $p = 0.099$ |

Table 6.6: H₂O extractable P values recorded in different sized MSW screenings prior to and following 192 h residence within a bench top heating vessel. All values represent means ± SEM. Significance is for each screen size from start to end (e.g. 50mm Start to 50mm End)

In direct comparison to the results for Olsen P, the H₂O extractable content was observed to decrease in the majority of samples; only the material screened to 30 mm was recorded to increase. ANOVA analysis identified no significant differences that are caused by screening of the materials.

6.4.1.7. Ammonium acetate extractable cations

6.4.1.7.1. Calcium (Ca)

| Screen Size | Start (mg/kg) | End (mg/kg) | Significance |
|-------------|---------------|-------------|--------------|
| 50 mm | 1160 ± 100 | 1331 ± 98 | $p = 0.378$ |
| 40 mm | 1336 ± 163 | 1091 ± 200 | $p = 0.483$ |
| 30 mm | 1841 ± 219 | 1003 ± 162 | $p = 0.148$ |
| 20 mm | 1878 ± 370 | 1275 ± 208 | $p = 0.393$ |
| 10 mm | 2159 ± 238 | 977 ± 112 | $p = 0.063$ |

Table 6.7: Ammonium acetate extractable Ca recorded in different sized MSW screenings prior to and following 192 h residence within a bench top heating vessel. All values represent means ± SEM. Significance is for each screen size from start to end (e.g. 50mm Start to 50mm End)

All screening sizes, with the exception of the 50 mm material, were found to have decreased in Ca content following heating. There was also a slightly inverse relationship between Ca and screen size in that as the size of screen increased, Ca

content decreased. No significant differences were identified between any of the different sized screenings or between the heated and unheated materials.

6.4.1.7.2. Potassium (K)

| Screen Size | Start (mg/kg) | End (mg/kg) | Significance |
|-------------|---------------|-------------|------------------|
| 50 mm | 3238 ± 536 | 4592 ± 55 | <i>p</i> = 0.149 |
| 40 mm | 3371 ± 348 | 4197 ± 534 | <i>p</i> = 0.313 |
| 30 mm | 3613 ± 409 | 5426 ± 637 | <i>p</i> = 0.218 |
| 20 mm | 3934 ± 835 | 5544 ± 538 | <i>p</i> = 0.308 |
| 10 mm | 4158 ± 282 | 6355 ± 653 | <i>p</i> = 0.091 |

Table 6.8: Ammonium acetate extractable K recorded in different sized MSW screenings prior to and following 192 h residence within a bench top heating vessel. All values represent means ± SEM. Significance is for each screen size from start to end (e.g. 50mm Start to 50mm End)

A slight increase in K was recorded as heating progressed, whilst increased screen size appeared to produce lower K levels (a similar relationship is shown in the Ca data). As with the Ca data, no significant differences between screen sizes were identified here. The heated material showed differences between some of the samples and between the heated and unheated materials (the latter having lower K levels than the former (*p* < 0.01)).

6.4.1.7.3. Sodium (Na)

| Screen Size | Start (mg/kg) | End (mg/kg) | Significance |
|-------------|---------------|-------------|------------------|
| 50 mm | 3093 ± 395 | 3045 ± 37 | <i>p</i> = 0.923 |
| 40 mm | 3553 ± 443 | 2780 ± 358 | <i>p</i> = 0.301 |
| 30 mm | 3569 ± 300 | 3604 ± 423 | <i>p</i> = 0.966 |
| 20 mm | 3937 ± 734 | 3681 ± 355 | <i>p</i> = 0.792 |
| 10 mm | 4274 ± 377 | 4221 ± 434 | <i>p</i> = 0.932 |

Table 6.9: Ammonium acetate extractable Na recorded in different sized MSW screenings prior to and following 192 h residence within a bench top heating vessel. All values represent means ± SEM. Significance is for each screen size from start to end (e.g. 50mm Start to 50mm End)

As with both Ca and K, it was found that Na levels decreased with increasing screen size prior to heating. However, once the material had been heated, the

amount of extractable Na was only slightly altered, with all (excluding the 30 mm screenings) showing lower Na levels. Na also is similar to Ca and K in that there are no statistical differences between the materials when they are screened to different sizes; no differences were identified between the unheated and heated materials ($p > 0.05$).

6.4.1.8. Moisture content

| Screen Size | Start (%) | End (%) | Significance |
|-------------|------------|------------|--------------|
| 50 mm | 67.8 ± 0.9 | 60.1 ± 3.0 | $p = 0.220$ |
| 40 mm | 59.9 ± 8.2 | 53.8 ± 6.9 | $p = 0.713$ |
| 30 mm | 60.8 ± 2.7 | 51.3 ± 5.3 | $p = 0.385$ |
| 20 mm | 55.2 ± 4.8 | 53.5 ± 0.9 | $p = 0.787$ |
| 10 mm | 52.5 ± 3.9 | 52.5 ± 0.6 | $p = 0.999$ |

Table 6.10: Moisture content recorded in different sized MSW screenings prior to and following 192 h residence within a bench top heating vessel. All values represent means ± SEM. Significance is for each screen size from start to end (e.g. 50mm Start to 50mm End)

Moisture content remained relatively stable for the duration of heating; the larger fractions (>30 mm) did appear to lose around 5% of their original moisture. It can also be seen that the coarser a material was, the higher the initial moisture content a sample had. It was found that no significant differences existed between the moisture content readings between any samples.

6.4.1.9. Organic matter content

| Screen Size | Start (%) | End (%) | Significance |
|-------------|-------------|-------------|--------------|
| 50 mm | 69.1 ± 2.6 | 53.6 ± 10.8 | $p = 0.205$ |
| 40 mm | 47.8 ± 12.8 | 47.6 ± 8.3 | $p = 0.921$ |
| 30 mm | 64.0 ± 4.1 | 58.4 ± 1.5 | $p = 0.427$ |
| 20 mm | 50.9 ± 7.0 | 46.1 ± 4.9 | $p = 0.612$ |
| 10 mm | 47.3 ± 6.9 | 45.6 ± 0.1 | $p = 0.829$ |

Table 6.11: Organic matter recorded in different sized MSW screenings prior to and following 192 h residence within a bench top heating vessel. All values represent means ± SEM. Significance is for each screen size from start to end (e.g. 50mm Start to 50mm End)

The general trend for organic matter (OM) content was that the coarser fractions had higher levels, although all 5 treatments had reduced OM content following heating. Following heating, the 30 mm fraction was found to contain a significantly higher amount of OM than the 10mm fraction ($p < 0.05$) but no other differences could be identified.

6.4.2. Experiment 2: Organic nutrient limitation

6.4.2.1. Compost temperature

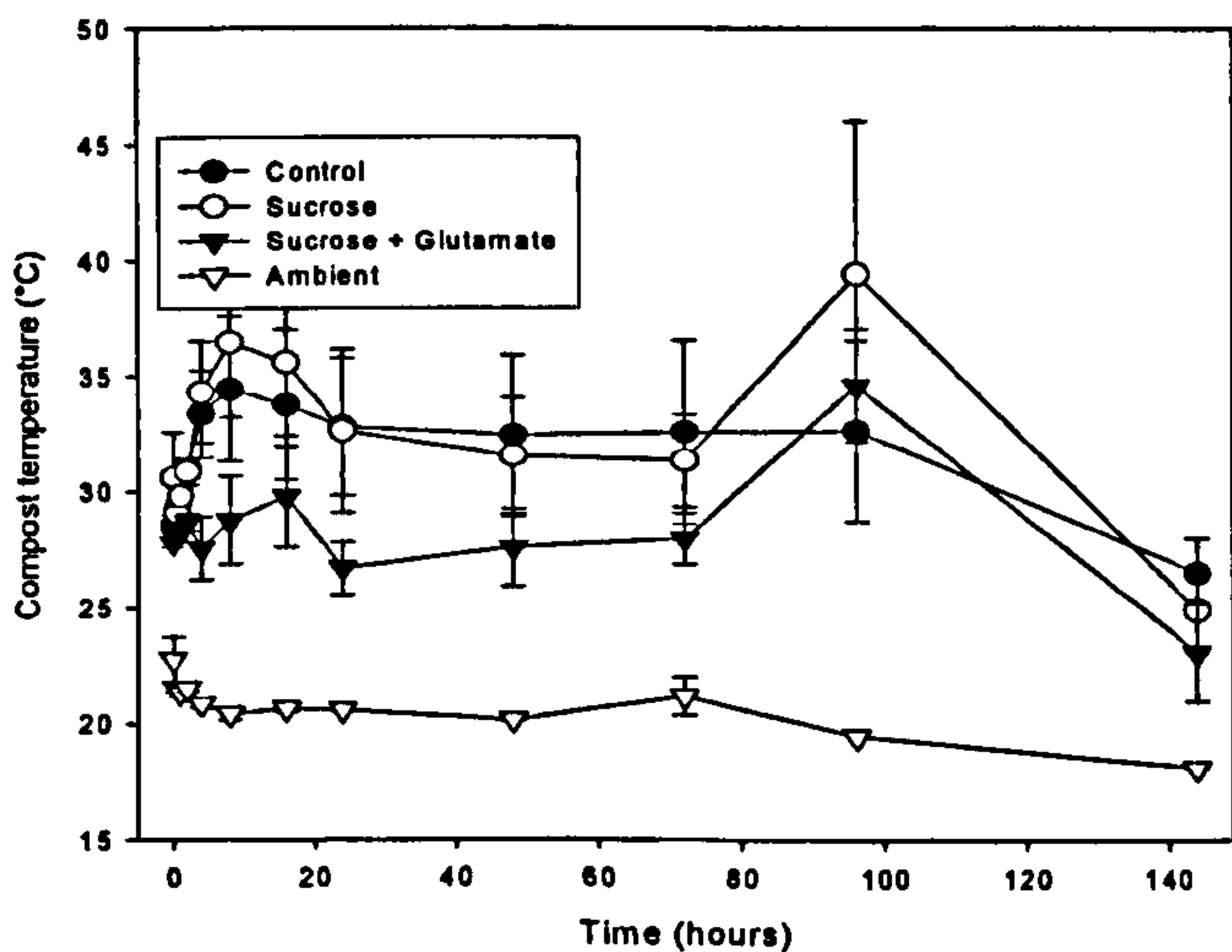


Figure 6.3: Changes in temperature in MSW Fines mixed with various organic nutrient solutions over a period of 144 h during a benchtop scale trial. All values represent means ± SEM

During the Organic Nutrient Trial, two distinct peaks in temperature occurred, one around 10-15 h and a second at 96 h; neither of these peaks appears to be a result of any changes recorded in ambient temperature at any time, rather they are directly related to changes within the MSW Fines. The Sucrose treatment had significantly higher temperatures than the Sucrose + Glutamate treatment ($p < 0.001$) whilst the Control MSW Fines (no treatment) were also found to have significantly higher temperatures than the Sucrose + Glutamate treatment ($p < 0.001$).

6.4.2.2 Chemical analysis results

| | MSW Fines (at start $t = 0$) | Control material | Fines + sucrose solution | Fines + sucrose and glutamate solution |
|--|----------------------------------|------------------|-----------------------------|---|
| pH | 7.26 ± 0.48 | 7.84 ± 0.21 | 7.80 ± 0.42 | 7.37 ± 0.53 |
| EC (mS cm ⁻¹) | 3.5 ± 0.6 | 4.2 ± 0.6 | 3.4 ± 0.4 | 3.8 ± 0.5 |
| KCl NO ₃ ⁻ (mg N kg ⁻¹) | 2.27 ± 1.05 | 0.10 ± 0.10 | 1.04 ± 0.43 | <0.01 |
| KCl NH ₄ ⁺ (mg N kg ⁻¹) | 166 ± 29 | 244 ± 26 | 247 ± 20 | 252 ± 10 |
| H ₂ O NO ₃ ⁻ (mg N kg ⁻¹) | 0.98 ± 0.72 | 0.57 ± 0.41 | 0.83 ± 0.37 | 0.36 ± 0.22 |
| H ₂ O NH ₄ ⁺ (mg N kg ⁻¹) | 163 ± 25 | 232 ± 27 | 237 ± 20 | 248 ± 14 |
| Olsen P (mg kg ⁻¹) | 239 ± 54 | 237 ± 75 | 210 ± 76 | 323 ± 60 |
| H ₂ O P (mg kg ⁻¹) | 60 ± 49 | 88 ± 54 | 148 ± 76 | 25 ± 5 |
| Moisture content (%) | 58.5 ± 0.8 | 63.4 ± 3.6 | 62.0 ± 1.3 | 63.0 ± 1.2 |
| Organic matter content (%) | 55.9 ± 1.6 | 57.8 ± 4.7 | 58.4 ± 3.1 | 57.8 ± 3.3 |
| <u>Ammonium acetate extractable cations</u> | | | | |
| Ca (mg kg ⁻¹) | 1602 ± 84 | 1954 ± 219 | 1582 ± 84 | 1615 ± 141 |
| K (mg kg ⁻¹) | 4213 ± 104 | 4059 ± 301 | 4219 ± 289 | 3997 ± 249 |
| Na (mg kg ⁻¹) | 2666 ± 77 | 3288 ± 234 | 3362 ± 155 | 3844 ± 284 |
| <u>Water extractable cations</u> | | | | |
| Ca (mg kg ⁻¹) | 2146 ± 38 | 2212 ± 209 | 2649 ± 138 | 2817 ± 143 |
| K (mg kg ⁻¹) | 2884 ± 146 | 3075 ± 273 | 3079 ± 139 | 3177 ± 117 |
| Na (mg kg ⁻¹) | 1951 ± 138 | 2641 ± 200 | 2456 ± 83 | 2644 ± 138 |

Table 6.12: Chemical analysis results from the MSW compost to which sucrose and glutamate had been added as soluble forms of C and N at the start and after 144 h residence within a bench top heating vessel. All values represent means ± SEM

6.4.2.3. Compost pH

pH varied only slightly between the original MSW Fines and the treated material; both the material with no additive and that treated with Sucrose solution showed an increase in pH. No significant differences were identified by statistical analysis between each of the treatments ($p > 0.05$).

6.4.2.4. Compost electrical conductivity

EC was observed to increase in all three treatments following heating; yet this increase was found to be greatest in the control Fines that had not received any nutrient solution. Despite this increase, there were no significant differences recorded between the EC readings of any of the treatments ($p > 0.05$).

6.4.2.5. Extractable nitrate

6.4.2.5.1. KCl-extractable nitrate

There was a large amount of variation between the different materials tested here. All three treatments decreased in extractable NO_3^- content, although the control material was found to have less extractable NO_3^- than the material which had Sucrose added to it. The Fines which had the Sucrose + Glutamate added were recorded as containing no detectable extractable NO_3^- in any replicate. No significant differences were identified between the NO_3^- contents of the materials, despite the decreases that had occurred ($p > 0.05$).

6.4.2.5.2. Water-extractable nitrate

As for the KCl extraction data, the Sucrose treatment showed a higher NO_3^- content than the other treatments. However, all three treatments were found to have a lower NO_3^- content than the unheated material. No significant differences were found between the readings taken for this stage of the analysis ($p > 0.05$).

6.4.2.6. Extractable ammonium

6.4.2.6.1. KCl-extractable ammonium

NH_4^+ was present in much higher levels than NO_3^- and increased by around 100 mg/kg in all samples following heating. No large differences were recorded between the three heated materials. Statistical testing shows that there are no significant differences between any of the treatments following heating ($p > 0.05$).

6.4.2.6.2. Water-extractable ammonium

As with the KCl extraction data, there was an overall increase in NH_4^+ following heating of the MSW Fines with a small amount of variation observed between the different nutrient solution treatments. No significant differences were identified between any of the treatments here ($p > 0.05$).

6.4.2.7. Extractable P

6.4.2.7.1. Olsen P

The addition of the Sucrose + Glutamate solution increased the amount of P present; yet, adding Sucrose alone resulted in a slight drop when compared to the control material. However, no significant differences were found between any of the samples here ($p > 0.05$).

6.4.2.7.2. Water-extractable P

The changes exhibited by H_2O extractable P are quite different to those for the Olsen P data; both the Control and the material with Sucrose added show an increase in available P. Compared to this, the addition of Sucrose + Glutamate solution appeared to reduce the amount of H_2O available P. No significant differences were identified between any of the treatments here ($p > 0.05$).

6.4.2.8. Ammonium acetate extractable cations

6.4.2.8.1. Calcium (Ca)

Heating increased the extractable Ca levels within the control material, yet treating Fines with organic nutrients appeared to negate this effect. These two treatments

had Extractable Ca levels similar to the levels found in the original (untreated/unheated) Fines. No significant differences were found between any treatments ($p > 0.05$).

6.4.2.8.2. Potassium (K)

Very little change was recorded in K concentrations in any of the treatments here, with the Control and the Sucrose + Glutamate treatment falling slightly when compared to the original Fines. No significant differences between treatments were identified here ($p > 0.05$).

6.4.2.8.2. Sodium (Na)

There was a clear increase in extractable Na recorded following treatment and heating (as indicated by the statistical analysis below), with the Control and the Sucrose treatments having similar Na contents at the end of the trial period. The largest increase recorded was in the Sucrose/Glutamate treatment. Analysis showed that the Sucrose/Glutamate treatment had a higher Na content than the original Fines analysed prior to treatment and heating; this was also true for the Sucrose and the Control treatments when compared to the original Fines ($p > 0.05$ in all cases).

6.4.2.9. Water extractable cations

6.4.2.9.1. Calcium (Ca)

As for the ammonium acetate extraction, H₂O extracted Ca was seen to increase following heating in both materials that had been treated with organic nutrients. The Control material also increased, albeit by a smaller amount than the other treatments. The Sucrose + Glutamate and Sucrose treatments were found to contain significantly higher Ca levels than the original Fines following heating ($p < 0.01$ and $p < 0.05$ respectively).

6.4.2.9.2. Potassium (K)

Again, there was very little increase shown in any of the treatments compared to the readings from the original Fines. The addition of the Sucrose + Glutamate

solution did give the largest increase yet this was only slightly larger than the other treatments. No significant differences were found between any of the treatments here ($p > 0.05$).

6.4.2.9.3. Sodium (Na)

All three materials were recorded as increasing in Na levels (as with the Ammonium Acetate extraction). However, the Control material and the Sucrose + Glutamate treatment showed a greater increase than the treatment of Sucrose alone. All of the post-heating treatments were found to contain significantly more Na than the original Fines (all $p < 0.05$); as such, it can be concluded that the changes in Na are a result of heating rather than any nutrient treatment.

6.4.2.10. Compost moisture content

Moisture content did not vary greatly between any of the treatments, although when compared to the original Fines, the treatments did have greater moisture readings. This can be attributed to the addition of water to all of the treatments materials prior to heating (as part of the nutrient solutions). No significant differences were shown up using ANOVA or T-test analysis ($p > 0.05$).

6.4.2.11. Organic matter content

Organic matter (OM) levels in all three treatments increased following heating but no large scale differences were apparent. As with the majority of data recorded here, the levels of OM present did not differ significantly between treatments ($p > 0.05$).

6.4.3. Experiment 3: Inorganic nutrient limitation

6.4.3.1. Compost temperature

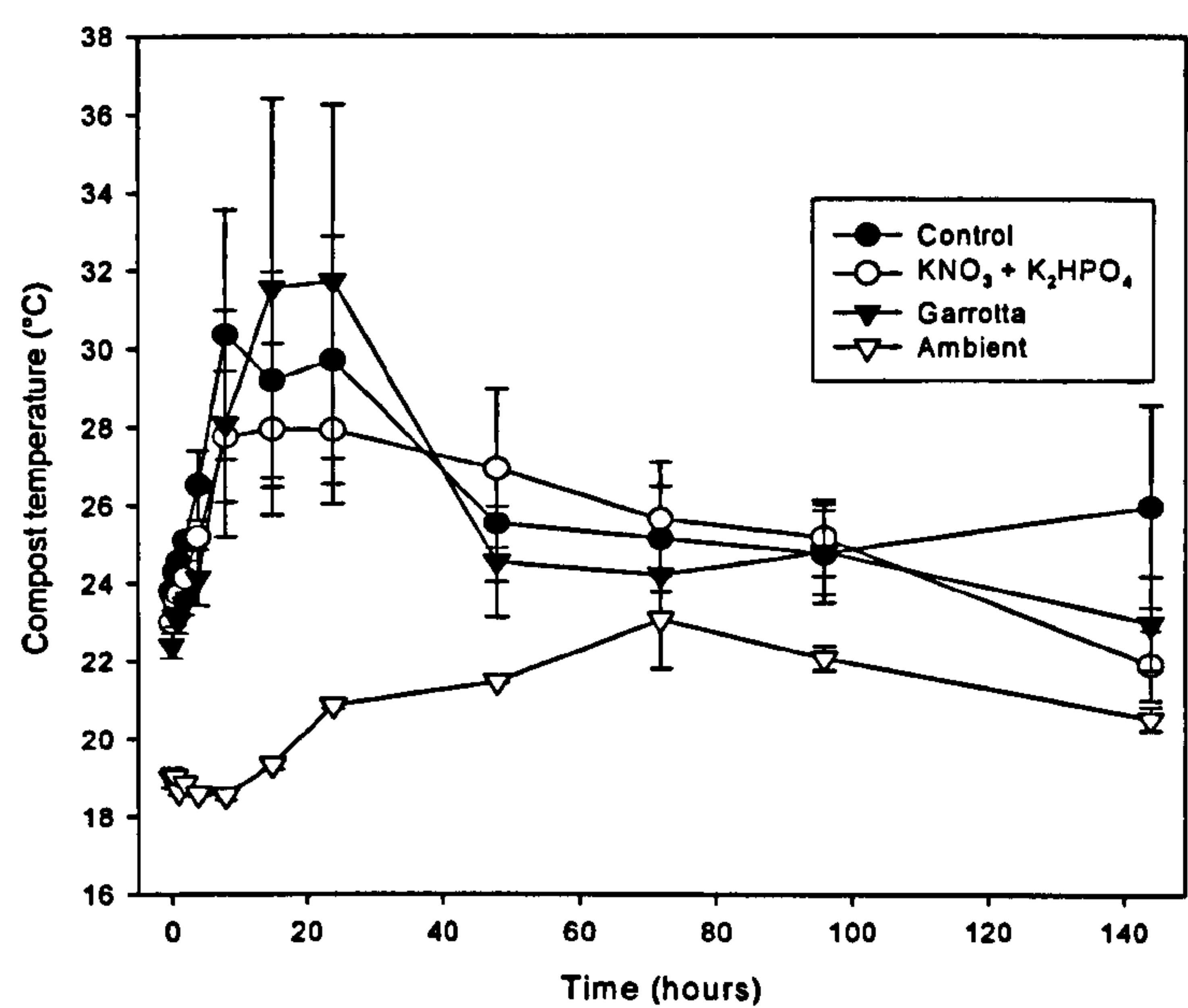


Figure 6.4: Changes in temperature in MSW Fines mixed with various inorganic nutrient solutions over a period of 144 h during a benchtop scale trial.

Temperature peaked during the first 24-36 h for all treatments, with the Garotta® treatment achieving the highest overall temperature. Addition of a N+P solution appeared to prevent the MSW Fines from reaching as high temperatures as the other materials; however, this treatment was also the one that cooled at the slowest rate. The material treated with the N+P solution had significantly higher temperatures than the Control Fines during this trial ($p < 0.05$).

6.4.3.2. Chemical analysis results

| | MSW Fines (at start $t = 0$) | Control material | Fines + inorganic nutrient N+P solution | Fines + Garotta® compost additive |
|--|----------------------------------|------------------|---|--------------------------------------|
| pH | 5.67 ± 0.30 | 5.45 ± 0.34 | 6.17 ± 0.41 | 5.64 ± 0.27 |
| EC (mS cm ⁻¹) | 3.6 ± 0.3 | 4.3 ± 0.5 | 3.2 ± 0.6 | 6.3 ± 1.0 |
| KCl NO ₃ ⁻ (mg N kg ⁻¹) | <0.01 | <0.01 | 0.03 ± 0.02 | <0.01 |
| KCl NH ₄ ⁺ (mg N kg ⁻¹) | 168 ± 30 | 226 ± 14 | 222 ± 20 | 255 ± 12 |
| H ₂ O NO ₃ ⁻ (mg N kg ⁻¹) | 0.18 ± 0.18 | <0.01 | 0.12 ± 0.12 | <0.01 |
| H ₂ O NH ₄ ⁺ (mg N kg ⁻¹) | 128 ± 39 | 239 ± 20 | 211 ± 29 | 271 ± 14 |
| Olsen P (mg kg ⁻¹) | 328 ± 21 | 157 ± 75 | 259 ± 33 | 164 ± 28 |
| H ₂ O P (mg kg ⁻¹) | 121 ± 24 | 104 ± 19 | 68 ± 15 | 89 ± 26 |
| Moisture content (%) | 59 ± 2 | 62 ± 2 | 61 ± 5 | 66 ± 1 |
| Organic matter content (%) | 57.4 ± 2.5 | 61.9 ± 5.9 | 67.6 ± 1.4 | 65.2 ± 3.8 |
| <u>Ammonium acetate extractable cations</u> | | | | |
| Ca (mg kg ⁻¹) | 1221 ± 149 | 1655 ± 80 | 1789 ± 181 | 2158 ± 195 |
| K (mg kg ⁻¹) | 3634 ± 208 | 3670 ± 328 | 4259 ± 160 | 4342 ± 230 |
| Na (mg kg ⁻¹) | 3222 ± 196 | 2860 ± 192 | 2995 ± 175 | 3189 ± 194 |
| <u>Water extractable cations</u> | | | | |
| Ca (mg kg ⁻¹) | 1144 ± 157 | 1742 ± 138 | 1388 ± 204 | 2105 ± 123 |
| K (mg kg ⁻¹) | 2869 ± 177 | 3866 ± 384 | 4456 ± 302 | 4018 ± 339 |
| Na (mg kg ⁻¹) | 2165 ± 179 | 2982 ± 249 | 3160 ± 148 | 3374 ± 197 |

Table 6.13: Chemical analysis results from the MSW compost to which inorganic N+P or Garotta® had been added at the start and after 144 h residence within a bench top heating vessel. All values represent means ± SEM

6.4.3.3. Compost pH

pH reduced very slightly in both the Control and Garotta® treatments whilst the material amended with Nitrate/Phosphate solution was the only treatment to show a rise in pH, yet this rise was only slight. No significant differences were identified between the pH readings of any of the samples here ($p > 0.05$).

6.4.3.4. Compost electrical conductivity

Addition of the Garotta® solution resulted in a large increase in EC readings ($p < 0.01$). The Control treatment was expected to show a drop in conductivity following heating, as shown in the results of other experiments, however, an increase was recorded over the period of heating. The Nitrate/Phosphate treatment was the only one to actually show a drop in EC.

6.4.3.5 Extractable nitrate

6.4.3.5.1. KCl-extractable nitrate

With the exception of the material that had KNO_3 added, all materials were recorded as having no KCl extractable NO_3^- present. No significant differences were found between any of the treatments for KCl extractable NO_3^- ($p > 0.05$).

6.4.3.5.2. Water-extractable nitrate

Although the original Fines were found to contain NO_3^- , both the Control material and that treated with Garotta® were recorded as having no extractable NO_3^- present. No significant differences were identified between any samples ($p > 0.05$).

6.4.3.6. Extractable ammonium

6.4.3.6.1. KCl-extractable ammonium

As with previous trials, the NH_4^+ levels present were found to be several times greater than the amount of NO_3^- observed in any of the treatments. The Garotta® treatment showed the greatest increase of NH_4^+ compared to the original MSW Fines. The Control material was recorded as having a greater NH_4^+ concentration than the material treated with the Nitrate/Phosphate solution. This result suggests that the extra NO_3^-

added had not been converted to NH_4^+ in the Nitrate/Phosphate treatment during the time frame of this experiment as the control had a higher NO_3^- content. Statistical analysis identified that the Garotta® treatment had a significantly higher NH_4^+ content than the original Fines in the KCl extracts, with no other significant differences being found ($p < 0.05$).

6.4.3.6.2. Water-extractable ammonium

All three treatments were found to increase in NH_4^+ concentration compared to the original Fines; again this suggests that the extra NO_3^- added had not been converted to NH_4^+ in the Nitrate/Phosphate treatment during this trial.

Statistical analysis determined that the Garotta® treatment contained significantly higher NH_4^+ than the original Fines ($p < 0.01$); no other significant differences were identified.

6.4.3.7. Extractable P

6.4.3.7.1. Olsen P

By the end of the experiment, all treatments were found to contain less P than the original Fines. Unsurprisingly, the treatment involving the Nitrate/Phosphate solution was found to have the highest amount of extractable P of the treatments but, surprisingly, this was still lower than what had been recorded in the original, unheated fines material. Analysis showed that the Garotta® treatment had significantly lower Olsen P than the heated, non-treated Fines ($p < 0.05$).

6.4.3.7.2. Water-extractable P

As with the Olsen P extractions, all three treatments were observed as having reduced P levels following heating. The material that was treated with the Nitrate/Phosphate solution was found to contain less P than both the Control and the Garotta® treated Fines in this test, unlike the Olsen P extraction. The original Fines (non-heated) were found to contain significantly higher P than the Nitrate/Phosphate treatment ($p < 0.05$); the Control (non-heated, untreated) Fines also had a higher P than the

Nitrate/Phosphate material ($p < 0.05$). These results were unexpected given that P had been added to the MSW Fines as Phosphate.

6.4.3.8. Ammonium acetate extractable cations

6.4.3.8.1. Calcium (Ca)

Extractable Ca increased in all three treatments following heating, with the Garotta® treatment giving the greatest level of increase, to nearly double that present in the original Fines ($p < 0.05$).

6.4.3.8.2. Potassium (K)

As with Ca, K was observed to increase in all three treatments following heating; the two treatments with nutrient solutions added showed higher concentrations than the Control Fines. It had been expected that the Nitrate/Phosphate treatment would have increased K as the nutrients added were in the form of Potassium salts. The Garotta® and Nitrate/Phosphate treatments were both found to contain significantly higher levels of K than (unheated) Fines ($p < 0.05$ in both cases).

6.4.3.8.3. Sodium (Na)

Na was found to decrease slightly following heating in all treatments, with the Control Fines showing the greatest drop from original levels. This result is in direct comparison to those for Ca and K, where levels of both cations rose in all treatments. No significant differences were found between any of the data obtained for the treatments analysed here ($p > 0.05$).

6.4.3.9. Water-extractable cations

6.4.3.9.1. Calcium (Ca)

As with the ammonium acetate extractions, all three treatments showed a rise in Ca following heating; the Garotta® treatment had the greatest increase. Statistical analysis showed that the Garotta® treatment contained significantly higher amounts of Ca than both the original Fines (unheated) and the material treated with the Nitrate/Phosphate solution ($p < 0.05$).

6.4.3.9.2. Potassium (K)

K increased in all three treatments following heating, with the change in the Nitrate/Phosphate solution expected due to the presence of both KNO_3 and K_2PO_4 . In statistical terms, all treatments contained significantly higher levels of K than the original Fines ($p < 0.05$ in all cases, except for the Nitrate/Phosphate treatment which was $p < 0.01$). Such a result suggests that it was the heating of the materials and not the addition of nutrient solutions that resulted in these changes.

6.4.3.9.3. Sodium (Na)

Extractable Na content was observed to increase in all three treatments, with the Garotta® treatment recording the largest rise overall. Statistical analysis showed that all three treatments had significantly more Na than the original Fines ($p < 0.05$ for the Control material and $p < 0.01$ for both the Nitrate/Phosphate and Garotta® treatments). This suggests that as for the K content, the main factor controlling the change in levels was heating of the material rather than the addition of nutrient solutions.

6.4.3.10. Moisture content

Moisture Content remained around 60-65% for all treatments, before and after heating. No significant differences were identified between treatments ($p > 0.05$).

6.4.3.11. Organic matter (OM) content

OM rose slightly in all three treatments following heating, with the Nitrate/Phosphate treatment exhibiting the highest increase. Here, only the Nitrate/Phosphate treatment was found to contain significantly more OM than the original Fines material ($p < 0.05$).

6.5. Discussion

6.5.1. Screening trial

During the course of this trial, many changes occurred with respect to the chemical composition of the MSW Fines being tested. The most obvious ones observed included the pH readings decreasing with heating, something that is not expected with large scale composting operations; Bernal and Kirchmann (1992) concluded that pH should

increase with composting as organic matter breakdown is a process that increases alkalinity. However, De Bertoldi (1983) found that during the short term, pH tends to decrease as organic matter degrades to acidic intermediate forms; due to the short length of this trial, this is likely to be what was observed. pH levels remained within the optimum range for composting for the duration of the trial; this is between 5.5 and 8.0 according to De Bertoldi *et al.* (1983). Kapetanios *et al.* (1993) state that this is the optimum range for fungal growth, whilst bacterial growth occurs best at a pH between 6.0 and 7.5.

It was noted, by the same study, that nitrification, if unbalanced by other factors, would cause the pH to fall, as observed here. However, nitrification did not appear to progress well during this experiment, as the amount of NH_4^+ present remained much higher than NO_3^- and as such, can be said not to be responsible for the pH change. pH itself is linked to the changes in state of NH_3 and is involved with a loss of this form of N at higher pHs (Kapetanios *et al.*, 1993); this also helps to explain why there is no sudden loss of NH_3 during this trial. The water extractable forms of NO_3^- and NH_4^+ decreased in content, whilst the KCl-extractable equivalents did not.

It had been expected that electrical conductivity would have decreased due to volatilization of NH_3 and soluble salts being leached to the base of the vessel. However, as mentioned above, there was no sudden loss of NH_3 , causing NH_3 to accumulate and conductivity to increase. This could have been caused by a reduced rate of airflow or increased pH (as above).

The major conclusion that can be drawn from this experiment is that bench-top scale composting does not appear to follow established rules regarding the progression of the composting process. For example, it would be expected that EC should decrease with composting time, yet over the 8 d (192 h) trialled here an increase was recorded. Overall results of this nature could be caused by several factors, such as the “freshness” of the material, the chemical composition or the presence of physical contaminants such as glass or plastics. However, the major factor in an experiment of this scale is generally the mass of the composting material; i.e. a larger composting heap provides more material and therefore tends to reach greater temperatures as more suitable nutrients are present for microbiological action (Schulze, 1962). Manios *et al.* (2007)

state that using bulking agents in windrow composting has the potential to combat the onset of anaerobic conditions within the composting material. It is interesting to note that despite this, the temperatures reached were not much lower than those in conventionally sized composting facilities.

Overall, it can be concluded that the coarsest material (50 mm screenings) did not reach initial temperatures as high as the finer matter and as a result, does not tend to show variations as large as the other samples. Both the 30 and 40 mm screenings were recorded as being at 50°C for more than 24 h, yet the former then decreased rapidly. It appears that removal of large amounts of physical contaminants, whilst physically “cleaning up” the material somewhat, does reduce the initial heating potential of MSW Fines. When bulking agents are used in larger scale trials, even in small amounts (e.g. 10% v/v), higher temperatures can be reached due to increased porosity; it was noted that the physical contaminants present in the MSW Fines used in this trial were akin to bulking agents more commonly used (e.g. green waste) (Manios, 2004; Dignac *et al.*, 2005; Manios *et al.*, 2007).

6.5.2. Organic nutrient limitation trial

Adding organic nutrients to composting materials increases the amount of substrates available to the microorganisms that are involved in the composting process (Hirsch *et al.*, 1961); low molecular weight additions are used in the earlier stages of composting. Dell’Agnola and Ferrari (1979) found that low molecular weight compounds, such as sugars, are often utilised as energy sources by the microbial population and so produce more biogenic heat; this use of such molecules in turn can lead to the accumulation of inorganic products of microbial metabolism, e.g. NO_3^- , although such a rise was not recorded here.

The inclusion of extra organic nutrients appeared to reduce the highest temperatures recorded during this trial, when compared to the previous two trials; although the second rise in temperature (recorded around 96 h) is not apparent in the 8 d (192 h) screening trial. According to Chanter and Spencer (1974), some studies found that the addition of sucrose to a composting material encouraged thermophilic bacterial growth and links to the increases in temperature. Thus, it can be concluded that adding

a solution of Sucrose allows material to heat up after several days; the fact that the Control material and ambient temperatures show no such spikes give further evidence for this theory. It is theorised that, to some extent, the presence of the low molecular weight organic nutrients (sucrose/glutamate) provides a secondary substrate for heterotrophic microorganisms (e.g. bacteria, actinomycetes and fungi) to utilise once the readily available nutrients in the MSW have been exhausted; the time lag between the two heat peaks may provide evidence for this.

The addition of Sucrose alone appears to give the greatest benefit in terms of heating and chemical change. A mixture of Sucrose and Glutamate, whilst having some benefits, seems (generally speaking) to prevent heating of the material to levels as high as for Sucrose and also affects the changes in chemical composition.

6.5.3. Inorganic nutrient trial

The main way that addition of inorganic nutrients can help to improve composting is by lowering the C/N ratio of the material, especially if readily available forms of N are added, such as NO_3^- . Several studies (Eylar and Schmidt, 1959; Hirsch *et al.*, 1961; Marshall and Alexander, 1962) have concluded that heterotrophic nitrifiers tended to be the microorganisms responsible for any nitrification that occurs during the early phases of composting, and as such, addition of NO_3^- would not have as great a benefit as it would when added at a later stage of the process. Contrary to this, the addition of small amounts of N as NO_3^- is hypothesised to provide a source of N to composting bacteria that would otherwise be lacking in this due to the inhibition of nitrification by the high temperatures associated with the early stages of composting (De Bertoldi *et al.*, 1988). Such additions would also help to reduce variation in pH levels that can be caused by the loss of N via ammonia volatilization; the changes in pH recorded here did not reflect this however. It should also be noted that the levels of NH_4^+ increased, meaning that little or none was lost by volatilization. These results could also imply that this composting process does not follow the more widely accepted progression of events.

Adding Nitrate/Phosphate appeared to halt the temperature increase in the composting material when compared to both the Control material and that treated with

Garotta®. Again, there was only a small initial increase in temperature, despite the material used being transferred to the reactors within a short period of being collected.

Apart from temperature, several factors exhibited significant differences between the different treatments following heating for 48 h. The material treated with Garotta® (a material that contains ammonium sulphate, ground limestone and lignite) showed the most differences between many factors, although as these tended to be those associated with the contents of the Garotta® (e.g. EC, Potassium and Calcium); such a trend is also apparent with the Nitrate/Phosphate treatment which shows both high Nitrate and Phosphorous contents (as expected).

6.6. Conclusions

- Many of the facets observed in this trial did not follow established rules of composting; this is most likely due to the small spatial scales of the vessels and the short nature of the trials.
- Finer MSW Fines derived material reached higher temperatures than more coarse matter, negating the idea that inert material helped to increase porosity of the MSW Fines. However, this could be due to the increased proportion of biodegradable matter present in some samples.
- Increased time between the production/collection of MSW Fines from MRFs and the start of trials can reduce the maximum temperature reached during the composting phase.
- Adding labile carbon (e.g. sucrose) to MSW Fines can produce a secondary peak in vessel temperature.
- Addition of solutions containing either Organic/Inorganic nutrients did not have any major effects upon the temperature or chemical composition of MSW Fines being composted in small scale benchtop reactors.
- Any differences that were observed tended to result from the addition of certain substances within the nutrient solutions; e.g. increased Nitrate/Phosphate content in the material treated with a Nitrate/Phosphate solution.
- Disappointingly, strongly insulated vacuum vessels poorly mimicked large in-vessel composting systems.

Further work could be used to identify the influence of these solutions over longer time periods or under conditions where larger volumes of material were used as feedstock. Adding to this, comparing screened material treated with nutrient solutions to those without could further show if this has any influence upon the composting process and the aeration of the material in a more representative composting process.

6.7. References

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General Discussion and Conclusions

7.1. Overview of thesis

As stated in Chapter 1, this thesis aimed to evaluate the potential of two in-vessel composting methods for successfully treating MSW Fines mixed with Green Waste along with evaluating the suitability of the end product for use as a growth media. Broadly speaking, the aims of this thesis were:

- 1) to review previous studies on composting, including composting of Municipal Wastes, specifically biodegradable fractions and why such systems are needed.
- 2) to further knowledge of the use of in-vessel systems for treating MSW based feedstocks and the degree to which the useful material is contaminated by inert matter such as plastic/glass.
- 3) to evaluate the changes in composition of MSW Fines taken from an MRF based system when screened with changing aperture sizes.
- 4) to draw conclusions as to what further work is needed to evaluate these systems and identify potential for improvement.

The use of biodegradable matter that originates from municipal sources as feedstock for composting has been occurring for many decades, yet no definite conclusions as to wholly suitable methods for treatment have been drawn up. As there is no completely accepted definition on the composition of MSW within the literature, any conclusions drawn by studies using this as feedstock tend to be specific to the work conducted and any differences to other studies must be acknowledged.

This thesis evaluated 8 hypotheses throughout Chapters 2-6, which are discussed in depth in the individual Chapters. All hypotheses were satisfied and in this discussion, the implications of the research conducted are considered.

7.2. Implications of research

The in-vessel composting methods employed here (Chapters 2 and 3) both successfully reached the minimum required temperatures laid down in the ABPR (2005), indicating that MSW-based materials have potential for use as compost feedstock, but only following some degree of pre-treatment to remove larger items of inert material.

Adding of a bulking agent, in this case Green Waste, was shown to increase the rate of decomposition compared to MSW alone (Chapter 2); further to this, the use of increased amounts of Green Waste in the feedstock did not appear to have any significant benefits over a smaller proportion, especially when the temperature reached during composting is considered (Chapter 3). This is a significant effect in itself, as reducing the amount of Green Waste needed to produce a suitable end-product has economic implications and would allow material to be composted during times of reduced Green Waste production (e.g. winter months) or in areas that have poorer collection facilities.

The presence of contaminants within MSW based composts is an important factor in determining the potential end uses of such materials. As such, the fact that the heavy metal content of the materials produced during the work detailed in Chapter 2 fell within the ranges stated by several other studies is encouraging. However, the composted MSW Fines had a Pb level greater than that deemed acceptable by the PAS100 Specifications, something that could have implications upon the end use; the heavy metal content of the MSW Fines + Green Waste mixture passed all requirements and so has a wider potential for use. Thus, the presence of Green Waste appears to reduce the available heavy metal content following composting.

Whereas the methods used for composting in Chapter 2 allow for increased residence time within the composting vessel, those employed in Chapter 3 tended to favour a rapid “composting” period and focussed upon treating the maximum volume within a specific time period. This itself has implications as composted matter requires as long a maturation period as can be allowed to produce a product that is low in potentially harmful substances. As such, any system will tend to have a theoretical maximum volume of treatable material, with one that utilises rapid throughput being less efficient than a “slower” system as the composting vessels will inevitably have lengthy periods of not being used due to saturation of the maturation facilities. It is noted that the composts produced using the methods studied in Chapter 3 could not be considered to have reached maturity when compared to widely accepted standards in the literature; this has the potential to impact upon the end uses of the materials as they

would be subject to the possibility of further decomposition past the time periods measured here.

Increasing the length of time that the MSW-based material matures was shown to result in increased yields of both *Lolium perenne* and *Trifolium repens*. This continued even to the point where after approximately 120 d (approx 17 wk), the *T. repens* yields recorded exceeded those of the commercial peat compost (Chapter 4). Such a result would suggest that it is both economically and environmentally favourable to allow material to undergo a lengthier maturation period. The addition of “young” materials that have not undergone a sufficient length of maturation has been noted to be one of the most common causes of reduced crop yields (Iglesias-Jiménez and Pérez García, 1989; Serra-Wittling *et al.*, 1995).

As such, in connection with the results discussed in Chapters 2 and 3, it is recommended that a more favourable, and valuable, product comes from longer composting times and lengthier maturation periods.

The end quality of compost produced from organic/biowastes is dependant upon the optimisation of many key factors during composting. Initially, a material must be well aerated to allow sufficient O₂ to reach the aerobic microbes degrading the feedstock; this is often achieved using bulking agents to increase the porosity of the material (e.g. green waste). The levels of O₂ within the composting mass are critical to the success of composting and preventing the onset of anaerobic conditions. Such conditions can result in the production of low molecular weight compounds (e.g. NH₄⁺ and low molecular weight fatty acids) which can increase the phytotoxicity of a material and also reduce the pH, causing an increase in heavy metal availability.

One other key facet of composting is the maturity/stability of the end material. A “stable” compost is one which is fully decomposed and has low rates of respiration (linked to low microbial action). A “mature compost” is where the levels of phytotoxic substances are low. Such compounds are often produced as the intermediate products of decomposition (De Bertoldi *et al.*, 1983; Tang and Millar, 1993 and Bernal *et al.*, 1998c). As such, increasing the length of the composting period allows these

compounds to be broken down and reduce potential problems with using the composts for plant growth.

Many authors have noted that plant growth trials/assays need to form an important part of analysis of a compost to determine its maturity/stability and therefore the end quality of the product (Wong *et al.*, 1981; Zucconi *et al.*, 1981 a,b; Das, 1988; Iglesias-Jiménez and Pérez García, 1992; García-Gómez *et al.*, 2003). Another important part of composting is the removal of pathogens (sterilization) caused by the heating up of the feedstock in the vessel. However, if a material is free of pathogens, the conditions in which the finished compost is subsequently stored become intrinsically important as to whether or not the material will then stay free of pathogens or become reinfected (e.g. by bird faeces if stored in the open).

Many studies (e.g. De Bertoldi *et al.*, 1983; Senesi, 1989 and Kulhman, 1990) state that the C/N ratio of a feedstock plays an important role in determining the successful progression of composting and hence the end quality of the product. Studies, including those referenced above, state that a C/N ratio of around 20-25 is suitable for allowing composting to progress acceptably. However, those materials with C/N ratios of >35 often compost too slowly to retain sufficient heat to eliminate pathogens, whilst a value of <10 promotes the loss of N via volatilization, reducing the potential for the material to be successfully used as a plant growth media. With MSW-based composts, there is also the presence of inert materials (e.g. glass/plastic) which do not degrade within the timeframes of composting and therefore can cause a reduction in the “visual quality” of a material and have the potential to cause physical harm to humans or animals depending on use.

Combining MSW-based composts with agricultural soils increases yield of ryegrass (*Lolium perenne*), which identifies land disposal as a possible end use of composted MSW. However, addition of such material to agricultural soil reduced the yield of White Clover (*Trifolium repens*). From this, it is concluded that the effect of compost addition is species specific as found in Zucconi *et al.* (1981a,b) and Van Assche and Uyttenbroeck (1982); expanded testing could be utilised to confirm theory.

The type of land to which the material is disposed, however, depends upon the level of physical and chemical contamination; especially if the material is destined for use that may result in contamination of potential food stuffs (Purves and Mackenzie, 1973). Despite a small degree of physical contamination, these materials could possibly be utilised as soil replacements for construction or remediation schemes where the presence of glass shards, plastic pieces or other contaminants is not as much of a hazard to human/environmental health. This is possibly one end use for material that does not meet the requirements of the relevant legislation.

As composition of both the parent material and the composted product plays a key role in the potential for use and also in how the material degrades during composting, the physical composition of material taken from an MRF was analysed. The composition of wastes vary with a number of factors across the globe, with composition often being highly dependant upon the economic status and consumption and disposal habits of the waste producer (Daskalopoulos *et al.*, 1998; Gidarakos *et al.*, 2006; Hansen *et al.*, 2007).

The wastes analysed here (Chapter 5) contained around 60% biodegradable matter (by dry weight using a weight-based method), a figure consistent with other studies conducted in the UK (Welsh Assembly Government, 2002; Burnley *et al.*, 2007); this would suggest that this material has potential for use as compost feedstock. However, when the proportion of glass present was recorded, this was found to be around 23% in a material screened to 50 mm, a figure that could pose problems with potential end uses. Employing screens smaller than 50 mm reduced the amount of physical contaminants within the material to around 8.5% of the sample (when screened to 10 mm). Reducing the screen size to 10 mm also removed around half of the biodegradable material illustrating that removal of contaminants is not simply a case of using smaller apertures on the screen.

By comparison, using an area-based method the figures for paper and plastic were increased to over 21% and 10% respectively. This difference to the figures from the weight-based method was attributed to the high surface area to volume ratio for these materials and having a lower bulk density than the biodegradable matter. The presence of higher amounts of plastic, a factor accentuated by the brighter colours of

this fraction compared to biodegradable matter, resulted in a material that is likely to appear disproportionately contaminated compared to when analysed on a dry weight basis. As such, any future analysis of composition should include both methods and formulate tolerances for each material.

In economic terms, the potential to increase the amount of recyclable material recovered from the MSW Fines stream would also increase the economic potential of such ventures, if a suitable method of materials recovery could be identified.

Reducing the amount of inert material present within a sample is one way to improve the end quality of material derived from the MSW stream. Such a process was employed to evaluate the potential for allowing higher temperatures to be reached when material was placed in a small scale insulated vessel. There appeared to be a loose connection between the particle size of the material (decreased size resulted in higher temperatures); this nullified the idea that the presence of larger pieces of inert matter help to increase aeration of this material during composting. The addition of organic or inorganic nutrient solutions has no real impact upon materials being treated in small scale vessels, with the exception of labile C (such as Sucrose) which can cause a second temperature peak following initial increase.

7.3. Further work

Several areas for potential future work exist as a result of the findings of Chapters 2-6. Despite the wide ranging studies that have been conducted on the composting of MSW, there is a clear need to carry out trials in line with up to date legislation including the 2005 Animal By-Products Regulations amongst others. Future composting studies should also include evaluation of MSW Fines taken from several MRFs and homogenised to reduce the potential variation in composition of materials between sites. The composition of material taken from several sites would require evaluation to determine the degree of variation present between producers.

Composting trials utilising other in-vessel methods could be conducted to highlight any variations between methods both in operational terms and in terms of the quality of the end product when compared to those carried out here (Chapters 2 and 3).

As composting is seen as a growing industry, the economics of composting require evaluation to determine where savings can be made to increase the economic potential of such operations. The efficiency of MSW composting also requires substantial investigation as part of the economic evaluation. This would also provide an insight into the amount of energy present in such material and if it had a high enough calorific value to be used as a fuel for incineration.

To expand upon the effect of adding compost to land, growth trials should be conducted both under greenhouse and field conditions to test the effects of land spreading of such materials and expand upon the findings of Chapter 4.

Identification of potential markets and end uses for the products of composting and how these are influenced by the composition of the feedstock, the method of composting used and composition of the end product is required; without suitable end uses, the economic value of composted products would be vastly reduced.

To further this work, studies into the public perception of MSW-based composts compared to more traditional materials such as peat compost would provide an insight into their commercial viability and what the public feel is required to allow widespread use of this.

7.4. Summary

MSW Fines contains a high enough proportion of biodegradable matter to make it potentially suitable for use as a feedstock for in-vessel composting and allowing it to reach sanitisation temperatures required by legislation ($>60^{\circ}\text{C}$). However, this material is highly variable and can vary in composition between different sites, even within samples taken at the same facility at different times of the day. Therefore, it is important to know the composition of the material prior to use as this can affect the potential end uses for the end product.

Future work should take into account the fact that during the studies carried out for this thesis, it was noted that the MSW Fines material being produced was highly

heterogeneous. As a result, obtaining representative samples of the material was difficult to achieve. This was also hampered by the variations in the parent MSW collections, the composition of which is ultimately influenced by the disposal habits of the collection locations on a particular day. As such, to obtain representative samples for future studies, as much material should be retrieved and homogenised as far as practicable. For larger scale composting trials, it would be beneficial to utilise multiple composting vessels to attempt to minimise the effect of variation in compost feedstock composition. However, it is likely that due to limitations (both practical and financial), variation between samples will always occur and will therefore need to be acknowledged with any studies such as those conducted here.

Co-composting MSW Fines with Green Waste produces a compost which is more favourable to potential users, with a material containing 30% Green Waste producing higher yields of common pasture land species than one with 50% Green Waste. The length of the composting period can affect the quality of the end product, especially if the maturation time is shortened. This can result in a poor quality compost containing high levels of phytotoxic substances which reduce the potential for use as growth media.

Screening of MSW Fines to remove physical contaminants is successful to some degree; however, reducing the screen size can remove some of the biodegradable matter required for composting. Therefore, more complex systems and changes in waste producer habits are more likely to increase materials recovery and produce a more desirable compost feedstock.

Adding nutrient solutions to try and provide “boosts” to the system has no real effect upon the changes occurring during composting. However, screening of materials to remove inert contaminants can, potentially, lead to higher temperatures being reached by the composting material.

As the waste management landscape continues to change with greater public awareness of environmental issues, changing legislation and the need for disposal to become more sustainable, composting of biodegradable matter will become more prevalent. Yet the processes employed for such endeavours must evolve to meet the requirements of the producer and, more importantly, end user of the compost.

7.5 References

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