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An evaluation of the physical and biochemical characteristics of green waste compost in relation to end use and quality

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AN EVALUATION OF THE PHYSICAL AND BIOCHEMICAL CHARACTERISTICS OF GREEN WASTE COMPOST IN RELATION TO END USE AND QUALITY

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ABSTRACT

Biodegradable waste represents an ongoing challenge for modern societies and developing cost effective, safe and sustainable alternatives to landfills are essential. The implementation of the European Landfill Directive in the UK has meant that there are clear targets for reducing the levels of biodegradable wastes sent to landfill. Consequently, there has been a sharp rise in the number of composting facilities in the UK, as composting is seen as a more sustainable management option for many biodegradable wastes. Green waste represents the most commonly composted waste stream in the UK and as part of a drive to conserve peat habitats, developing high quality peat replacements in horticulture represents a potential end market for these composts.

The effect of sterilizing the compost to improve plant growth and establishment was studied (chapter 3). A reduction in phytotoxicity as a result of sterilization meant the composts were comparable with the peat-based equivalents in the germination and plant growth trial, compared with the unsterilized equivalents. Compost maturity represents an ongoing challenge with regard to assuring the quality of composts. The results from an earlier trial showed that *in vitro* compost tea disease suppression of three fungal plant pathogens occurred from a tea produced using immature compost (Appendix 1). Coupled with the rising interest in using composts and compost teas to suppress plant diseases we studied the potentially phytotoxic and allelopathic effects of compost teas (chapter 4). Four economically important plants were used in a germination bioassay to assess the phytotoxic and allelopathic conditions of mature and immature compost teas. Results showed that plant response was highly variable and/or species specific, inferring that the PAS: 100 quality assurance test of phytotoxicity in solid compost could be insufficient owing to only one plant species being used. Vermicomposting green waste is regarded as a way of adding value to the green waste compost and many studies have reported improved plant growth for vermicomposts. Chapter 5 addressed the lack of research on the water characteristics and physical stability of vermicomposts compared with standard green waste-derived compost and peat. Vermicompost was less hydrophobic at low moisture contents, compared with peat and green waste compost. The feedstock used to produce the vermicompost resulted in very different responses to drying and wetting, with paper pulp amended vermicompost shrinking irreversibly following drying. Using surfactants to alleviate the hydrophobicity of containerised media was studied in chapter 6. Surfactants applied to peat and green waste compost successfully reduced hydrophobicity. This work and chapter 5 indicated that highly organic media represent a challenge when applying the theoretical and analytical methods developed for mineral soils for defining physical properties.

In summary, this work has demonstrated that green waste-derived compost is highly heterogeneous but that there are options for improving the end quality and creating high quality peat replacements. Chapters 3, 4 & 5 indicated that the PAS: 100 Protocol for quality assurance has some potential shortcomings with regard to operational regulations and the parameter and tests used to provide quality assurance.

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ABBREVIATIONS

ABA	Phytohormone abscisic acid
ACT	Aerated compost tea
ARH	Ambient relative humidity
BSI	British Standards Institution
CC	Container capacity
CFU	Colony-forming unit
CMC	Critical micelle concentration
DEFRA	Department of Environment and Rural Affairs
DOC	Dissolved organic carbon
EC	Electrical conductivity
EU	European Union
ESF	European Social Fund
FC	Field capacity
GI	Germination Index
IPCC	Intergovernmental Panel on Climate Change
ITC	Allyl-isothiocyanite
MGT	Mean germination time
MPN	Most probable number
NCT	Non-aerated compost tea
NVZ	Nitrate vulnerable zone
OM	Organic matter
PAS: 100	The Publicly Available Specification 100 (BSI PAS 100) for composted materials was sponsored by WRAP and developed in conjunction with the Association for Organics Recycling.
PDA	Potato-dextrose agar
PTE	Potentially toxic element
RV	Radicle vigour
SEM	Standard Error of Mean

SME	Small to Medium Enterprise
UV	Ultra violet
VFA	Volatile fatty acids
WDPT	Water Drop Penetration Time
WEOC	Water extractable organic carbon
WR	Water repellency
WRAP	Waste Resources Action Programme
WRC	Water release curve
WSOC	Water soluble organic carbon

Experimental Treatments Abbreviations

Chapter Three

GW	Green waste compost
GW AC	Green waste compost - Autoclaved
GW M	Green waste compost - Microwaved
GW NS	Green waste compost – Non-sterilized
P	Peat-based (benchmark) control
PF	Peat-free (benchmark) control

Chapter Four

A	Autoclaved
CTu	Compost tea (unstable compost preparation)
CTs	Compost tea (stable compost preparation)
CTsF	Compost tea (stable) Filtered
CTsM	Compost tea (stable) Microfiltered
CTsA	Compost tea (stable) Autoclaved
CTuA	Compost tea (unstable) Autoclaved
CTuF	Compost tea (unstable) Filtered
CTuM	Compost tea (unstable) Microfiltered
F	Filtered
M	Microfiltered

NS	Non-sterilized seed
SS	Sterilized seed

Chapter 5

V	Vermicompost
G	Aerobically composted green waste
P	Peat horticultural compost

Chapter 6

UF	Ultraflow non-ionic surfactant
FN	50:90 non-ionic surfactant

Appendix 1

CTa	Autoclaved compost tea - sterile
CTf	Filtered compost tea – retaining its microbial biomass
CTm	Microfiltered compost tea - sterile

CHAPTER 1: INTRODUCTION

1.1 General introduction and the need for research

The UK has seen a rapid shift in the way waste is managed over the last decade. Throughout the EU greater concern about the impact of waste on wider environmental functioning has resulted in the ratification of several laws governing the disposal of different waste streams throughout member states. The EU Landfill Directive (EC, 1999) sets out clear targets for the diversion of biodegradable waste from landfill sites. In landfills biodegradation tends to occur in anaerobic conditions resulting in the emission of methane – a potent greenhouse gas (Faviono & Hogg, 2008). The ratification of the Landfill Directive (EC, 1999) into UK law has resulted in a sharp rise in composting facilities, owing to the view that this was a low carbon alternative to landfilling biodegradable waste.

In the waste hierarchy (DEFRA, 2007), composting equates to recycling (reduce, reuse, recycle). Aerobic composting is the biological decomposition of organic matter, resulting in an end product (compost) that is rich in humic substances and plant nutrients (Gajdos, 1997; Liang *et al.*, 2003). The main revenue for composting sites still comes from the gate fee, funded by Local Authorities for diverting waste from landfills. Consequently, there has been an initial focus by many composting operations to maximise the volume of waste entering the site and less attention has been paid to the quality of the end compost.

Green and woody waste is widely considered to be the least hazardous fraction of the biodegradable waste stream (Smith *et al.*, 2008). Therefore, the initial rise in composting operations in the UK were predominantly licensed to compost only green and woody waste, owing to the comparatively relaxed control in terms of planning, regulation and monitoring (Smith, 2009) and availability of the feedstock. Despite the availability and low cost of green waste-derived compost, consumers have been reluctant to purchase it for horticultural purposes, preferring reliable peat-based products (WRAP, 2008). Natural peat-based compost typically has a low pH and nutrient content, meaning that producers can readily adjust these properties to suit the end use (Schimileski, 2008). Added to this, the homogenous structure and low bulk density makes it cheap to transport and easier to handle in large-scale horticulture (WRAP, 2008; Owen, 2005) with the price of peat being non-prohibitive to consumers (Jones *et al.*, 2009). Consequently, the bulk of green waste-derived compost produced tends to be sold as low grade soil improvers or even as landfill cover. In order for the industry to become more economically resilient a diversity of compost end markets are required.

This need for a greater diversity of end markets coincided with the raised awareness of the wider ecosystem services delivered by peat habitats (Barkham, 1993). One of the primary reasons for the destruction of peat habitats in the UK and Ireland has been the extraction of peat for use in horticulture (Tomlinson, 2010). Not only was the consideration of the quasi-irreversible damage to peat habitats paramount but also the loss of carbon associated with the extraction of peat for fuel and horticulture (Tomlinson, 2010; Parish *et al.*, 2008). With stringent targets for phasing out the use of peat in horticulture (Wallace *et al.*, 2005), creating high quality composts from waste has increasingly become as important as simply diverting biodegradable waste from landfills (Faviono & Hogg, 2008). In 2005, the revised PAS: 100 Quality Assurance Protocol (BSI, 2005) for the specification of composted materials was introduced as an optional industry standard for those aiming to develop a range of end markets for their composts. This stimulated further interest in developing high quality peat replacements for use as containerised media, as these are typically marketed at higher values. Soilless plant growth media are less resistant to changes in temperature, water content and solute concentration, than field soil, owing to the limited volume of substrate in containers (Michel, 2010; Naasz *et al.*, 2005) and there is a paucity of comparative studies looking at the impact of wetting and drying on containerised peat-based and waste-derived composts. This was particularly the case for physical properties.

Vermicomposting has recently received increasing research interest, as an additional method for improving the end quality of waste-derived composts (Roberts *et al.*, 2007). Despite a growing body of research examining the bio-chemical properties of vermicompost in relation to plant growth, no studies were available concerning the physical and water characteristics of vermicomposts as plant growth media. Finally, at the outset of this project the industry and research communities were very interested in the possibilities of using waste-derived compost as plant disease suppressants. Composts typically made from animal wastes had begun to be studied in relation to bio-control but there were very few available for green waste compost, both as soil amendments or as compost teas (Noble *et al.*, 2006).

1.2 Plan of the thesis

This research was funded by the European Social Fund (ESF) and Wormtech Ltd. This company was active in two key areas, in-vessel aerobic composting of green waste and the secondary treatment of compost via vermicomposting. The ESF funding was specifically aimed to twin researchers with small to medium enterprises

(SMEs) in Objective 1 areas, to stimulate an increase in knowledge and understanding in a relevant area that would provide economic benefits and operational solutions for the company in question.

This thesis consists of seven chapters; following the introduction is a review of the relevant literature, looking broadly at research relating to green waste compost quality. Typical barriers to quality assurance, such as compost maturity and stability when producing saleable growing media were evaluated in chapter 2. Studies on the disease suppression potential of waste-derived composts and compost tea were also reviewed. The literature review ends by addressing relevant research in the specific areas targeted in this thesis for improving the end quality of the compost. These include vermicomposting partially composted green waste to assess whether this was a superior growing media compared with green waste compost, the relative merits and methods for sterilising compost and using surfactants to alleviate hydrophobicity. Owing to the novel focus of these topics in terms of composting science, some of the literature reviewed pertains more to the field of soil science and so its applicability is limited in this regard.

Chapter 3 examines the effect of sterilizing composts prior to plant growth following some positive results from preliminary studies conducted by Bangor and Cardiff University researchers. Wormtech Ltd had initiated the operational regime required to acquire the PAS: 100 quality assurance. Owing to the increased manpower required to fulfil the level of compost turning that the protocol demanded, the company were interested in researching alternative methods for assuring the sanitisation of the compost. Regulators proposed that turning ensured even high temperatures throughout the compost heap in order to ensure sanitisation of the compost during the thermophilic phase.

Chapter 4 initially focuses on assessing the disease suppressant potential of compost tea made from unstable green waste compost. The potentially phytotoxic effects of compost teas produced from unstable and stable green wastes to determine the safety of using these, as either a soil or foliar amendment. Phytotoxic conditions were assessed via a germination bioassay. This focus was owing to studies reporting favourable disease suppression in compost tea produced from both unstable and stable compost.

Chapter 5 reports the findings from an extensive study, examining the water characteristics and physical stability of standard green waste compost and vermicompost following drying and rewetting. In this study physical stability is taken as meaning compost resilience to shrink and swell during drying and wetting. The water repellency of composts was seen by the company as a particular problem when marketing the compost as containerised media, where these effects were typically amplified.

Chapter 6 examines the effect of using non-ionic surfactants on the compost water repellency and any concurrent effects the surfactants had on plant growth and microbial activity.

Chapter 7 presents the general discussion based on the findings from the experiments and suggests areas for further research.

1.3 Aim and Objectives

The aims and objectives were agreed with the funding company during the first few months of the research period. These were:

- To assess the impact of sterilizing green waste composts prior to plant growth, using autoclaving and microwaving. The hypothesis being that via the destruction of the compost biota both plant germination and growth would be improved due to reduced competition for nutrients, air and water. Added to this, the potential transformation of potentially allelopathic chemicals incurred by the heat during sterilization could result in a reduction of phytotoxic conditions, frequently found in waste-derived composts.

- Using an *in vitro* microbial bioassay the level of plant disease suppression from compost tea made from unstable green waste compost was assessed. The issue of using unstable and stable compost for the production of compost teas was further examined. We used a germination bioassay to assess the level of allelopathic and phytotoxic effects of compost teas from unstable and stable green waste composts on four plant species.
- To assess whether vermicompost water characteristics and physical properties were more closely related to peat composts compared to standard green waste compost. The influence of 3 different feedstocks (green waste, sewage sludge and paper pulp) on the end quality of vermicompost was also examined in relation to the wetting and drying effects. There was a particular focus on the level of shrinkage and re-wettability (including water repellency) of the composts after increasing levels of drying.
- The use of surfactants to reduce hydrophobicity in green waste and peat composts was assessed.

CHAPTER 2: LITERATURE REVIEW

2.1 Climate Change, Soils and Waste Management

The Intergovernmental Panel on Climate Change (IPCC; Houghton *et al.*, 1995) estimated that 20% of all methane (CH_4) emissions arise from landfills (20 – 70 Tg year^{-1}). CH_4 has a greater irradiative forcing potential than CO_2 and has greater impact in terms of global warming (Galle *et al.*, 2001). In 2005 biodegradable waste was estimated to emit 1300 Mt CO_2-eq in 2005 (IPCC, 2007). The EU Landfill Directive (1999/31/EC) sets out clear targets for the diversion of biodegradable waste from landfill, with the specific aims of both reducing the production of CH_4 and the volumes of waste sent to landfill.

In order to stabilise global warming by reducing carbon emissions, enhancing the carbon sink capacity of soils (Mondini *et al.*, 2008) has become another primary focus. The EU are in the process of ratifying a Thematic Strategy for Soil Protection and in the meantime the Environment Agency has assessed soils in England and Wales (EA, 2004) resulting in the first Soil Action Plan (DEFRA, 2004) which has been adopted in England but nothing similar has been done for Wales as yet. The solutions for global warming, the remediation of soil health and the management of biodegradable waste are inextricably linked. By incorporating organic matter into soils, the water and nutrient holding capacity is increased as the physical structure is improved (Bastida *et al.*, 2007). Important elements are released during the biodegradation of the organic matter by macro- and micro-organisms and organic matter has been shown to bind potentially toxic elements (PTE's; Alloway, 1990). Appropriate levels of organic additions can reduce the need for artificial fertilisers, and irrigation and potentially increase soil carbon sequestration (Favoino & Hogg, 2008; Mondini *et al.,* 2008; Schaefer *et al.*, 2009). However, there is still a lack of knowledge about the impact of climate change on soil carbon and the important role soils play as a source and a sink for carbon (EA, 2004). Increasingly, research is examining the residence time of organic compounds and the stability of organic matter in soils in the light of climate change (Brookes *et al.*, 2008; Crow *et al.*, 2009).

Over the next century scientists predict that soil moisture will decrease in most regions and in the subtropics and the Mediterranean soil moisture could decrease by 25 % (Meehl *et al.*, 2007). Drought is also a phenomenon predicted to rise as the climate changes, meaning that soil organisms and plants are likely to be subject to periods of enhanced stress through water shortages. Mitigating soil water stress is becoming an important research consideration and increasingly the use of waste-derived organic amendments is seen as part of the solution. Approximately 70% of water withdrawals throughout the world are accounted for by irrigation in agriculture (IPCC, 2007).

Coastal, river and flash flooding are becoming more frequent in the UK and the risk and intensity of flooding are predicted to rise as the climate change advances (Mokrech *et al.*, 2008). Many studies have shown that organic amendments to the soil can improve soil structure, aggregate stability, increase rainfall infiltration which can then reduce surface run-off, sediment yield and soil erosion (Sun *et al.,* 1995; Albiach *et al.*, 2001; Fristensky & Grismer, 2009). However as the broad research focus was creating high quality growth media from green waste, the extensive literature available concerning organic waste inputs to land was not reviewed here.

2.2 Peat & Horticulture

2.2.1 Peat Excavation

The horticultural industry in the UK is growing and Plassman & Edward-Jones (2007) estimate that it is currently worth £350 million in Wales alone. The need to produce alternatives to peat for use as a growing media has become increasingly important in terms of climate change mitigation (Favoino & Hogg, 2008). The degradation of peat lands has resulted in approximately 3000 million tonnes of anthropogenic greenhouse gas emissions, which equates to more than 10 % of global fossil fuel emissions (Parish *et al.*, 2008). In the UK, this rapid degradation of lowland bogs can be attributed to the drainage of sites for agriculture in the 1950s (Alexander, 2008) and the excavation of peat for the horticultural industry (Tomlinson, 2010). Since the 1970's loam in potting media, which was difficult to source was substituted by peat and as demand for containerised media grew, so did the commercial extraction of peat (Alexander, 2008).

In 1990 members from 10 organisations (Table 1) formed a voluntary group dedicated to preventing any further exploitation of peat habitats in the UK by launching the Peat Campaign 1990 (Barkham, 1993). Scientists and environmentalists argued that the continued extraction and drainage of peat lands was highly exploitative and unsustainable.

Table 2.1 The ten member organizations of the consortium responsible for the Peat Campaign (Barkham, 1993)

Peat Campaign Consortium
British Association of Nature Conservationists
British Dragonfly Society
Friends of the Earth
Greenland White-Fronted Goose Study
Irish Peatland Conservation Council
Plantlife
RSNC The Wildlife Trusts Partnership
Royal Society for the Protection of Birds
Wildfowl and Wetlands Trust
World Wide Fund for Nature (UK)

Barkham (1993) identified the value of peat habitats:

- Wilderness – an increasingly valued ecosystem service for the well-being of people on a health and spiritual level. Equally these areas of low populations provide a valuable refuge for rare species
- Biodiversity – peat habitats support a unique assemblage of plants and animals adapted to living in conditions that are 98 % water
- Biological Indicators – peat bogs can act as highly sensitive indicators of environmental change, a factor that is becoming increasingly important in understanding the impact of climate change

- Genetic Resource – the characteristic conditions of peatlands has meant that species have evolved distinctly and as such can provide valuable information on functions such as protein digestion
- Hydrological Systems – peat bogs are natural filters and water stores, supplying clean water to many organisms and in periods or low rainfall bogs maintain their own water supplying streams and rivers
- Carbon Sinks – Peat lands constitute the most important long term carbon stock
- Carbon Storage – Although peat lands cover only 3 % of the earth's surface, they constitute the greatest terrestrial sink for carbon, holding twice as much as all global forest biomass (Parish *et al.*, 2008)

2.2.2 *Peat as a growing media*

The extent to which peat has been used as an organic raw material in fertilisers for agriculture and horticulture has varied around the world and tends to be higher in regions where natural peat lands are local (Holmes, 2000). The key properties of peat that make it such an appealing growing media are:

- The structure ensures a high water holding capacity and air content
- The low pH and nutrient status means peat can be manually adjusted to suit the target plants
- It is generally free from weed propagules and pathogens and safe to handle
- Relatively homogenous physical properties make it easy to grade and specify
- Low bulk density – cheap transportation and plant despatch
- Widely available and currently cheap to purchase

Adapted from: Anon (2004); Joosten & Clarke (2002); Robertson, (1993)

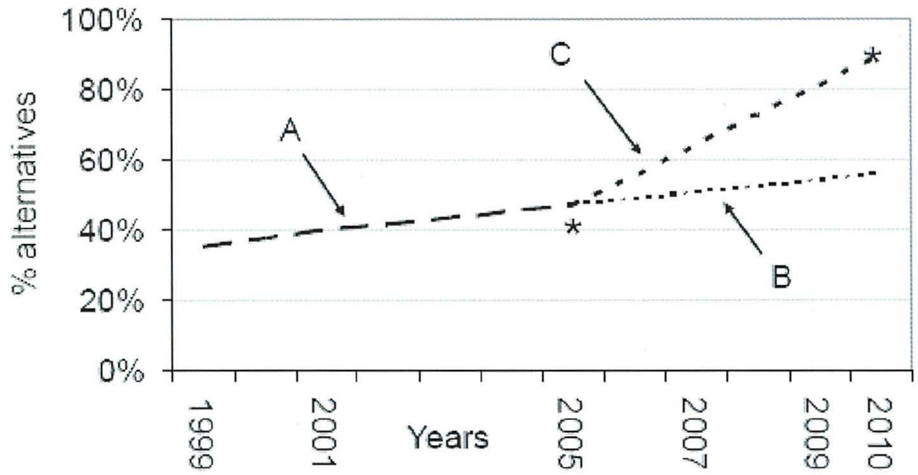
2.2.3 *Peat Replacement*

The horticultural industry depends upon high quality growing media and a peat substrate tends to assure a consistency in quality, along with a competitive price when compared with the non-peat counterparts. There have been a growing number of peat replacements for the horticultural industry that have had variable success both in plant growth trials and with consumers (Alexander, 2008). There is evidence to suggest that most alternatives benefit from some amendment with peat, which is not ideal in terms of sustainability (Holmes, 2000; Schmilewski, 2008). For large-scale horticulturalists the low bulk density of peat keeps transportation and handling costs down, providing another barrier to switching to waste derived alternatives that tend to have a higher bulk density than peat, with less consistent physical properties (Holmes, 2000; Anon, 2004). The mineral content of most waste-derived composts tends to be >50% and this means the bulk density is higher than that of peat (Schmilewski, 2008).

In 2006/07 3.6 million tonnes of greenwaste was source-segregated and composted in the UK. However, with only 542 000 tonnes achieving the PAS: 100 quality assurance (BSI, 2005), only 7 % of the source-segregated feedstock was sold for use as growing media (Smith *et al.*, 2008). This percentage has fallen from 29% in 2004-05, indicating that the market for growing media still remains largely untapped, with peat as the main constituent (Favoino & Hogg; 2008). The drop in sales is most likely due to the introduction of the PAS: 100 (BSI, 2005) resulting in a delay in marketing compost as growing media while companies acquired the standards required for quality assurance. The operational regulations and end quality of compost produced by companies opting to adopt the PAS: 100 standard (BSI, 2005) depends upon the specified end use for the compost. Several different analyses are required, with upper and lower limits of acceptability for each analysis being clearly defined in the PAS: 100 (e.g. physical contaminants, PTEs and human pathogens). Should a consecutive series of three batches not pass each parameter, then, the compost retains its label of 'waste' (Smith *et al.*, 2008; Nicols, 2008). The range of analyses required depends entirely upon the specified end use but there is increasing concern about the requirement for only one sample to be used per test (WRAP, 2009b) in terms of the reliability of the results.

In the *State of Composting in the UK* (Smith et al., 2008) the composting industry was consulted concerning which sectors they felt had the most market potential for waste-derived compost. More than half replied that the agricultural market was the most ripe for development with landscaping close behind. Agriculture and landscaping are currently the largest markets for composted wastes and the added attraction is that these sectors tend to allow operators to shift large volumes via a single customer. However, 49% of the composting industry felt that the professional and amateur horticultural sector also held great market potential but there is a degree of incongruity between the aims of the sector and the regulations which safeguard human, animal and plant health, thus causing this sector to be a more difficult undertaking (Smith *et al.,* 2008; Walker *et al.*, 2006; Rainbow & Wilson, 1998). This is largely due to the expertise, man power and finance required to maintain a consistently high quality product. There is also the need to convince consumers of the validity of peat replacements particularly when derived from waste (Wallace *et al.*, 2005).The amateur horticultural market utilises the largest volume of peat (66% of total volume used; (Wallace *et al.*, 2005) and so finding high quality alternatives is even more important with regard to protecting peat habitats. Peat is most commonly used as a growing media and so the market potential for waste derived composts is very promising and indeed this represents the greatest added value market for compost operators (WRAP, 2008).

Peatering Out (Anon., 2003) lays out the targets for peat replacement in the UK. It plans a steady phasing-out of peat over a 10 year timescale and this is in line with the Government's Habitat Action Plan for lowland raised bogs (UK Biodiversity Plan, 1999). As part of the Biodiversity Action Programme the UK government has set out clear targets for the reduction of peat in horticultural materials. Reaching the 2005 target was successful with 40% of all market requirements fulfilled via non-peat based materials. By 2010 the target is for this figure to be 90% (Figure 2.1) (Wallace *et al.*, 2005)



* BAP targets (40% peat alternatives in 2005, 90% in 2010)
A. Trend in proportion of alternatives to 2005.
B. Indication of trend in the future, based on extrapolation assuming the constant rate of change in increasing the proportion of alternatives shown in A.
C. Required trend in the future, based on a constant, but **greater** rate of change from 2005 onwards that is required to meet the BAP target in 2010

Figure 2.1 The proportion of peat alternatives in line with the Biodiversity Action Plan targets up to 2010 (Wallace *et al.*, 2005)

Peatering Out advises a phase of progressive dilution of peat products with sustainable alternatives. Since the introduction of the Landfill Directive (1999/31/EC), the availability of peat alternatives have grown, and there has been a concurrent research focus comparing the merits of these alternatives with peat. Research has shown that green waste compost has had variable success in matching plant growth in peat-based media but it is clear that a well managed composting operation can produce alternatives to peat that are successful containerised media and soil improvers (Favoino & Hogg, 2008).

2.3 Green waste Compost

Green waste compost is the most abundant waste-derived compost available for sale in the UK (Smith *et al.*, 2008). According to the most recent survey on the state of composting in the UK (Smith *et al.*, 2008) approximately 75% of the waste composted was the green and woody fraction and the highest volume is generated from the general public dropping it off at civic amenity sites. 73% of the compost produced from source segregated waste in 2006/07 was used as soil conditioner, with only 7% supplying the growing media or peat replacement market. The Habitats Action Plan for lowland peat bogs and The Peat Campaign have laid ambitious targets for diluting peat products with alternatives and see the growing volume of greenwaste compost as a major part of the solution (Anon., 2003). Operators in 2007 viewed this sector as the market as having the greatest potential for development.

2.3.1. Compost Markets

Developing a diversity of end markets for biowaste-derived composts is important for the composting industry to become widely sustainable (Dimambro *et* *al.*, 2007; Slater *et al.*, 2005) but establishing bulk and niche compost markets is an ongoing issue for the composting industry in the UK. Hogg *et al.* (2002) identified that by specifying the end uses for compost, then operators could work towards creating a variety of end products to suit the demand. Equally there needs to be a clear marketing strategy that is harmonised at least within a country but there is scope for EU standards that would facilitate product recognition. The Quality Protocol (discussed in section 2.3.2) aims to increase market confidence in the quality of waste-derived compost in the UK (WRAP, 2007). Improving end quality and consistency is pivotal to achieving their sale in horticulture with the industry aim that quality assurance will abate consumer reluctance to switch from peat products to waste-derived composts. Added to this Jones *et al.* (2009) identified that there is currently a lack of financial impetus for consumers to switch from the peat based growing media to green waste ones. On face value, peat is more expensive (ca. €30 – 45 t^{-1}) than green waste compost (ca €12 – 20 t^{-1}) but once the bulk density is considered, green waste bulk density being typically much higher than peat, then the prices are more comparable on a volume basis (peat = €12 m^3 and green waste compost €10 m^3).

2.3.2 Green waste Quality

Initially maximising the rate and volume of waste composted are the primary economic drivers for most composting operations (Veekan *et al.*, 2005) as the gate fee received for the waste (by mass) is the primary source of revenue. Consequently the tendency has been for low quality composts to be sold in bulk, in low value markets, or even given away e.g. as landfill cover or on-site soil improvers (Hauke *et al.*, 1996; Veekan *et al.*, 2005; Walker *et al.*, 2006; Farrell & Jones, 2009). However, as the production of green waste compost has increased, there has been a greater focus on selling the end compost in a variety of markets. The economic driver to develop high quality composts with specified end uses has meant that EU countries realised a protocol for minimum quality was required to foster the production of consistently, high quality compost and increase consumer confidence (Probert *et al.*, 2006). Developing compost for use as a growth media required higher standards than those ideal as a soil improver for example. Hogg *et al.* (2002) identified that compost quality standards developed differently across countries and within local authorities, owing to different political and industry drivers. The common aims indentified in this study (Hogg *et al.*, 2002) as the most likely to increase market confidence were:

1. Production of quality products to specific standards
2. Facilitate marketing through the use of symbols on the product
3. Tailor products for specific end-uses

Producers and consumers identified a suite of problems with green waste compost that have been relatively universal from site to site and the main ones have been summarised in Table 2.2. In the UK the PAS 100:2005 quality assurance scheme (BSI, 2005) aims to ensure that compost from a specific site and operation is of consistent quality, whilst specifying the appropriate end use for the compost. Operators need to comply with stringent operational procedures (e.g. frequency of aeration) and for the end compost to pass a minimum standard in a number of categories illustrated in Table 2.2.

The main aim of the Quality Protocol was to increase market confidence in the quality of waste-derived compost and to ensure that the compost posed no risks to human, animal or plant health (WRAP, 2007). If operators in the UK comply with the Quality Protocol (WRAP, 2007) then this is considered to be sufficient to ensure that the compost is safe for the environment and performs well depending on the specified end use. Once an operator has become certified then the composting end product no longer requires waste management regulation. In contrast, if compost has not been approved via the PAS: 100 scheme, then the end product retains its classification as waste, incurring all kinds of repercussions to both the operator trying to market it and the purchaser wanting to use it (Hogg *et al.*, 2002; WRAP, 2007; Nichol, 2008).

Table 2.2 Green waste derived compost quality issues when considering its efficacy as a growing media. Adapted from: Rainbow & Wilson (1998) & BSI (2005)

Quality parameter	Main effect	PAS: 100 test available?
Physical Contaminants (e.g. glass, plastics, rocks)	Health and safety of consumer and quality of growth media.	√
Particle size distribution	Typically too coarse - reduces efficacy as plant growth media	√
Moisture content	> 50% (w/w) – affects stability (microbial activity).Too dense for packaging and transportation. < 20% w/w – dry and dusty, prone to becoming persistently hydrophobic	√
Bulk density	Too high – inhibits purchase particularly by professional horticulturalists	X
Sanitisation	Incomplete eradication human, animal and plant pathogens and weed seeds and propagules poses health and safety risk and reduces efficacy as a plant growth media	√
Stability	Unstable compost can result in changes to key physico-chemical and biological parameters during storage and use, often leading to phytotoxic conditions	√
Phytotoxicity	Compost physico-chemical conditions can limit or prevent plant growth in some plant species	√
Hydrophobicity	Reduces wettability and therefore efficacy as a plant growth media	X
Plant available nutrients	Can be either too low (e.g. N, P) or too high (e.g. K) limiting plant growth	√
pH	Typically too high resulting in phytotoxic conditions for some plant species	√
EC	Typically too high resulting in phytotoxic conditions for some plant species	√

Attaining consistently high quality compost and fulfilling all of the requirements that enable sites to hold PAS: 100 certification is fundamental if producers want to supply the high value markets such as growing media (Ward, 2005). Producing high quality peat replacements from the increasing volume of green waste composts would establish a sustainable end market for the composting industry (Faviono & Hogg, 2008). As yet the scheme has not been sufficient to abate consumer reticence to use biowaste-derived potting composts and soil improvers and there is debate concerning the evidence base for some of the guidelines (Nicols, 2008; Westerman & Bicudo, 2005; Lasaridi *et al*, 2006). Probert et al. (2005) identified the important issue that consumers may indicate during a survey that being 'green' was an important influence in their buying but this is not reflected in the reality of the sales figures. Walker *et al.* (2006) support these findings reiterating that the majority of compost purchase is in bulk for use as soil improvers and that a quality protocol will not convince consumers to trust waste derived composts as plant growth media. Overall, studies indicate that price is the most important factor determining purchasing behaviour and owing to the cost of gaining and retaining PAS: 100 status for a site, then quality assured composts tend to cost more than peat based products (Probert *et al.*, 2006). The following section explores the broad categories associated with greenwaste compost quality, relevant to the thesis and any ramifications of the PAS 100 protocol are also considered.

2.3.2.1 Feedstock variation

The lack of consistency in green waste quality has often been attributed to feedstock supply and seasonal variations (Ward, 2005). Hoekstra *et al.* (2002) found that four different feeding strategies for cattle resulted in quite different manures, with vastly inconsistent levels of phytotoxicity, indicating how variation in the end compost is affected at stages well before the actual composting operation. More than most biodegradable waste streams, the green waste fraction of the waste stream is subject to considerable seasonal variation both in terms of volume and quality (Ward, 2005). Vegetation entering the waste stream is more abundant in the summer months whereas the winter months see a sharp decline in the availability of green waste and a proportional increase in material high in lignin. Winter garden trimmings have much higher C: N ratio as the material tends to be woodier, containing a higher percentage of lignin compounds compared with the more labile substrates in summer feedstock (e.g. grass cuttings) (Ward, 2005). The rate and extent of decomposition is affected by the ratio of C: N, with optimal C: N being approx. 25 – 35 for the initial greenwaste feedstock (Komilis & Ham, 2003). However mixing waste streams to achieve a good C: N has become more difficult as co-composting regulations have made this much less flexible for operators and most wastes approved for composting must be source segregated (WRAP, 2007). Research has shown that mixing certain waste steams can enhance compost market value (Garcia-Gomez *et al.*, 2002; Jones *et al.*, 2009). Co-composting green waste with food waste ensures lower C: N that is more conducive to aerobic composting (Smith *et al.*, 2008) Food waste is not subject to so much seasonal variation and has a C: N as low as 2:1 and moisture contents in excess of 90% (Adhikari *et al.,* 2008). Mixing the two waste streams alleviates the high dry matter content of ligneous materials in the green waste feedstock, providing the microbial biomass with some labile carbon sources thus facilitating the rate of decomposition. This operation though, would be subject to greater regulations laid out in the waste licence agreement, as food waste is regulated by the Animal By-products Order (Anon, 2002) as it poses greater potential risks to human and animal health. However, the variation in both feedstock characteristics and seasonal supply will continue to pose problems for operators but more tightly controlled and consistent composting processes have been shown to significantly reduce any variation in the end product (Ward, 2005).

2.3.2.2 Potentially Toxic Elements (PTE)

PTEs in the PAS: 100 regulations (BSI, 2005) are listed as cadmium, chromium, copper, lead, mercury, nickel and zinc. Compost is increasingly used as an organic amendment for soils and so monitoring levels of PTEs is important, particularly if they are to be applied to food crops (Deportés *et al.*, 1995). The concentration of PTEs generally increases during composting due to mass loss via carbon and water emitted with microbial respiration (Smith, 2009). However, this does not necessarily mean that the metals become more available; rather, formation of recalcitrant humic carbon causes the elements to bind strongly to organic compounds, ultimately making them less available (Farrell & Jones, 2009). Raw waste materials, or feedstocks, containing PTEs tend to be mixed waste streams, like municipal solid waste (MSW), sewage sludge or hazardous wastes (Richard, 1992) both of which are outside the remit for this study. Green waste compost typically contains low concentrations of PTEs, that fall within the permitted range stipulated in the PAS: 100 (BSI, 2005). For this reason it was perceived as an unnecessary research focus.

2.3.2.3 Plant and microbial available nutrients

Research has long identified that green waste compost can end up with either too much or too little plant macro and micro nutrients. The main influences on the level of available N in finished compost are the initial characteristics of the parent material or feedstock (Cambardella *et al.*, 2003; Ward, 2005) and the composting process itself (Kronër *et al.*, 2003). Increasing the frequency of turning events can lead to losses in N – increased turning results in losses of NH_4^+-N through volatilisation as NH_3 and of NO_3^--N via leachate losses (Parkinson *et al.*, 2004). On the whole, green waste compost tends to be low in NO_3^- but many studies have shown this can be rectified by the addition of mineral fertilisers or by co-composting with more N - rich feedstocks such as manures and food wastes (Smith *et al.*, 2008). A well managed curing phase is very important for the generation of NO_3 from NH_4^+, however, as most operations are limited for space this phase can often be significantly shortened (Farrell & Jones, 2009). Peat is also typically low in available N and has generally always been amended with mineral fertilisers. P is typically low in greenwaste composts (Jones *et al.*, 2009) and this is highly dependent on the parent material but as with N, the operation can influence the end concentrations of phosphates (Felton *et al.*, 2004). Phosphorus is most available between pH 5.5 and 7.0 (Englestad & Terman, 1980) and green waste compost tends to have a pH higher than pH 7.0 (Jones *et al.*, 2009). Most studies continue to focus upon the impact on P availability with compost applications to soil (Takeda *et al.*, 2009; Speir *et al.*, 2004). Uygur & Karabatak (2009) found that increasing the application rate of composts to soil fostered the retention of P in more labile fractions for greater periods of time, causing overall increase in P availability in the soil. As with N, the general trend has been to amend growing media with inorganic P fertilisers.

Soluble salts, as represented by electrical conductivity (EC), are typically high in end composts derived from waste (Jones *et al.*, 2009). During the composting operation, as nitrification increases pH falls due to the release of H^+ ions and soluble salt content (EC) rises (Sanchez-Monedero *et al.*, 2001). The rapid biodegradation incurred by the aerobic composting process increases the release of inorganic nutrients via the oxidation of organic matter (Bernal *et al.*, 1998; Farrell & Jones, 2009). Potassium availability is typically high in green waste composts, often at levels that would be phytotoxic to most plants (Ward, 2005). Wet sieving the end compost is a low-tech way of reducing soluble salt concentrations (Veekan *et al.*, 2005) but this could also result in the loss of NO_3^-, which considering NO_3^- is typically low in green waste compost (Sanchez-Mondero *et al.*, 2001) means that this method for remediating high salt content could result in the need for N addition.

2.3.2.4 Compost Sanitisation

A variety of disease-causing organisms have been isolated in composts including viruses, bacteria, fungi, protozoa, nematodes and helminths (Christensen *et al.*, 2002). A plethora of research has demonstrated that the thermophilic phase of composting, where high temperatures are generated via microbial activity, can result in the eradication or significant reduction of plant and animal (including human) pathogens (Ghaly *et al.*, 2006: Hassen *et al.*, 2001; Lung *et al.*, 2001; Turner, 2002) providing the basis for the operational regulations for sanitisation of biowaste (BSI, 2005). Following the outbreak of the Foot & Mouth disease in the UK, further measures were taken to safeguard the health of wildlife and livestock by imposing increased regulation on the composting of waste containing - or that has been in contact with - animal by-products (meat, milk etc.) (DEFRA, 2008). Along with setting clear temperature requirements, the frequency of turning – largely to aerate but also to mix the compost so that any effects of the spatial effects in the heap are remediated - and moisture content were also seen as important parameters to ensure sanitisation is achieved during composting. The PAS: 100 (BSI, 2005) lays out the minimal operational requirements for sanitisation (Table 2.3). For feedstock containing animal by-products a site specific regime is devised by the Environment Agency and State Veterinary Service (Nichols, 2008) which generally incurs greater input by the operator.

Table 2.3 PAS: 100 Regime for eradication of selected pathogens during sanitization Source: PAS: 100 (BSI, 2005)

Temperature	Time	Moisture	Turning (aeration)
$\geq$ 65 °C	7 days	$\geq$ 50 % mass/mass	$\geq$ 2 times

Composts are highly bioactive even after regulatory temperatures are reached during the thermophilic phase of aerobic composting (Burge *et al.*, 1987; Casadei *et al.*, 2001; Williamson *et al.*, 2006; Christensen *et al.*, 2002; Grewal *et al*, 2007). Research has shown that pathogens have also been known to re-grow or recolonize the compost in this second mesophilic phase and during maturation (Burge *et al.*, 1987; Williamson *et al.*, 2006 and Christensen *et al.*, 2002). Williamson *et al.* (2006) analysed compost for *Salmonella* sp. and *E. coli* Spp., some species of which are pathogens known to be a significant human health hazard. The results demonstrated that the green waste feedstock, despite reaching the PAS100 critical limits (2 days at > 60 °C or 1 hour at > 70 °C; (BSI, 2005)) exceeded the critical limits of 1000 CFU g^{-1} for *E. coli* and absence in 25 g for *Salmonella* sp. at the end of the composting process. It was not clear whether this was bacterial re-growth or re-colonisation. Sidhu *et al.* (2001) state that the complete eradication of pathogens rarely occurs during composting and the potential for regrowth during storage is high. Regrowth of pathogens can occur as long as the moisture content, bioavailable nutrients, temperature and the indigenous microflora are favourable and the deactivation of salmonella via the indigenous microflora declined with storage and time. Casadei *et al.* (2001) demonstrated that the tolerance of heat resistant bacteria to temperature was affected by environmental conditions such as pH, water activity and salt content. Considering the variation in the initial characteristics of the feedstock, a generic upper temperature reading as the key parameter for assuring sanitisation during composting, is arguably insufficient (Bernal *et al.,* 1998).

Some research has indicated that high temperatures may not be enough to ensure sanitisation, when the heterogeneous nature of compost means that factors such as pH and soluble salt concentration can vary within a single heap, having a significant impact on the effectiveness of temperature as the main parameter for sanitisation. Droffner *et al.* (1995) demonstrated that temperature alone is not sufficient a mechanism to explain the sanitisation process in composting. De Bertoli *et al.* (1983) investigated the impact of high dry matter content resulting in physically stable but highly biologically unstable compost. Low water availability is more typical when compost is produced from highly ligneous feedstock and this can ultimately mean that microbial activity becomes limited. Therefore the material undergoes incomplete decomposition and is therefore biologically unstable. As the green waste feedstock is prone to containing highly ligneous material this study is of particular relevance and maintaining adequate moisture levels is important. The elimination of pathogens in aerobic composting depends upon biological activity (Christensen *et al.,* 2002) and so without consistent monitoring of other physico-chemical factors apart from temperature, during composting it is not clear whether the sanitisation procedure has really occurred throughout the heap. If anaerobic conditions develop in parts of the compost heap, the minimum temperature is unlikely to be reached meaning that parts of the heap will not have been sanitised. Even in research where compost sanitisation is achieved, recommendations are made concerning the risks of re-infection via poor management practice or natural re-colonization (Déportes, 1998). Grewal *et al.* (2007). Van Rijn *et al.* (2007) and Sidhu *et al.* (2001) showed storage method to have a significant effect on the persistence of microbes in compost. This study showed that the re-growth potential of *Salmonella* spp increased with compost storage time, as the antagonistic effect of indigenous compost microbes decreased over time. Grewal (2007) showed that both *Salmonella sp.* and *Listeria sp.* were most successfully destroyed by thermophilic composting. However, low level (<1000 MPN g^{-1}) of persistence of *Listeria sp.* occurred for the greatest period of time in composts compared with liquid manures. Low temperature composting and pack storage resulted in the longest persistence in both pathogens.

During composting most operators keep the aerated heaps at a moisture content $\geq$ 50 % mass/mass (BSI, 2005) to ensure the eradication of selected pathogens during sanitisation. Once the heaps are left to cool and mature, due to issues such as space, the compost is often left to cool and mature outside and is no longer managed. The process of drying and then rewetting causes the mineralization of dead cells creating a readily available nutrient source for microbes, which results in an initial steep rise in microbial activity (van Rijn, 2007). If the compost develops dry or cold spots during the thermophilic phase then sanitisation would not be uniform and it is likely that once the compost has been rewet after drying out during maturation that any low levels of pathogens can rapidly multiply. Lasaridi *et al.* (2006) found *Staphylococcus aureus* and *Clostridium perfringens* in 17 and 96% of composts respectively and they found that despite reaching the required thermophilic temperatures, pathogen indicator microorganisms were present at levels above suggested limits in all their composts. Some studies have also shown that nematodes and oligochaetes can act as vectors for pathogens and that the transmission of the pathogens to the surface and tissues of salad foods grown in infected composts or soils is also possible (Solomon *et al.*, 2002; Kenney *et al.*, 2006; Williams *et al.*, 2006). For this reason, ensuring that the composts are free from harmful pathogens is important to horticulturalists using the composts for salad crops.

The research presented has shown that the regulations for sanitisation set out in the PAS: 100 (BSI, 2005) may not be sufficient in guaranteeing the removal of all harmful organisms and as one of the most monitored aspects of a composting operation this throws doubt on the efficacy of these operational controls in the quality protocol. Certified composts have to pass microbiological tests for *Salmonella* and *E. coli* only and weed germination tests if they are to be used as growing media (BSI, 2005) but this is generally one sample taken as representative for the whole batch. Added to this, the composting operation is not carbon neutral and achieving quality assurance for sanitisation potentially leads to the greatest input of fossil-fuel based machinery and manpower on top of which, turning compost can result in high C and N greenhouse gas emissions as the anaerobic patches in the heap are disturbed (Fukumoto *et al.,* 2003). It is even more imperative therefore that the turning regime required for PAS: 100 certification, are sufficiently evidence-based..

2.3.2.5 Compost stability and maturity

Compost maturity and stability are possibly the most controversial aspects of compost quality, as there are still no universally agreed indices for their assessment (Gazi *et al.*, 2007; Mangkoedihardjo, 2006; Wang *et al.*, 2004). Furthermore a variety of studies and international protocols (including the PAS: 100, BSI, 2005) use the terms, maturity and stability synonymously with the caveat that maturity is related to phytotoxicity in the glossary. Maturity is frequently related to a variety of different parameters, which makes comparisons between studies on this topic difficult. Both terms describe the stage of decomposition and the conversion of organic matter into humic substances and are generally considered as a function of composting time (Zmora-Nahum, 2005). Compost stability is used by most studies to describe the rate of organic matter decomposition and is expressed by the level of microbial activity in moist, well aerated compost (Komilis & Tziouvaras, 2009). For this reason, stability is most commonly assessed by the respiration rate or the self-heating ability of the compost in standard conditions (Brinton *et al.*, 1995; Lasardi & Stentiford, 1998).

Compost maturity is actually more synonymous with phytotoxicity. Compost maturity is broadly used to describe the quality of the compost in direct relation to plant growth and is increasingly assessed with the specific end uses in mind (BSI, 2005; Zmora-Nahum *et al.*, 2005). If the intended end use for greenwaste compost is a growing medium then achieving maturity is of particular importance (Ward, 2005). Typically maturity is assessed via germination and plant growth response tests, with poor performance indicative of immaturity or phytotoxicity (Komilis & Tziouvaras, 2009). Mature compost typically has lower levels of volatile organic acids, which are often phytotoxic and obtaining mature compost, as with stability, is in part a function of time (Said-Pullicino *et al.*, 2007). However, stable compost does not necessarily imply that the same compost will have reached maturity (Wang *et al.*, 2004).

From a social and economic perspective a key factor influencing stability and maturity is the propensity for operators to circumvent the tertiary phase (the maturation of curing stage) of the operation. This tendency can be explained by the industry aim to maximise the turnover of waste, due to the economic reliance on the gate fee and the regulatory limits on the quantity of waste permitted on site at any one time (Farrell & Jones, 2009; Veekan *et al.*, 2005). In order for the biologically available C to be transformed into stable humic substances by the actinomycetes and fungi, adequate biodegradation is required during this tertiary curing phase.

Immature compost can be detrimental to root growth, by limiting oxygen content in the rhizosphere (Ward, 2005). One common cause of poor germination rates and low yield in waste derived media is owing to high pH and high salt concentrations (EC), particularly K (Ward, 2005). EC in well cured compost is usually most influenced by the initial characteristics of the feedstock (Ward, 2004) and no amount of decomposition time will remediate this parameter. However, Veekan *et al.* (2005) found that wet sieving biowaste prior to composting successfully lowered the electrical conductivity and pH of the end composts, to non-phytotoxic levels. The composting process itself also exerts an influence on properties like pH and EC (Said-Pullicino *et al.*, 2007). Another important point to consider is that plant response in any bioassay is highly species specific, meaning that maturity is difficult to standardise from compost to compost and in relation to different plants (Francou *et al.*, 2005; Zmora-Nahum *et al.*, 2005). Importantly, maturity also infers that the thermophilic phase of the composting operation has sanitised the compost as pathogens and weed propagules are destroyed by the heat generated via microbial metabolism (Bernal *et al.*, 1998).

In the UK composting industry, in an attempt to provide a quality assurance to consumers, the PAS: 100 Quality Assurance Scheme (BSI, 2005) was developed by the Composting Association (now known as Organics Recycling Association). Here, the accepted method for determining compost stability is the level of CO_2 evolution from the volatile solids component in the compost (Llewellyn, 2005). This stability standard assumes that by reaching a level of biological stability, as indicated by low levels of respiration (< 16 mg CO_2 g^{-1} VS d^{-1}), then the compost will no longer undergo any significant biological and physico-chemical changes in the phase between storage and use by the consumer. In the PAS: 100 protocol, the term stability is used interchangeably with maturity (BS1, 2005), making comparisons with non-UK standards confusing. In order to assess phytotoxicity of composts the PAS: 100 employs a germination trial using tomatoes but at no point does it refer to this bioassay as an indicator of maturity. Despite this accepted standard by the UK industry, research has shown that a single parameter and the upper limit set by the PAS: 100 (BSI, 2005) may not be a sufficient indicator of stability or maturity (Wang *et al.*, 2004; Mangkoedihardjo, 2006).

For the past ten years the main proposed indices for indicating the level of compost maturity have been:

- C : N
- Inorganic N dynamics (ratio of NH_4^+ : NO_3^-)
- Germination rate
- Organic C
- DOC content – rate of mineralization
- Hydrophilic water-extractable organic carbon (WEOC) : hydrophobic WEOC
- Cellulose: lignin ratio

(Sanchez-Monedero *et al.*, 1998; Komilis & Ham, 2003; Wang *et al.*, 2004; Zmora-nahum *et al.*, 2005; Mangkoedihardjo, 2006; Said-Pullicino *et al.*, 2007)

Typically, mature compost has much lower concentrations of NH_4^+ compared with NO_3^- but for greenwaste, if there is a higher proportion of woody materials in the feedstock, then NO_3^- levels will be lowered (Ward, 2004). During the early stages of composting organic fatty acids are produced accounting for the initial drop in pH and these organic compounds can be highly phytotoxic and so indicative of immaturity (Eklind & Kirchmann, 2000). Dissolved organic carbon (DOC) is the most active fraction of compost both biologically and chemically and so monitoring the changes occurring in this organic fraction can directly reflect the level of transformation of the organic matter (Said-Pullicino *et al.*, 2007). Aslam *et al.* (2008) have developed models linking carbon mineralization rate with phytotoxicity. This study aimed to design a model capable of advising users of the appropriate amendment rate to remediate any phytotoxic effects in relation to the level of carbon mineralization, which is generally is positively correlated to composting time. Said-Pullicino *et al.* (2007) found that with time, the ratio of hydrophilic to hydrophobic compounds in the composting material decreased to the stage where there were more hydrophobic substances in the water-extractable organic carbon (WEOC) than hydrophilic compounds, which were prolific in the raw materials with the more labile compounds still present. The authors suggested that the negative correlation between the concentration of hydrophobic compounds in the WEOC and compost maturity (as assessed by bioassay) means that this could be used as a predictive test for maturity.

2.3.2.6 Physical Properties

The physical properties of compost have received comparatively less research attention than the biology and chemistry of the composting process and the final resulting product. This is exemplified by current UK compost standards (BSI, 2005) which don't specify minimum limits for properties such as water availability, porosity and hydraulic conductivity. Of the research that has been undertaken, most has focused on the physical properties of feedstocks prior to composting, with the aim of optimising the efficiency of the composting process (Ahn *et al.*, 2008; Mohee & Mudhoo, 2005). In terms of compost application to soil, again the prevalent focus has tended to be on biological and chemical effects of the amendment (Strauss, 2001), although an improvement of soil structure is widely accepted despite the mechanisms only just becoming apparent (Abiven *et al.*, 2009). Soil structural stability and strength is of paramount importance in soil productivity and declining soil structure is increasingly being recognised as a limitation in crop production (Albiach *et al.*, 2001; Ostle *et al.*, 2009).

2.3.2.6.1 Influence of organic matter inputs on soil physical properties

Soils are increasingly valued for the pivotal role they play in food and water supply and delivery of other ecosystem goods and services (Haygarth & Ritz, 2009). The productivity of soil is defined by the complex interactions of biological, chemical and physical processes (Johnston, 2006). The generally positive impacts of organic amendments on the soil physico-chemical properties, such as nutrient balance, buffer capacity or pH are well acknowledged (Farrell & Jones, 2009) and compost is widely reputed to benefit soil physical properties, such as improved soil structure and aggregate stability (Abiven *et al.*, 2009), increased water infiltration and retention (Zhang *et al.*, 1997), increased porosity (Ahn *et al.*, 2008) and lowering the bulk density (Khaleel *et al.*, 1987). The improvements in soil structure following organic matter amendment are largely attributed to the impact on aggregate stability (Weinfurtner, 2001).

There is also the potential for organic matter (OM) inputs to sequester carbon in the soil with appropriate land management (Ostle *et al.*, 2009; Porter *et al.*, 2009).

Haygarth & Ritz (2009) argue that as agro-ecosystems account for such large areas of land globally (27-38%) there is an urgent need to examine the ecosystem services they provide and work towards carbon neutral farming, where carbon released is off-set by sequestering carbon (Figure 2.4). The capacity of any soil to store carbon is linked to the soil structure and the range of particle sizes (Feeney *et al.*, 2006). Organic amendment via raw and composted wastes can play a valuable part in maintaining soil health. However, for the purposes of this thesis, the main focus has been on creating high value potting composts for the industry partner (Wormtech Ltd), and as such the effects of composts on soil properties are not covered in detail here.

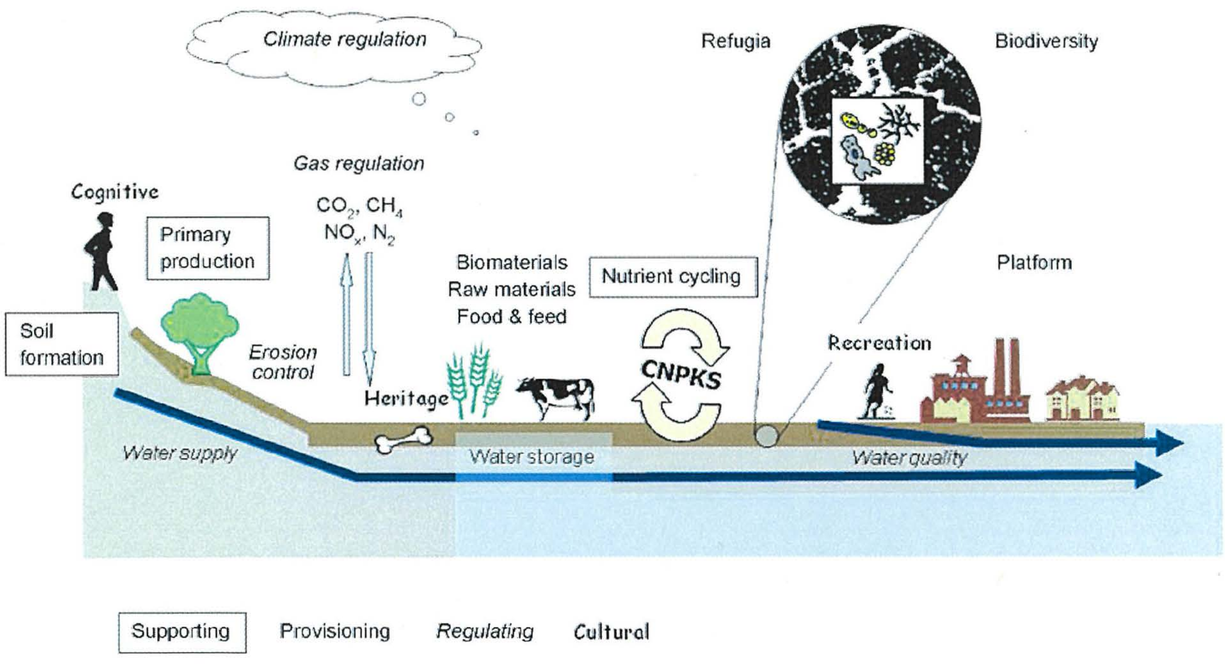


Figure 2.2 Categorisation and nature of the key ecosystem goods and services provided by soil systems (Haygarth & Ritz, 2009)

2.3.2.6.2 Green waste compost physical properties compared with peat

Peat is the most widely used component of containerised media and its specific properties are unlikely to apply to any other growing media (Boudreau *et al.*, 2009) owing to the unique anaerobic conditions in which it is slowly formed. Green waste-derived compost typically has a higher bulk density than horticultural peat. Papafotiou *et al.* (2005) found that even substituting peat with 25% of olive-waste compost resulted in a significant rise in the bulk density. Where the amendment rate was increased to 50–75% (v/v) there was a considerable decrease in porosity and water holding capacity. Clemmensen (2004) found that four different green waste-derived composts held significantly less water at the same matric potentials compared with peat. Despite the high water holding capacity associated with peat composts, its tendency to waterlog, resulting in a anaerobic conditions does not always make it ideal for use as containerised media (Clemmensen, 2004) without some additional material such as vermiculite or waste-derived composts. Caron *et al.* (2005) state that horticultural peat formed by decomposing Sphagnum moss typically has better hydraulic conductivity and capillary rise properties compared with sedge peat. This is due to the unique structure of hollow vessels and pores in the partially decomposed Sphagnum effectively storing and transporting water.

Large-scale horticulturalists continue to be reluctant to switch to green waste-derived compost because of the inconsistencies in bulk density. Green waste-derived composts in particular often possess a higher mineral fraction than peat (often as high as 60%, compared with peats 2%) leading to a higher bulk density (Schmilewski, 2008). This increases transportation costs and can cause handling problems for mechanised seedling nurseries, increasing reluctance for consumers to switch from peat-based composts. However, Clemmensen (2004) found the green and woody waste composts had a lower bulk density and higher porosity than the peat composts, owing to smaller particle sizes in the composts.

Noguera *et al.* (2000) reported very positive results when comparing peat with coconut coir compost, with regard to properties such as porosity and water-holding capacity. However, there was a significant difference in both physical and chemical properties between different batches of coir. Therefore, the inherent heterogeneity of the feedstock and the end compost continues to be a problem for the composting industry when attempting to develop high quality end products (Ward, 2005). Owing to the very different degradation processes, peat, and waste-derived compost vary in the proportions of their carbon fractions, helping to explain differences in behaviour between the media (Eklind & Kirchmann, 2000). Eklind & Kirchmann, (2000) demonstrated that the initial lignin content is highly correlated with the residual levels of organic C in the end compost and so the feedstock exerts an influence on C dynamics as well as the decomposition process (Ward, 2005).

2.3.2.6.3 Water Characteristics of Composts

Water scarcity is globally becoming more widespread, so finding ways of reducing its use in agriculture and horticulture is advantageous (Bastida *et al.*, 2007). Approximately 70% of water withdrawals throughout the world are accounted for by crop irrigation (IPCC, 2007). A deeper understanding of water dynamics can significantly enhance our knowledge of the plant-soil system in relation to irrigation and nutrient transport and is of particular importance in soilless media (Naasz *et al.,* 2005). This is particularly relevant to containerised media which frequently incur cycles of watering and drying owing to the limited amount of substrate in the pot, implying the need for careful water management (Naasz *et al.*, 2005). Blending peat with waste-derived composts tends to result in a reduction of water holding capacity (Medina *et al.*, 2009; Sanchez-Monedero *et al.*, 2004) which means that finding peat replacements that can mimic the high water holding capacity and hydraulic conductivity of peat is desirable (Caron *et al.*, 2005). Organic matter is an important determinant for the formation and stability of micro and mesopores, ultimately influencing the behaviour of water in the matrix (Ahn *et al.*, 2008). There is evidence to suggest that the organic matter (OM) present in the micropore space (< 5 μm diameter) aids a strong retention of water and the water offers physical protection from decay (Zhaung *et al.*, 2008).

Water availability measured via pressure potential provides a clearer idea of the how tightly the water is held in the compost and ultimately how much water is available to the plant (Wallach & Raviv, 2005) compared with gravimetric water content which is the most common water measurement when assessing compost quality (BSI, 2005). Compost maturity can exert an influence on the water characteristics, as can the properties of the initial feedstock. Carmona *et al.* (2003) demonstrated that industrial compost derived from cork residue and matured for 7 months, resulted in significantly higher water retention than the raw equivalent. Michel *et al.* (2002) showed fairly key distinctions between water retention and structure in less decomposed peat compared with well decomposed peat. The less decomposed peat exhibited less shrinkage during desiccation and was more likely to rewet to the same water content. Earlier research though has frequently reported the benefits to soil hydrology following the additions of raw organic wastes (Khaleel, 1981), however, current legislation makes this increasingly difficult (due to the potential risk of introducing noxious weeds, and plant and animal pathogens).

Most of the research looking at the hydrology of highly organic materials or soils arises from the discipline of soil science where in situ measures are made, or otherwise largely undisturbed soil cores are analysed. There is therefore the question of how transferable the findings are to horticultural peat for example, which is more comparable to compost in terms of having been screened and amended for use as a plant growth media.

2.3.2.6.4 Hydrophobicity/ Water repellency

Research into hydrophobicity or water repellency (WR) of soils has seen an exponential rise in interest over the past decade (Dekker, 2005), owing to an increasing realisation that water repellent soils are common. Understanding the effects of WR on soil physics and other vital soil functions is seen as increasingly important due to the rise in ambient temperatures and decreasing rainfall which is exacerbating the incidence of WR (Dekker, 2005; Doerr & Ristsema, 2005). Another area where WR has received attention is following forest fires, where soils can become severely water repellent (Mataix-Solera *et al.*, 2008; Vadilonga *et al.*, 2008). The following section provides an overview of the main effects of hydrophobicity at the landscape scale (as this is where most of the research is conducted/funded), followed by an examination of the mechanisms of WR currently proposed and finishes with an overview of the main method for assessing WR employed. Typically, little has been done on the WR of organic soils and composts, although Dekker (1996), did look at both peaty-clay and clayey-peat soils in Holland and found that the dry surface of the peat soils was persistently hydrophobic and that in many cases this extended to the subsoil as well. Strong WR typically only becomes manifest after peat soils have become very dry, similar to that occurring with peat and other composts used in horticulture (Michel *et al.*, 2002; Rainbow & Wilson, 1998).

The temporal and spatial heterogeneity of WR makes it very hard to predict the hydrological responses to WR in any given landscape (Leighton-Boyce *et al.*, 2005). As rain droplets hit the soil surface, the infiltration rate is mainly impacted by the persistency of hydrophobicity (Wessel, 1988). As a result of reduced infiltration in WR soils and composts, water transport into the matrix becomes impeded (Hurrab and Schaumann, 2006). Hydrophobic soils experience an increase in surface run-off and a reduction in water infiltration rate, resulting in uneven wetting or preferential flow pathways throughout the matrix (Bryant *et al.*, 2007). Landscapes with highly stable, persistent WR tend to have enhanced rates of soil erosion, leading to the loss of valuable productive land (Doerr *et al.*, 2000). The increase in surface run-off can also have a negative impact on the water quality in the main waterways draining a catchment (Bryant *et al.*, 2007). In soils where WR is a frequent phenomenon, erosion and loss of vegetation can occur (Doerr, 2000).

There is a growing research interest in the impact of warmer temperatures on upland peat bogs responsible for storing considerable volumes of water and soil carbon (Toberman *et al.*, 2008). The concern being, that should the highly organic soil surfaces of peat bogs become exposed to repeated drying cycles, then persistent hydrophobicity could easily develop, coupled with quasi-irreversible changes on the physical structure (Dekker, 1996). This could have serious implications for water storage in peat bogs and an associated increase in flooding on lowland parts of the catchments. Add to this the increase in dissolved organic carbon (DOC) in freshwater systems, thought to occur as a result of increased temperatures increasing its transport from peatlands (Freeman *et al.*, 2001), then there is a real issue concerning water quality as a result of raised temperatures and concomitant soil hydrophobicity.

2.3.2.6.5 Causes of hydrophobicity

Doerr *et al.* (2006) identified three clear variables affecting water repellency: soil moisture content, textural composition and organic matter content. The causes of hydrophobicity or water repellency in soils and composts are variable but the main effect is a decrease in soil wettability, a phenomenon which is rarely spatially uniform or temporally stable (Doerr *et al.*, 2007). Water repellency rarely occurs in moist soils and tends to correlate with decreasing moisture content (Doerr *et al.*, 2000) however this is not the only determining factor as some dry soils are wettable (Doerr *et al.*, 2002). Factors such as drying temperature, actual water content and the level of wetting and drying can all affect the severity of water repellency (Doerr *et al.*, 2002). The persistence of hydrophobicity is largely related to fluctuations in soil moisture (Doerr et al, 2000). Hurrab & Schaumann (2006) clarified this finding that only soils with water contents less than 28 % by volume exhibit hydrophobic tendencies. It is widely accepted that a hydrophobic organic coating of soil particles can also cause water repellency (Oostindie *et al.*, 2008; Dekker, 2005; Hallet *et al.*, 2001 Czarnes *et al.*, 2000). The organic material tends to be the hydrophobic constituent of soils and growing media (Michel *et al.*, 2002; Czarnes *et al.*, 2000). WR organic compounds originate from decaying plant material (Doerr *et al.*, 2000) and root and fungal exudates (Czarnes *et al.*, 2000). Many forms of vegetation have been implicated in releasing hydrophobic organic compounds, including evergreen tree types, heathland shrubs and turf grasslands (Doerr *et al.*, 2000). It has been suggested as a mechanism for plant survival much like allelopathy whereby the hydrophobic litter can ensure preferential pathways and reducing surface evaporation (Doerr *et al.*, 2000). Mixing particulate organic matter with mineral soils has been shown to result in severe hydrophobicity (Oostindie *et al.*, 2008). Coarse grained soils become coated with thin and often undetectable layers of organic compounds that are generally aliphatic hydrocarbons or polar substances with an amphiphilic structure such as fatty acids and waxes (Doerr *et al.*, 2000). Figure 2.5 is a schematic representation of how organic molecules orient on a mineral surface while in contact with water creating a hydrophobic layer.

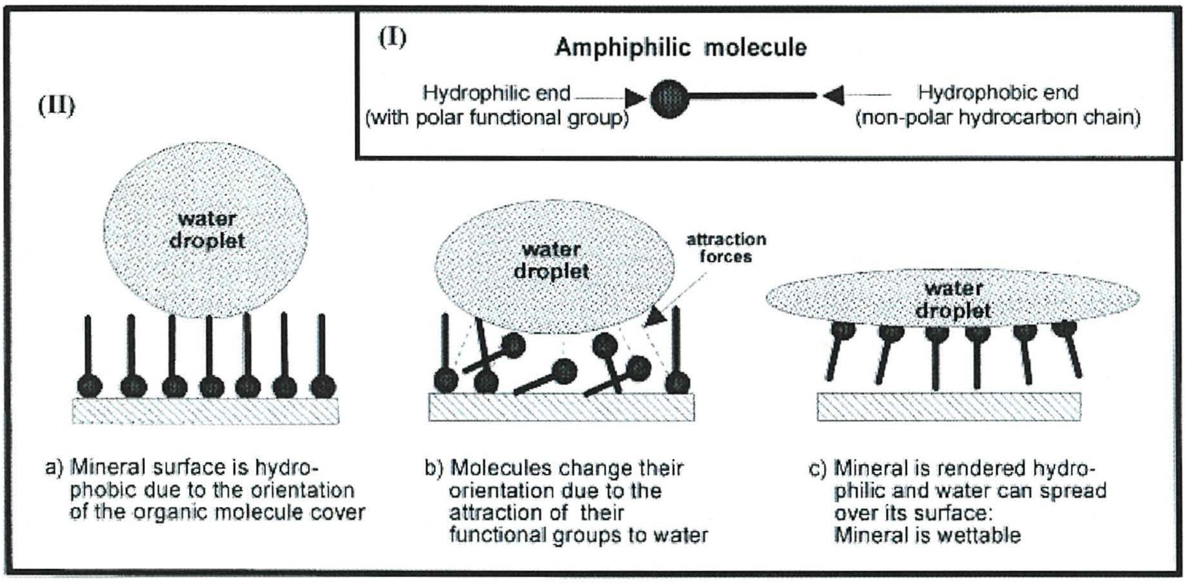


Figure 2.3 Schematic representation of (I) an amphiphilic molecule and (II) change in orientation of such molecules on a mineral surface while in contact with water (Source: Doerr *et al.*, 2000)

Research has attempted to define the relationship between organic matter and water repellency but results are always highly variable (White *et al.*, 2000). Doerr *et al.* (2000) state that the level of water repellency in soils is not proportional to the amount of organic matter in the soil. It is far more likely that type or fraction of the organic matter is responsible for the hydrophobicity and as such developing techniques to isolate OM fractions is increasingly studied (Atanassova & Doerr, 2010).

The effect of fungal exudates have been shown to cause WR in a number of studies (Czarnes, *et al.*, 2000; White *et al.*, 2000; Feeney *et al.*, 2006) Extracellular compounds, such as glomalin related soil proteins, not only bind soil/compost particles but they can influence the wetting behaviours by creating hydrophobic layers, slowing infiltration (Hallett *et al.*, 2009). White *et al.* (2000) demonstrated WR and reduced sorptivity were most likely due to the hydrophobic exudates produced by two fungal species *Coriolus versicolor* and *Phanerochaete chrysosporium.* Other studies have shown that WR is more related to changes in the physical structure of a matrix, brought about by drying and rewetting for example (White *et al.*, 2000). Huarrab & Schaumann (2006) demonstrated the highly dynamic nature or the manifestations of WR. In pH > 6.5 they saw a decrease in WR, owing to the increased solubility of humic acids responsible for WR. They hypothesised that this could be reason liming can reduce incidences of WR.

2.3.2.6.6 Methods for assessing hydrophobicity

Letey *et al.* (2000) put together a comprehensive review of various approaches to characterizing the degree of soil water repellency. The key consideration is determining what aspect of water repellency is of most interest. For the purposes of this research the persistence or stability of WR is the key consideration and so the main method for measuring this is outlined below.

'Water Drop Penetration Time' (WDPT)

The WDPT test was developed by Letey (1969) and standardised by Doerr (1998). Essentially it measures the time period that hydrophobicity persists on a porous surface. A drop of water is placed on the compost surface and the time taken for it to penetrate the matrix is recorded. If the drop does not penetrate immediately it indicates that the water surface tension is above that of the soil surface and so the soil – water contact angle (θ) is $\geq 90°$. Since water enters a porous surface if θ is $\leq 90°$ then this procedure measures the time taken for θ to alter from $\geq 90°$ to $\leq 90°$. In summary, the contact angle of water on a plane surface therefore provides an index of water repellency. As soil is neither flat nor 2 dimensional and has a complex geometry of pores the capillary rise equation can be manipulated to calculate the water repellency

$$h = 2\lambda \cos\theta / r\rho g \qquad (1)$$

where h is the height of the rise, λ the liquid – air surface tension, θ the liquid – solid contact angle, r the capillary radius, ρ the liquid density and g the gravitational constant. It follows that if $\cos\theta$ in equation (1) is zero or a negative number, then water will not spontaneously enter the soil.

The stability or persistence of hydrophobicity in soils is an important determinant of factors such as surface run-off at a landscape scale and the WDPT test can characterize the degree of WR (Doerr, 1998). However if the radius of the capillary tube at the surface is greater than that of the water droplet, then it will penetrate the matrix, even when θ is $> 90°$.

2.4 Improving Compost Quality

There is an identified need for industry to develop a wider variety of sustainable end uses for waste-derived compost other than simple land disposal (Smith *et al.*, 2008; Owen, 2005; Hauke *et al.*, 1996). The following section examines potential methods for enhancing the quality of green waste-derived compost as a plant growth media. The key areas examined, relate to the interests of the industrial sponsor of this PhD.

2.4.1 Vermicomposting partially composted waste

Vermicomposting is a relatively low input mesophilic operation, where worms are used to process organic waste (Ndegwa & Thompson, 2001). Worms pass organic matter through their gut (a grinding gizzard), living off the microorganisms in the material and rapidly fragmenting the material. The high activity of hydrolytic enzymes released from both the earthworms and microbes in the gut leads to humification and stabilization of organic matter (Vivas *et al.*, 2009). The earthworm gut contains a diverse microbial community and alongside the consortia contained in the organic matter the casts (excreta) (Edwards and Bohlen, 1996) tend to be stabilised (humified) more quickly compared with just aerobic composting (Roberts *et al.*, 2007). Compared with conventional aerobic composting, vermicomposting produces a less heterogeneous end compost product, particularly in terms of particle size (Figure 2.6) (Edwards & Bohlen, 1996). Further, waste transformation can be significantly more rapid than with conventional composting which can take up to 6 months until a stable end product is acquired (Ndegwa & Thompson, 2001). The most common worm species used by the composting industry are *Dendrobaena veneta* and *Eisenia foetida*, the latter being more commonly used in the USA (Roberts *et al.*, 2007). Although raw green waste can be vermicomposted, Singh & Sharma (2002) found that partially composting materials with high lignin contents, such as greenwaste, provide the best feedstock for vermicomposting compared with the raw materials, producing a compost with greater concentrations of available N and P compared with the aerobically composted equivalent and reducing processing time by 40 days. Due to the cost of producing vermicompost, it must produce a high value-added product capable of being marketed to the horticultural and amateur gardening sector where higher prices can be charged (Owen, 2005).

Figure 2.4: Composted green waste (left) and vermicomposted green waste (right). The vermicompost was produced from the same feedstock after 6 weeks with the worms (Source: Wormtech Ltd)

There has been an exponential rise in studies looking at the use of vermicomposting to transform waste into high quality growth media (Atiyeh *et al.*, 2000; Atiyeh *et al.*, 2001; Roberts *et al.*, 2007; Ali *et al.*, 2007). Typically, pure vermicompost can contain high concentrations of some plant nutrients, in particular soluble salts, inducing phytotoxic conditions and so is commonly diluted with conventional compost at a rate of around 20% (v/v) to achieve optimal plant growth (Roberts *et al.*, 2007). Research has shown that vermicomposting enhances the quality of conventional compost in a range of ways including:

- Increased particulate surface area
- Increased cation exchange capacity
- Reduced pH – often too high in composted wastes
- Presence of plant growth promoting hormones present in humic structure
- Stabilisation rate is accelerated
- Increased levels of polysaccharides and structural stability
- Greater homogeneity in the finished product

(Edwards & Bohlen, 1996; Vinceslas-Akpa & Loquet, 1997; Singh & Sharma, 2002; Contreras-Ramos *et al.*, 2005; Roberts *et al.*, 2007).

Vermicomposting waste has been shown to promote greater bacterial diversity, and greater functional diversity in the end product (Vivas *et al.*, 2009). In a wider context, microbial diversity is linked to ecosystem stability with increased microbial diversity raising the ecosystem resistance to environmental perturbation (Wittebolle *et al.*, 2009). So arguably, vermicompost may be a superior additive to soil and natural ecosystems than compost in terms of microbial diversity. The worm cast (excreta) produced is rich in polysaccharides attracting a diverse array of microorganisms, ultimately speeding up the stabilisation of the humic material. The different transformation of organic matter via vermicomposting compared with aerobic composting can be explained by the mutualistic relationship between the intestinal microflora and intestinal mucus of the worms (Vinceslas-Akpa & Loquet, 1997). The humic substances in vermicompost have been reputed to contain auxinlike components which may promote plant growth (Atiyeh *et al.*, 2001; Cannellas *et al.*, 2010; Dobbss *et al.*, 2010). By isolating humic substances from vermicomposts to understand the role they play in promoting plant growth, studies have shown that hydrophobicity of certain humic acids can essentially protect bioactive molecules from further degradation (Dobbss *et al.,* 2010).

A variety of studies have assessed the potential for pathogens to be spread by vermicompost. The high temperatures generated during the thermophilic phase of composting (50–70 °C) can eradicate mesophilic organisms such as *E. coli* O157:H7, whereas in vermicomposting these temperatures are never reached. Some studies, however, have shown that pathogen reduction is still achieved (Contreras-Ramos, 2005), while others have demonstrated that earthworms can be vectors for diseases such as *E. coli* O157:H7 (Williams *et al.*, 2005). However, it has been also been shown that even the temperatures reached in the thermophilic phase of conventional composting are not always sufficient to ensure complete pathogen elimination as composts can be readily recolonized by pathogens (e.g. *Salmonella*) after composting has completed (Christensen *et al.,* 2002; Déportes, 1998)

In comparison to peat-derived composts, non-peat derived composts often have a higher bulk density and become readily hydrophobic, properties which are not desirable for containerised plant growth (Rainbow & Wilson, 1998). One key question is therefore whether vermicompost behaves more favourably in this respect. Anicua-Sanchez *et al.* (2008) reported that vermicompost, peat and compost all have internally connected pores that account for the high porosity and water holding capacity. The parameters evaluated were pore size, abundance, shape, surface roughness, orientation and distribution pattern. The porosity values for the compost and vermicompost were comparable (ca. 70%) while peat was considerably higher

(94%). In addition, the original waste feedstock material used is likely to exert a strong influence on the final properties of the vermicompost, as occurs with composting (Contreras-Ramos *et al.*, 2005). Decreased wettability (hydrophobicity) in composts has been linked to carbohydrates and humic substance content (Albiach *et al.*, 2001; Annabi *et al.*, 2007) and so establishing whether vermicomposts are more prone to hydrophobicity than compost and peat is important in terms of developing high quality growing media (Seaby, 1999).

2.4.2 Sterilization of compost pre-plant germination and growth

Traditionally, gardeners have manually heat-sterilized composts to ensure that harmful pathogens and weed propagules are destroyed or minimised. This approach has been adopted as small scale composting rarely achieves sufficient sanitization temperatures due to the physical size of the operation and the lower management levels, both of which limit microbial activity and heat production (Lawrence & Newell, 1945). By sterilising compost, it ensures that the end compost is hygienic and can relieve operators of the high time and cost inputs required to achieve the PAS: 100 compost quality protocol (BSI, 2005). Eliminating the biotic community in soil or compost by sterilization has generally been employed as a way to experimentally differentiate between the relative importance of biotic and abiotic processes. This can be achieved in a variety of ways including:

- Fumigation – chloroform ($CHCl_3$), and methyl bromide (CH_3Br)
- Autoclaving
- Gamma (γ) irradiation
- Microwave radiation

Problems associated with different sterilization approaches, however, have been demonstrated. For example, most sterilisation methods result in the perturbation of the physical and chemical properties of the soil/compost and often fail to guarantee a complete and prolonged deactivation of the microbial biomass (Bank et al., 2008; Egli et al., 2006; McNamara et al., 2003). Further, standardized protocols for sterilization rarely exist and key parameters vary greatly between studies (McNamara et al., 2003). The following section outlines the basic processes that result in the deactivation of the biota.

- **Fumigation**

Fumigation was a common method for sterilizing soil for most of the 20th century. However, fumigation using methyl bromide is being phased out worldwide due to the risk associated with ozone-depletion (Egli et al., 2006). With most fumigation methods toxic residues are left behind, which may cause subsequent contamination of the soil matrix affecting plant growth (Bank, 2008).

- **Autoclaving**

In this procedure, thermal energy is supplied under pressure forcing steam through the soil matrix at temperature and pressures in the region of 120°C and 150 kPa. Autoclaving is highly effective in killing organisms (through denaturation of proteins) and is widely used in research and industry. Autoclaving, however, has been shown to decrease soil surface area due to the aggregation of the clay particles reducing the adsorption capacity for nutrients and organic matter (McNamara, 2003, Trevors, 1996). In addition, the high heat means that the water has greater solvent ability and more organic acids are solubilized (Goto *et al.*, 2004).

- **Gamma irradiation**

Sterilization of soils is achieved by exposing the soil to γ-irradiation emitted by cobalt-60 which destroys cellular DNA and kills organisms. It has been shown to be an effective method of sterilization and large quantities of soil can be effectively sterilized without incurring any heat or pressure effects (McNamara, 2003; Herbert et al., 2005). While it is an attractive option, rarely, however, have the indirect effects of ionizing radiation on soils been examined (Bank, 2008).

- **Microwave irradiation**

A study by Wang et al. (2001) demonstrated that the key factors affecting the efficacy of sterilization by microwaving were: position in the microwave; soil moisture; soil weight; soil texture and irradiation time. The findings most relevant to irradiating composts: ensuring similar moisture contents in all composts; regulating the quantity of compost irradiated; ensuring the microwave plate rotates to negate the effects of position in the microwave ensuring more irradiation is received.

2.4.3 Plant disease suppression via compost amendment & compost tea

It is widely accepted that organic amendments can reduce disease incidence caused by a wide range of plant pathogens and plant pests including fungi, bacteria and nematodes (Daddabbo *et al.*, 2009; Noble *et al.*, 2006; Bailey & Lazarovitis, 2003). Using waste-derived compost as a natural disease suppressant is becoming increasingly well researched but the mechanisms involved are not straight-forward and the factors influencing the suppressive potential of composts are highly interactive (Zhang *et al.*, 2009; Zmora-Nahum *et al.*, 2008). Following a comprehensive report from Noble *et al.* (2006) it was clear that most research into disease suppression via organic amendments had been performed using vegetables not flowering plants/ornamentals – and on containerised media, as oppose to in the field. To ensure a level of disease suppression in the field, estimated application rates would have to exceed 100 t ha^{-1} (Noble *et al.*, 2006) but this would contravene the guidelines for Nitrate Vulnerable Zones (NVZs), which sets it at around 30 t ha^{-1} of green waste compost. This amount of compost would roughly supply 250 kg N ha^{-1}, which is the maximum for NVZs (WRAP, 2009)

Composts have always been highly variable in the pathogens that are inhibited and the different circumstances in which suppression occurs (Kavroulakis *et al.*, 2005; Noble *et al.*, 2006). The marginal understanding of the mechanisms for disease suppression is partly due to the inherent heterogeneity of composting materials, causing suppressive potential to be highly variable at the outset (Noble *et al.*, 2006; Kanroulakis *et al.*, 2005) and also the lack of a standard method for ascertaining whether organic amendments are indeed disease suppressant (Noble *et al.*, 2006) . Other key factors influencing the biocontrol potential of compost relate to the complex interaction between the soil and compost physico-chemical properties; host plant species; pathogen species and the actual population dynamics of the compost and soil microbial consortia, making it difficult to establish whether a given compost is suppressive or not, for a specific plant, in a specific soil (Berg *et al.*, 2006; Van der Gaag, 2007). Importantly these are key reasons why it is difficult to ascertain a predictive test for disease suppression (Noble *et al.*, 2006).

A number of bacteria and fungi have been isolated from composts and have been shown to reduce or suppress disease incidence, most commonly for *Phythium* spp. and *Phytophthora* spp. (Boehm *et al.*, 1997; Carisse *et al.*, 2003). But laboratory scale demonstrations of suppression may not necessarily translate into suppression of pathogens in the plant/soil environment (McKellar & Nelson, 2003). Compost and compost extracts produced from greenwaste were shown to inhibit the growth and survival of pathogens *in vitro* but it is less clear that this will necessarily translate into suppressiveness (Palazzini *et al.*, 2009; El-Masry *et al.,* 2002). Interestingly peat, being a highly stable material does not support the same levels of microbial activity as waste-derived compost and has been shown to be comparatively disease conducive (i.e. plant pathogens successfully infect plants) concerning soil borne plant pathogens (Krause *et al.*, 2001). Borrero *et al* (2006) found that the microbial consortia conducive to wilt in peat utilised sugars, whereas the suppressive consortia in the composts utilised mainly carboxylic acids, amino acids and phenolic compounds, which are less easily biodegradable. This study used a well decomposed light peat and they found that microbes in well decomposed composts and peat tended to metabolise more accessible compounds like sugars, whereas less decomposed composts contained consortia that tended to metabolise more recalcitrant compounds and it was at this trophic level that the consortia were associated with suppression.

2.4.3.1 Compost Tea

Compost tea has been used as an effective fertiliser increasing plant yields (Haggag & Saber, 2007; Hargreaves *et al.*, 2008) and as a foliar spray to reduce plant pathogens (Yohalem *et al.*, 1994; Haggag & Saber, 2007). Most reports of effective disease suppression via compost tea have been using nutrient rich manure based compost and relatively few studies have looked at the efficacy of compost teas created from greenwaste (Noble *et al.*, 2006). Scheuerell & Mahafee (2002) did an extensive review on the use of compost teas for plant disease control. They draw an important distinction between compost tea and compost extracts by the fact that in teas there has been a fermentation process, compared with an extract which is filtered compost after mixing in a solvent. Compost tea production time can be anything from 1 day to several weeks and the 'steepage' process is either aerated or non-aerated (Ingram & Millner, 2006). Scheuerell (2006) found that for non-aerated teas (NCT) compost type and the length of brewing time was most closely associated with disease suppression. Continuously aerating the compost tea (ACT) did not improve the level of suppression compared with NCT but ACT with adjuvants – kelp extract, rock dust and humic acid saw a significant improvement in the level of disease suppression compared with ACT with no adjuvants. NCT have also been shown to contain higher biodiversity of actinomycetes, bacteria, filamentous fungi and yeasts compared with ACT (Haggag & Saber, 2007).

Some studies have found that suppression increases with the addition of nutrients to the compost tea (Scheuerell, 2006; Hagagg & Saber, 2007) but Ingram & Millner (2007) found that additional nutrients also increased the levels of the human pathogens *E Coli* spp. and *Salmonella* spp. in the compost teas, with comparatively higher CFU/g in ACTs over NCTs. Further work is required to assess the level of colonization of teas with human pathogens, especially when spraying on salad crops. However, as with compost substrates the mechanisms for disease suppression via compost tea, are relatively unclear and even when tea is produced from similar materials the level of disease control can be highly variable (Scheuerell & Mahaffee, 2002). The main benefits of producing compost tea is that it could potentially be high value product for the composting industry and that it can be applied as a foliar spray as well as to the rhizosphere (Noble *et al.*, 2006). However should compost tea fail as a disease suppressant then potentially the producer will experience greater financial losses from this market, compared with solid phase compost which can offer accumulative benefits to soil structure and is a sustainable peat replacement (Noble *et al.*, 2006).

The following sections on this topic consider the main mechanisms to explain disease reduction via compost and compost tea.

2.4.3.1.1 Antagonistic Activity

One of the most widely accepted explanations of the biocontrol of soil borne pathogens is the antagonistic activity of microbes that inhibit the growth and survival of the pathogen, or otherwise affect the pathogen infectivity of the plant under attack (Fravel, 1988; Berg *et al.*, 2005). The mechanisms underpinning antagonistic activity are summarised below:

- Antibiosis – the release of specific or non-specific metabolites such as antibiotics that inhibit or kill phytoparasites (Agrios, 2005). This chemical release confers a competitive adaptation for gaining space and nutrients within a given ecosystem (Fravel, 1988). The main chemicals released are: antibiotics, toxins and biosurfactants (Berg *et al.*, 2005) Chemical defence by microbes is comparable to the release of allelopathic chemicals by plants, particularly invasive species, to out-compete neighbouring plant species (Prati & Brassdorf, 2004). Microbiostasis describes the microbial release of chemicals that has a slow reduction in the survival of propagules (Borrero *et al.*, 2008)
- Parasitism/predation – release of cell wall hydrolytic enzymes to specifically destroy another organism for the direct benefit of the predating microorganism (Kavroulakis et al., 2005).
- Competition for nutrients (seed exudates - e.g. fatty acids)

It is very hard to predict the natural presence of antagonists in compost and some authors have suggested that the best solution would be to manually enrich the compost with known antagonists (Veekan *et al.*, 2005; De Ceuster & Hoitnik, 1999). Santos *et al.*, (2008) found that seeding substrates with isolated suppressive organisms was not effective and suppression did not transfer from one substrate to another by inoculation. In *in vitro* assays isolated species can be effective antagonists but once in *in vivo* - with all the associated microbial populations contained in it - disease suppression becomes much harder to predict (Scheuerell, 2006). It is clear though, that suppressive effects are not down to a single microbial species but rather the interaction of the microbial community, coupled with the complex interaction between the microbial community and substrate conditions (Santos *et al.*, 2008; Zmora-Nahum *et al.*, 2008; Berg *et al.*, 2005). Studies have also shown that population densities of suppressive organisms are frequently the same in suppressive as in conducive composts (McKellar & Nelson, 2003), providing further support that a systems approach is imperative when studying microbial communities (O'Donnell *et al.*, 2007). Added to this is the plant-dependant nature of not only infection by pathogens but importantly the colonization by antagonists in the spermosphere and/or rhizosphere is also plant species specific (Berg *et al.*, 2005).

2.4.3.1.2 Physico-chemical conditions

The physical and chemical properties of the soil affect the growth and development of plants and microbes and have been shown to affect the level of disease incidence, either indirectly or directly (Tilston *et al.,* 2002; Grosch & Kofoet, 2003; Boulter, 2000). Some experiments using sterile compost have shown that the suppression of plant pathogens can still occur, indicating that it is not always the biota influencing the level of disease incidence (Kavroulakis N. *et al.*, 2005; Zmora-Nahum *et al.,* 2008). Other research has shown the opposite, supporting the biological explanation for suppression (Bailey & Lazarovits, 2003; McKellar & Nelson, 2003).

pH has been shown by many studies to affect the level of disease suppression in a given substrate but the effect of pH varies and is dependent on many factors (Tilston *et al.*, 2002). Having investigated several different composts, Termorshuizen *et al.* (2006) found that the only consistent relationship between compost biological and chemical characteristics and disease suppression was the pH, which tended to be raised following compost application. Other studies have shown that lowering the pH in some substrates induces a higher rate of disease suppression in some soils (Chet & Baker, 1980; van der Gaag *et al*, 2007). One hypothesis for this relationship is that the key soil organisms are stimulated at lower pH values and suppress the growth of pathogens, meaning that ultimately suppression is still microbial-induced but that this effect is in part dependent on the chemical conditions of the compost (van der Gaag *et al.*, 2007). Zmora-Nahum *et al.* (2008) demonstrated that the chemical conditions in compost can have significant disease suppressive effect. By sterilising the compost, the sterile immature compost completely inhibited *Sclerotia* indicating that it was the chemical conditions of the compost suppressing soil borne pathogens and not the microbial consortia. This study built on the findings of previous work (Punja & Jenkins, 1984) that identified that *Scelrotia* acidify their environment by secreting oxalic acid, creating more favourable conditions for themselves. Zmora-Nahum *et al.* (2008) found that in substrates containing carbon sources which have a high buffering capacity, *Sclerotia* exhausted their reserve of oxalic acid, remaining permanently inhibited. This study highlights how the highly interactive nature of the soil/compost environment makes it very difficult to define single mechanisms, and consequently tests, that could determine the disease suppressive potential in organic amendments (Noble *et al.*, 2006).

The balance of nutrients available to plants is a potential mechanism for disease suppressive qualities in an organic amendment (Arancon *et al.*, 2005; Marschner, 1995). Certainly key nutrients are linked to the incidence of infection via soil borne phytopathogens. Marschner (1995) reports that increasing soil calcium decreases infectivity via *Rhizoctonia solani* spp. and *Gaeumannomyces graminis* (take-all) is sensitive to manganese availability in the rhizosphere. It has been recognised for some time that the inorganic N dynamics influence the rate of plant infection in soils. An imbalance of inorganic N can develop plant tissue that is more susceptible to infection (Lucas, 1998) and a shortage of NO_3^- can delay germination increasing the chance of infection (Salisbury & Ross, 1992). Tilston *et al* (2002) studied the suppressive effects of greenwaste composts and were able to make a number of recommendations regarding the processing of the feedstock with regard to maximising suppressiveness. Their results suggested that mature composts were more suppressive particularly in relation to the high levels of extractable carbon (water soluble carbon (WSC)) and lower levels of nitrate (NO_3^-), with a pH in the region of 7.0. However despite these conclusions it is more likely that high levels of WSC would be found in more immature composts compared with mature compost (Said-Pallucino *et al.*, 2007). Tilston *et al.* (2002) recommended that particle size of composts should not be <4 mm as in their study compost was significantly less suppressive below this particle size. However, the treatments were amended by 25% with different materials and consequently there is no way of decoupling the effect of this ingredient and end particle size on the outcome of suppressiveness.

Water availability is a key determinant of microbial survival in the soil and so this parameter has a direct effect on disease development and suppression in the growing media and soils (Lucas, 1998). The ability of microorganisms to tolerate water stress can have a significant effect on their ability to grow and inhabit specific natural environments (Palazzini *et al.*, 2009). Microorganisms to some degree, engineer their immediate surroundings making significant alterations to their physical surroundings via the release of metabolites (e.g. polysaccharides), changing the distribution and amount of air and water in the matrix. (O'Donnell *et al.*, 2007). It may be that over time, with the addition of organic amendments to a soil, as soil structure and functionality improves then the accumulative benefits will afford greater natural disease suppression (Bailey & Lazarovits, 2003).

2.4.3.1.3 Compost Maturity

The length of the compost maturation phase and the storage method are both important factors determining the disease suppressant effects of composts. Some studies have shown that compost must reach sufficient maturity to go from disease conducive to disease suppressive (Veekan *et al.*, 2005; McKellar & Nelson, 2003; EL-Masry *et al.*, 2002; Sidhu, 2001). In contrast Danon *et al.* (2007) found that prolonged curing and storage depletes the disease suppressive quality of the compost, which is most likely due to the lack of antagonistic effects in highly stable composts. As certain carbon sources become exhausted, the number and diversity of microbial communities naturally subsides (Sidhu *et al.,* 2001; Gazi *et al.*, 2007). Zmora-Nahum *et* al. (2008) found that mature unsterilized compost did not inhibit the germination of *Sclerotia* providing further support for the claim that prolonged curing can deplete disease suppression. Van Rijn *et al.* (2007) found a significant interaction between compost sample and compost storage relating to disease suppression, suggesting that the effect on suppression due to storage depends upon the compost characteristics. Importantly compost stability and maturity (see section 2.2.3.5) is considered to be a key factor determining end quality and safety for the consumer and highly unstable composts are generally considered to be immature/phytotoxic. Although, Sidhu *et* al., (2001) found that following prolonged storage, mature composts had much higher pathogen regrowth potential, linked again to the lack of antagonistic colonization. This means that finding the optimal curing time that will produce disease suppressive compost or compost tea that are equally non-phytotoxic, is unlikely, owing to the highly interactive nature of disease suppression. The variation in the supply of initial compost materials or feedstock has long provided the industry with quality control issues as it makes the operational guidelines to enhance disease suppression via compost very difficult, as the properties of the initial ingredients has such a high impact on the end product (Borrero *et* al., 2006; Bernal *et al.*, 1998). The microbial species or community inducing either disease suppression or conduciveness are in part determined by the carbon compounds available for consumption (Borrero *et al.*, 2006). In reality the variation in the initial characteristics of feedstocks will make the development of an effective protocol for disease suppressive composts very remote.

The transformation of carbon during compost maturation has been linked to the level of suppressiveness. Tilston *et al.* (2002) states that there is a negative relationship between suppression and extractable or labile carbon (DOC) concentrations, meaning that higher levels of DOC generally found in immature composts, lower the suppressive capacity of the matrix. This obviously relates to the microbial populations which will differ significantly depending upon the level of DOC. Kavroulakis *et al.* (2005) found that suppressive compost, applied as an extract so as to test the suppressive potential of the water-soluble components, was a less effective suppressant than the solid substrate.

2.4.3.2 Allelopathy

Allelopathy is an interference mechanism whereby plants and microbes release chemicals that can have a detrimental impact on surrounding plants (Tawaha & Turk, 2003) microbes, including soil pathogens (Zhang *et al.*, 2009; Mckellar & Nelson, 2003) and herbivorous predators (Wardle *et al.*, 1998). Typically the chemicals are secondary metabolites and are released at different stages of a plant's life cycle, depending on the plant species and various ecosystem interactions (Prati & Bossidorf, 2004). Secondary metabolites synthesized by plants are present in different tissues including leaves, stems, flowers, fruits, seeds and roots. Their release into the environment occurs via volatilization, leaching, decomposition of residues and root exudation. As more sustainable farming is increasingly encouraged to promote biodiversity and soil health (Bailey & Lazarovits, 2003) integrated management techniques employ crop covers of plants like rye (Héraux *et al.*, 2005) and that release allelochemicals to reduce weed infestations (Jose & Gillespie, 1998).

Elemental allelopathy relates to plants causing an inhibitory effect to another plant or microbe via inorganic elements (Morris *et al.*, 2009). Via root exudates or phytoenrichment certain nutrients can become more available, perhaps at concentrations that only the host plant can tolerate, and this is often the success strategy employed by invasive plant species. Zhang *et al.* (2008) found that the root and rhizome exudates from the invasive plant species *Solidago Canadensis* L. suppressed local soil pathogens. This gives an indication that allelopathy is impacting at an ecosystem scale and not as a unilateral association between two plant species (Prati & Bossidorf, 2004; Wardle *et al.,* 1998).

Allelopathy has typically been assessed via laboratory scale bioassays and some scientists have concerns that this cannot represent ecological reality (Wardle *et* *al.*, 1998). The same has been noted for laboratory scale experiments on biocontrol where, *in vitro,* the microbial consortia from a compost extract suppress fungal pathogens and yet, at an ecosystem scale the same suppressive substrate does not inhibit the pathogen (McKellar & Nelson, 2003). Wardle *et al.* (1998) argued that the concept of allelopathy can be explained at the ecosystem scale, as there is growing evidence demonstrating the role of secondary plant metabolites in determining interactions beyond plant – plant, for example plant-herbivore interactions. Plants that are unpalatable, owing to the production of secondary metabolites (e.g. polyphenolics), have a competitive advantage over other plants as they are not heavily grazed compared with plants that are palatable. However most primary production is not consumed by herbivores but tends to end up as litter entering the soil system. Litter decomposition is a complex process with the initial characteristics of the litter, along with climate and soil properties being key determinants of the rate and level of degradation (Lindedam *et al.*, 2009). However the release of allelochemicals during the decomposition of litter is well documented (Ahmed *et al.*, 2008) and allelopathy during the end stage of the plant life cycle is the most relevant when considering allelochemicals in compost. Toxic compounds are present in many organic wastes. Olive mill solid waste contains a variety of polyphenols and these are toxic to many microorganisms. Olive waste also contains bactericide and bacteriostatic activity (Vivas *et al.*, 2009). Volatile fatty acids (VFA) have been have been detected in fresh compost extracts and have been linked to inhibiting the growth of various pathogens (e.g. *Brassica rapa* L). *Brassica nigra* L. Koch. (black mustard) contains water soluble allelochemicals that are released during plant decomposition (Tawaha & Turk, 2003). Allyl-isothiocyanite (ITC) isolated from black mustard residues was identified as the active allelochemical. Also the breakdown products of glucosinolate (e.g. ionic thiocyanite SCN^-) inhibit root and shoot growth in several species (Brown *et al.*, 1991). Volatile compounds are released from *Brassica* tissue residues like isprenoid and benzenoid suppress the growth of weeds (Tawaha & Turk, 2003) and this has prompted some researchers to consider the possibility of using allelochemicals in weed control. Wardle *et al* (1998) studied *Carduus nutans*, an invasive species that populates pastures reducing their value considerably. *Carduus nutan's* first growth is a rosette of leaves that occupy 1 m diameter land space. Once the flowering stems have grown the rosette dies off and during the rapid 15 day period of decomposition, a bare patch is created around the plant via the suppression and death of pasture species, particularly legumes. This study demonstrated that secondary metabolites released during the decay of the rosette interfered with symbiotic N fixation hence having more detrimental impact on legumes than grasses. Whether allelochemicals in decaying organic matter are transformed more effectively during the composting process compared with natural decay, has not been researched. As the water soluble carbon compounds are associated with phytoxicity and curing stage (Tilston *et al.*, 2002), it is important to assess whether compost extracts of teas could exert an allelochemicals action on treated plants Allelochemicals could be in some way responsible for the phytotoxic conditions in some compost and compost teas.

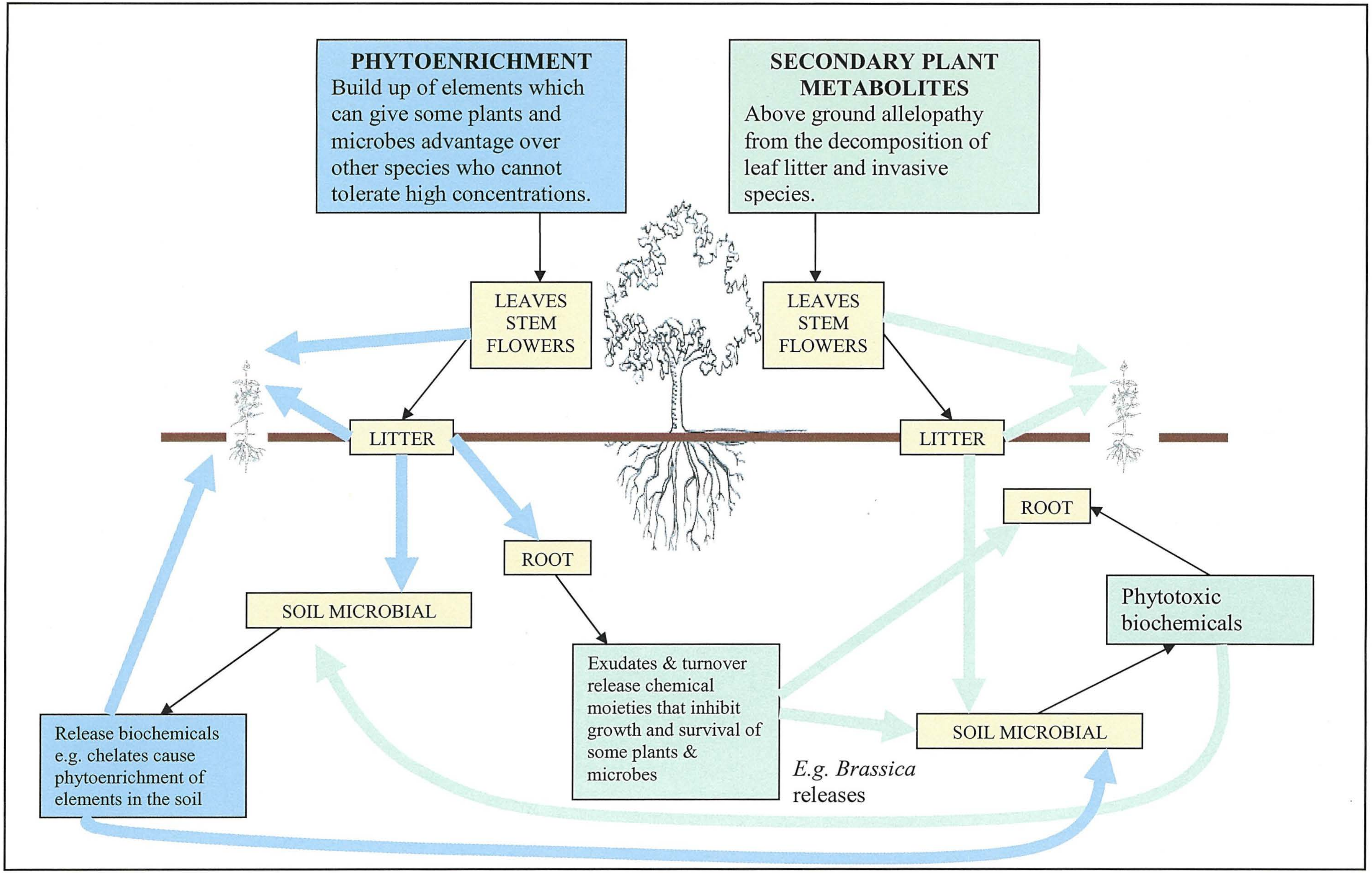


Figure 2.5: Ecosystem Scale Biochemical & Elemental Allelopathy – key synergistic effects

2.4.4 *Wetting agents – using surfactants to eliminate hydrophobicity*

Hydrophobicity in sand-based turf systems such as golf greens is a common phenomenon (Doerr & Ritsema, 2005) and has caused significant management problems (Cisar *et al.*, 2000). Finding practical and cost effective ways to alleviate hydrophobicity in soils are therefore urgently required. Coarse soils allow for good drainage and initially solutions for improving water retention were sought via organic matter inputs. However, this action frequently exacerbates water repellency, owing to hydrophobic organic coating of soil particles (Oostindie *et al.*, 2008). Hurrab & Schaumann (2006) attribute this effect to the high concentrations of polyvalent cations in humic substances resulting in a reduction in the reactivity of hydrophilic mineral surfaces. Using soil surfactants has become an increasingly popular remediation technique on golf courses (Miller, 2001) Figure 2.7 demonstrates how the hydrophobicity on the non-surfactant/untreated side of the fairway has resulted in significant die-back of the turf, whereas the surfactant remediated/treated side is not suffering from the effects of water repellency (i.e. lack of water infiltration and concurrent dry spots developing, affecting turf quality). Hydrophobicity is also commonly observed in other ecosystems such as soils exposed to forest fires (Darboux *et al.*, 2006).

Figure 2.6 Difference in grass quality between two parts of the fairway treated with surfactant (right hand side) and untreated (left hand side). (Source: Oostindie *et al.*, 2008)

Wetting agents are also commonly added to soilless media, such as peat (Urrestarazu *et al.*, 2008) and have been observed to have a similar effect as those applied at a landscape scale (i.e. improving wettability). The key benefits of surfactants seen in soil and soilless media are:

- Increased wettability - reduction in water repellency and localised dry spots
- Reduction in run-off and enhanced infiltration rate
- Even distribution of water in the wetting zone – eliminating preferential flow paths
- Greater water retention
- Improved plant growth and survival
- Increased irrigation efficiency

(Oostindie *et al.*, 2008; Olszewski *et al.*, 2008)

There are a number of products available to improve water infiltration and retention (Urrestarazu *et al.*, 2008). Non-ionic surfactants are biodegradable and relatively inexpensive to produce (Zhao *et al.*, 1998) but there is a paucity of literature regarding their application to improve the wettability of green waste composts (Urrestarazu *et al.*, 2008; Olszewski *et al.*, 2008). Non-ionic surfactants tend to be synthetic derivatives of ethylene oxides or propylene oxides (Hepworth, 2006) and are typically less chemically active than their ionic equivalents (Urrestarazu *et al.*, 2008). Organic or naturally occurring surfactants tend not to be used in horticulture owing to a lack of efficacy and because they essentially become a nutrient source for microbes (Olszewski *et al*, 2008). However research is looking at isolating surfactants from the humic content of waste derived composts such as food and green waste (Quagliotto *et al.*, 2006). Quagliotto *et al.* (2006) suggest that the compost humic acid-like polymeric matter isolated from green and food waste derived compost, compared favourably with leading biosurfactants with the added benefit of being a low cost production process.

Table 2.4 Names and abbreviations of the main classes of surfactants (Ivankovic *et al.*, 2010)

Common Name	Abbreviation	Class
Linear alkylbenzene sulphonic acid	LAS	Anionic
Sodium dodecyl sulphate	SDS	
Alkyl sulphate	AS	
Sodium lauryl sulphate	SLS	
Alkyl ethoxysulphate	AES	
Quarternary ammonium compound	QAC	
Benzalkonium chloride	BAC	Cationic
Cetylpyridinium bromide	CPB	
Cetylpyridinium chloride	CPC	
Hexadecyltrimethylammonium bromide	HDTMA	
Amine oxide	AO	Amphoteric
Alkylphenol ethoxylate	APE	Non-ionic
Alcohol ethoxylate	AE	
Fatty acid ethoxylate	FAE	

Anionic and cationic surfactants tend to have higher critical micelle concentration (CMC) than their non-ionic counterparts (Ying *et al.*, 2006; Hepworth, 2006). This is important as it describes the threshold whereby the concentration of the surfactant in water will result in the surfactant molecules aggregating – forming micelles – which reduces free energy in the system (Ivankovic *et al.*, 2010). Concentrations higher than the CMC result in the solubilisation of hydrophobic organic compounds and a concurrent increase in wettability.

There are however concerns about the levels of surfactants released into the environment and particularly their effects on aquatic organisms (Ivankovic *et al.*, 2010) and there is a lack of research into the long term effects of using surfactants as a soil and composts amendments for increasing wettability.

CHAPTER 3: STERILIZING GREEN WASTE COMPOST ENHANCES SEEDLING ESTABLISHMENT AND PLANT GROWTH

Pagella S.L., Roberts P. & Jones D.L.

3.1 Abstract

Due to the inherent environmental damage associated with peat extraction, there is a concurrent need to develop high quality peat replacements for the horticulture industry. Many of the current waste-derived replacements perform poorly in comparison to their peat-derived counterparts and public perception of the efficacy of these alternatives is still an issue. We considered the effects of sterilizing two green waste composts to see if the eradication or significant reduction of the compost microbial biomass would improve seedling establishment and plant growth. Two methods of sterilisation were compared (microwaving and autoclaving). Microwaving has more potential to be developed on a large scale by the composting industry due to the lower energy running costs. Greenhouse plant growth trials were conducted using lettuce (*Lactuca sativa* L.). Standard peat and peat-free composts were used as comparative controls. Percentage emergence, shoot biomass and chlorophyll concentration were measured. Respiration was monitored for 24 hours following sterilization. Results show that both sterilization methods improved percentage emergence and plant biomass in the biowaste-derived composts. There

was a significant effect of sterilization on the compost chemistry, most particularly in the autoclaved composts, where NO_3^- for example increased in concentration.

3.2 Introduction

3.2.1 Markets and Compost End Quality

Developing a diversity of end markets for waste-derived compost is a continuing concern if the composting industry is to become sustainable (Dimambro *et al.*, 2007; Slater *et al.*, 2004). Currently, maximising the rate and volume of waste composted is the primary economic driver for most composting operations (Veekan *et al.*, 2005). Consequently, the tendency is for low quality composts to be sold in bulk, in low value markets, or even given away e.g. as landfill cover or on-site soil improvers (Hauke *et al.*, 1996; Veekan *et al.*, 2004). Peat extraction for use in horticulture is being phased out, as peat habitats are widely recognised as providing a suite of valuable ecosystems services, many of which are linked to climate change mitigation (Freeman *et al.,*1992; Barkham, 1993; Schmilewski, 2008). Producing high quality peat replacements from the increasing volume of green waste-derived composts would establish a sustainable end market for the composting industry (Faviono & Hogg, 2008).

In the UK the PAS: 100 quality assurance scheme (BSI, 2005) aims to ensure that compost from a specific site and operation is of consistent quality, whilst specifying the appropriate end use for the compost. As yet, this scheme has not been sufficient to abate consumer reticence to use biowaste derived potting composts and soil improvers and there is debate concerning the scientific validity of some of the guidelines (Nicols, 2008; Westerman & Bicudo, 2005; Lasaridi *et al.*, 2006).

3.2.2 Compost Sterilization

Traditionally, gardeners have manually heat sterilized composts to ensure that harmful pathogens and weed propagules are destroyed or minimised, as small scale composting rarely achieves sufficient sanitization temperatures due to the typically lower management levels limiting microbial activity (Lawrence & Newell, 1945). In industrial or highly managed operations the thermophilic phase of the process generates temperatures that are often > 70 °C. A strict temperature, turning and monitoring regime is required if operators want to achieve their PAS: 100 quality assurance certification. Sanitization via the thermophilic phase is not a zero emission process, as large volumes of biowaste incur increased operational inputs in terms of fossil-fuel based machinery and manpower (shredding, turning), making the operation more costly both environmentally and economically (Goto *et al.,* 2004). Added to this, there is growing speculation concerning the validity of the sanitization guidelines, as research has shown that despite achieving the PAS 100 temperatures, pathogens were still present and persistent in compost (Burge *et al.*, 1987; Casadei *et al.*, 2001; Williamson *et al.*, 2006; Christensen *et al.*, 2007; Grewal *et al.*, 2007; van Rijn *et al.*, 2007). Key factors such as pH, salts content, water availability and C:N must be highly stable for the current temperature regulations to be efficacious; therefore a more dynamic approach to sanitization may be required (Noble & Roberts, 2005; Christensen *et al.*, 2002; De Bertoldi *et al.*, 1983). Potentially harmful pathogens could also reinfect or recolonize compost following the thermophilic phase, particularly during the maturation phase where composts are often left unmanaged and outside (Burge *et al.*, 1987; Williamson *et al.*, 2006; Christensen *et al.*, 2007).

This study employed the traditional method utilised by horticulturalists and amateur gardeners – steam treatment - via autoclaving (Lawrence & Newell, 1945;

Berns *et al.*, 2008). With the knowledge that autoclaving may perturb the chemical and physical properties of the compost (McNamara *et al.,* 2003; Herbert *et al*, 2005; Banks *et al.*, 2008) a second sterilization treatment, microwaving, was incorporated into the trial. Added to this, irradiation via microwaving had less potential to completely eradicate all of the biota and so may have had more of a pasteurizing effect, meaning that the biota would be severely depleted but not eradicated (Trevors, 1996; Gibson *et al.*, 1988). Large scale heat treatment (autoclaving) requires high energy inputs whereas microwaving requires significantly less energy to achieve adequate temperatures and as such microwaving would be the best practical environmental option (BPEO).

Most sterilization methods can result in the perturbation of the soil's/compost's abiotic properties (physical and chemical). There are also wider ecological implications for some methods and an inconsistent guarantee that the method will actually result in the complete deactivation of the biological biomass (Bank *et al.*, 2008; Egli *et al.*, 2006; McNamara *et al.*, 2003). The key differences between autoclaving and microwaving relate to the manner in which heat is generated throughout the substrate. In autoclaving, heat energy is directly supplied under pressure and permeates from the outside into the centre of the substrate. If sufficient heat and pressure is applied then water can enter a supercritical state with increased solvency potential, although standard autoclaving parameters would not be sufficient to force the water into this supercritical state (Goto *et al.*, 2004). Whereas, in microwaving, a beam of radiowaves supplied by the magnetron causes molecules to oscillate around their polar axis (particularly water molecules) and this intermolecular friction results in the production of heat (Wang *et al.*, 2001). Although most previous studies aimed to avoid significant changes to the physical or chemical properties of the matrix via sterilization, changes were not abated but monitored and assessed in terms of any likely impacts on seedling establishment and plant growth.

The aim of this experiment was to explore the effects of two sterilization methods on the end quality of green waste compost in relation to seedling establishment and plant growth. By sterilizing the compost, the biotic biomass would be eradicated or severely depleted. This could then aid seedling establishment and plant growth, as the competition for nutrients, air and water would be initially reduced (Lawrence & Newell, 1945; McNamara *et al.*, 2003). If this study could show that simple manual sterilization had a positive impact on plant growth then there would be a scientific case for relaxing parts of the PAS: 100 compost standards, relieving operators of the high inputs required to achieve compost sanitization via the thermophilic phase of composting. Based on the evidence that the thermophilic phase does not always guarantee sanitisation, by sterilising the compost after the maturation phase, sanitisation would be achieved as close to the bagging stage as possible and this may be a more attractive management plan to operators.

3.3 Materials and Methods

3.3.1 The Growth Media

Both of the green waste composts (GW1 and GW2) were produced by Wormtech Ltd, using large commercial scale in-vessel aerobic composting in Caerwen, Monmouthshire, UK. GW1 and GW2 were from distinct batches of mature compost of a similar age (approximately 7 months old ± 1 month) and were produced from Local Authority source segregated and household green and woody waste. Monitoring and turning lasted for 16 weeks following which the heaps were left to mature under cover for ≥ 10 weeks. The leading brands of peat based (P) and peat free composts (PF) were used as controls and as such were not subjected to any of the sterilisation treatments. Consequently these controls were to serve as benchmarks from which to measure the effect of the green waste compost treatments on plant growth.

3.3.2 Sterilization Procedures

Sterilization was the main treatment in this study. The non-sterilized green waste composts GW1 NS and GW2 NS were compared with their sterilized counterparts using two sterilization methods – autoclaving (GW1 AC and GW2 AC) and microwaving (GW1 M and GW2 M). P and PF controls were not sterilized. The moisture content was 40 % mass/mass (Wang *et al.*, 1991, Lensi, 1991) for all composts prior to sterilization (±2 %). Batches (3 kg) of compost were autoclaved at a time, at a temperature of 120 °C (393 K) for 50 minutes (pressure of 150 kPa). Compost was microwaved at 850 W for 4 minutes for 1.5 kg of compost at a time, and samples were rotated using a round plate according to the findings of Wang *et al.* (2001).

The compost was allowed to cool in a sterile environment and then sealed and stored at room temperature (18 – 20 °C) for 48 hours before using in the growth trial. Respiration was analysed for 24 hours, immediately after cooling, to assess the level of microbial activity, using a PP systems SR1 soil respirometer (Jones & Kielland, 2002).

3.3.3 The Growth Trial

The plant growth experiment was conducted at the Pen y Ffridd Field Station, Bangor, Gwynedd, North Wales (53°13′N, 4°10′W) using *Lactuca sativa. L.* (lettuce, cultivar Little Gem). Lettuce was used as it is known to be a crop with high water and nutrient demand (Jackson *et al.*, 2002) and so would benefit from the high water holding capacity of the composts and relatively high nutrient availability (see table 3.1). Percentage emergence was measured in a separate germination trial where 10 seeds were sown in the three separate trays for each of the 8 different treatments (GW1 NS, GW1 AC, GW1 M, GW2 NS, GW2 AC, GW2 M) and the two controls (P and PF) (n = 30 per treatment). For the main growth trial, seeds were germinated in John Innes® seed compost and then transplanted into the compost treatments after 3 weeks.

To assess plant growth in the different treatments, leaf chlorophyll content was measured using a SPAD 502 meter (Konica Minolta Inc., Tokyo, Japan). Plant shoot biomass (dry) was calculated following harvest after a total of 40 days in the treatments (growing period = 61 days from sowing). Plants were harvested by cutting the shoot at the soil surface and the samples dried for 24 hours at 80 °C.

3.3.4 Experimental Design

The growth trial was a fully factorial, randomized block design with three replicate blocks (n = 12 per block). The interactive effects of the main factors, sterilization, compost type and block were analysed using a Three-way ANOVA, for the shoot biomass and SPAD readings. A One-way ANOVA within each compost type (GW1 and GW2) was conducted to compare the effect of the two sterilization methods on lettuce biomass, SPAD, % emergence and the compost physico-chemical characteristics and also with the controls peat-based (P) and the peat-free (PF), which were used as a baseline comparison. Significant differences were further analysed with Least Significant Difference (LSD) and significance was defined as $P < 0.05$

unless otherwise stated. All the analyses were done with SPSS v. 16 (SPSS Inc., Chicago, IL).

3.3.5 Compost Analyses

Nutrients from the non-sterilized and sterilized compost media and the controls were extracted using distilled water at a 1:6 w/v ratio (compost-to-distilled water) according to the PAS: 100 guidelines (BSI, 2005). Samples were shaken at 250 rev min^{-1}, centrifuged for 10 minutes at 14,000 *g* and the supernatant was recovered having filtered through Whatman No. 40 filter paper.

Phosphate was determined colorimetrically (Murphy & Riley, 1962). NO_3^- and NH_4^+ were determined using a Skalar SAN^+ segmented flow analyser (Skalar Analytical, Breda, Netherlands) and the method was adapted from Mulvaney (1996). K, Ca and Na were measured using Sherwood 410 flame photometer (Sherwood Scientific, Cambridge, UK).

pH and electrical conductivity (EC) were determined from a compost/distilled water suspension (1:1, v/v), according to the method of Smith & Doran (1996) using standard probes. Water content was determined by drying in an oven at 80 °C for 24 hours. Organic matter content (OM) was determined by loss-on-ignition at 430 °C for 24 hours (Navarro *et al.*, 1991). Total carbon and nitrogen was analysed using a LECO CHN 2000 analyser (LECO Corp., St Joseph, MI). Bulk density was calculated using an adapted method from Blake & Hartge (1986).The physico-chemical properties are summarised in Table 3.1.

3.4 Results

3.4.1 Sterilization Methods

Both autoclaving (AC) and microwaving (M) had a significant reductive effect on the respiration rate in both of the green waste composts (Figure 3.1), however, CO_2 was still emitted at a very low level (< 0.002 mg CO_2 kg^{-1} (DM) s^{-1}). There was no significant difference between the two sterilization methods, in relation to CO_2 evolution ($P \leq 0.001$). However it is clear that there was more variance in the effectiveness of M, particularly in GW1 where the SEM was relatively high.

EC and pH readings were significantly raised by AC and M ($P \leq 0.001$), although there were no significant differences between the two methods. There were no significant differences between the non-sterilized compost and the sterilized counterparts for either compost in relation to bulk density, indicating that there were no major changes in the physical structure of the compost, as a result of sterilization. However, further analyses, such as porosity and pore size distribution would be necessary to provide greater clarity as to the effects on sterilization methods on the physical properties of the composts.

Sterilization had a significant effect on most of the physico-chemical parameters for each of the GW composts (Table 3.1). GW1 AC had a significant increase in the concentration of NO_3^- ($P \leq 0.01$) compared with GW1 NS. GW2 AC rather then GW2 M also incurred the greatest increase in NO_3^- but it was not significant. Both methods of sterilization incurred a rise in NH_4^+ concentrations but more so in the AC samples ($P \leq 0.05$). Total N was increased in GW1 AC and GW1 M but more so in GW1 AC. Total N was greater in the sterilized treatments for GW2 but the increase was very slight in comparison with GW1. P availability was significantly reduced in GW1 AC and GW1 M but more so by AC than M ($P \leq 0.01$).

GW2 AC and GW2 M also had decreased P availability but these results were not significant. Soluble salts (Na, K and Ca) were significantly increased in the autoclaved treatments for both compost types GW1 AC and GW2 AC ($P \leq 0.05$) but despite increased availability of Ca, Na and K in the microwaved treatments, the rise was not significant. Na availability in GW2 M was the only salt to decrease following sterilization.

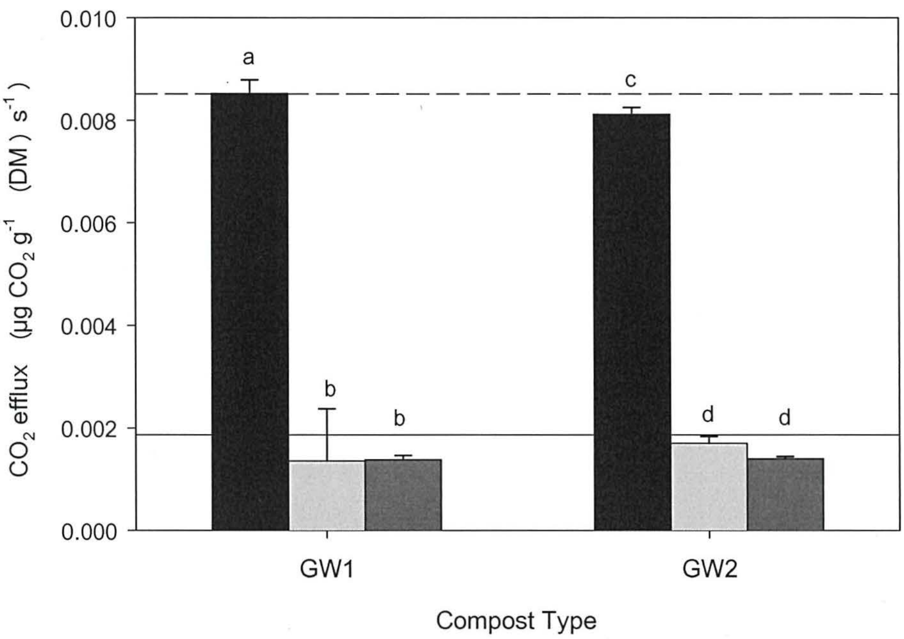


Figure 3.1 CO_2 evolution from non-sterilized (NS), sterilized (microwaved (M) and autoclaved (AC) treatments of two green waste composts (GW1 & GW2). The solid line represents the peat based (P) control and dashed line the peat free (PF) control. Black bars ▮ represent the control – non-sterilized. Light grey bars ▮ represent composts microwaved and dark grey bars ▮ represent composts autoclaved. Values represent mean ± SEM (*n* = 3). Statistical differences are represented by a and b for GW1 and c and d for GW2 with significance at the $P \leq 0.001$ level. The mean respiration rate for the peat-based compost was 0.002 µg CO_2 g^{-1} (DM) s^{-1} (± 0.00005) and for the peat-free compost it was 0.008 µg CO_2 g^{-1} (DM) s^{-1} (± 0.0002).

Table 3.1. **Chemical and physical properties of the non-sterilized and sterilized green waste-derived composts and the standard peat-based and peat-free controls. Non-sterilized composts (NS) were autoclaved (AC) and microwaved (M). Statistical differences between NS, M and AC, within each of the greenwaste composts (GW1 and GW2) were calculated using a One-way ANOVA to monitor the impact of sterilization on the compost physical and chemical properties. Statistical differences ($P < 0.05$) between treatments within each compost type are denoted by $^{a, b}$ and ^{c}.**

	Green waste 1 (GW1)			Green waste 2 (GW2)			Peat based Control (P)	Peat-free Control (PF)
	NS	AC	M	NS	AC	M		
pH	7.65 ± 0.03^{a}	8.10 ± 0.05^{b}	8.16 ± 0.05^{b}	7.87 ± 0.01^{a}	8.30 ± 0.01^{b}	8.22 ± 0.01^{b}	5.37 ± 0.03	6.01 ± 0.01
Bulk density, g cm^{-3}	0.42 ± 0.01	0.42 ± 0.01	0.42 ± 0.01	0.45 ± 0.01	0.45 ± 0.002	0.44 ± 0.01	0.22 ± 0.01	0.46 ± 0.04
Conductivity, mS cm^{-1}	2.9 ± 0.1^{a}	4.2 ± 0.2^{b}	$4.6 \pm .08^{b}$	5.4 ± 0.2^{a}	7.1 ± 0.3^{b}	5.9 ± 0.1^{c}	1.6 ± 0.1	1.18 ± 0.0
Total N, g kg^{-1}	18.8 ± 0.2^{a}	21.0 ± 0.5^{b}	20.4 ± 0.2^{b}	18.0 ± 0.5	18.8 ± 0.1	18.3 ± 0.5	10.3 ± 0.1	7.8 ± 0.1
Available NH_4^+, mg N kg^{-1}	13.1 ± 3.4^{a}	118.6 ± 18.7^{b}	70.8 ± 11.8^{c}	2.9 ± 0.3^{a}	142.5 ± 33.9^{b}	63.2 ± 4.7^{c}	110.6 ± 5.2	23.5 ± 9.4
Available NO_3^-, mg N kg^{-1}	588 ± 21^{a}	760 ± 28^{b}	692 ± 40^{ab}	998 ± 104	1281 ± 117	984 ± 44	940 ± 32	611 ± 145
Available P, mg kg^{-1}	126 ± 2^{a}	88 ± 1^{b}	$54 \pm 8.^{c}$	100 ± 10	85 ± 15	70 ± 17	685 ± 32	287 ± 30
Available K, mg kg^{-1}	3435 ± 129^{a}	5466 ± 212^{b}	4307 ± 406^{ab}	3932 ± 466^{a}	5653 ± 451^{b}	4716 ± 251^{ab}	577 ± 56	437 ± 172
Available Ca, mg kg^{-1}	255 ± 4^{a}	361 ± 26^{b}	339 ± 44^{ab}	326 ± 34^{a}	451 ± 12^{b}	369 ± 16^{a}	161 ± 6	215 ± 20
Available Na, mg kg^{-1}	517 ± 70^{a}	706 ± 11^{b}	520 ± 29^{a}	750 ± 52^{a}	964 ± 65^{b}	689 ± 55^{a}	272 ± 46	169 ± 22

3.4.2 Percentage Emergence

In the germination trial, results were very pronounced in terms of the positive effect of sterilization on percentage emergence (Figure 3.2). GW1 NS had a particularly low emergence rate with only 10 % of seedlings emerging, whereas the sterilized equivalents, GW1 M and GW1 AC, saw a significant increase in the rate of emergence to 87 % and 80 % respectively ($P \leq 0.001$) compared with the unsterilized equivalents. GW2 NS had a rate of 43 % emergence and the microwaved equivalent (GW2 M) saw a significant increase with 80 % of the total seeds sown emerging ($P \leq 0.01$). Despite GW2 AC also incurring an increase in seedling emergence (16 % more emergence than the unsterilized equivalent GW2 NS), this result was not significant.

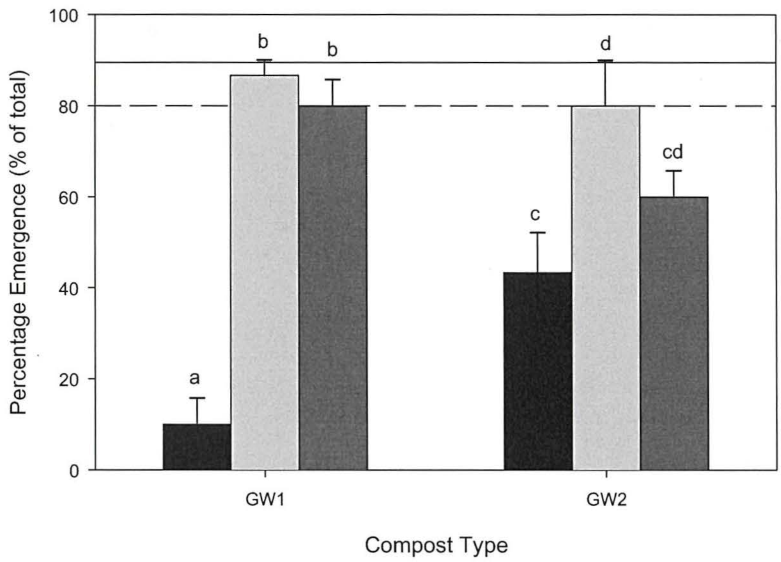


Figure 3.2 Number of seedlings emerging, as a percentage of the total number sown (10 seeds per tray, *n*= 3 trays) in two green waste composts GW1 and GW2. Compost was sterilized by microwaving (M) and autoclaving (AC) and compared with the non-sterilized equivalent. The solid line represents the peat-based (P) control and the dashed line the peat-free (PF) control. Black bars represent the control – non-sterilised. Light grey bars represent composts microwaved and dark grey bars represent composts autoclaved. Values represent mean ± SEM (n = 3). Statistical differences are represented by a and b for GW1 at the $P \leq 0.001$ level and by c and d for GW2 at the $P \leq 0.01$ level. Mean % emergence (n = 3, 10 seeds per tray) for the peat-based compost was 90% (± 6) and for the peat-free compost was 80% (± 6).

3.4.3 SPAD and Chlorophyll content

A fully factorial multivariate analysis revealed that there were no interactions between the main factors – block, compost type and treatment. Treatment was the only significant factor ($P \leq 0.001$). Figure 3.3 shows that the SPAD readings were significantly higher ($P < 0.001$) in the autoclaved samples (GW1 AC and GW2 AC) than for the lettuce grown in the non-sterilized and microwaved composts and controls (GW1 NS, GW2 NS, GW1 M and GW2 M) and both of the controls (P and PF) ($P \leq 0.001$).

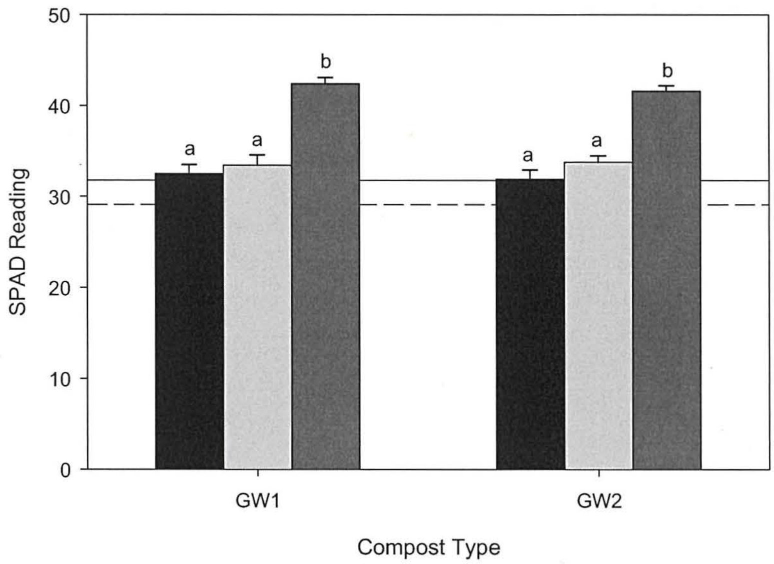


Figure 3.3 SPAD readings mid way through the growing season, for *Lactuca sativa* grown in 2 green waste composts (GW1 & GW2) having been sterilized or not (NS – non-sterilized, AC – autoclaved, M – microwaved). The solid line represents the peat-based (P) control and the dashed line the peat-free (PF) control. Black bars represent the control – non-sterilised. Light grey bars represent composts microwaved and dark grey bars represent composts autoclaved. Values represent mean of 3 leaves per plant ± SEM (n = 12). Statistical differences are represented by a and b at $P \leq 0.001$. The mean SPAD reading (n = 12) for the peat-based compost was 31.8 (± 1.3). The mean SPAD reading for the peat-free compost was 29.1 (± 0.6).

3.4.4 Plant Biomass

There were no interactions between the main factors – block, compost type and treatment - for shoot biomass. The peat-based control (P) produced lettuces with significantly higher biomasses than those grown in the non-sterilized green waste composts (GW1 NS and GW2 NS) (Figure 3.5). However the sterilized counterparts (GW1 AC, GW2 AC, GW1 M and GW2 M) matched the peat-based compost, with no significant differences in biomass found. There were no significant differences in the biomass, between the sterilization treatments, AC and M in either of the green waste composts. GW1 AC and GW1 M saw a significant increase in biomass when compared with the non-sterilized equivalent (GW1 NS) ($P \geq 0.05$). The critical finding was that by sterilizing both GW1 and GW2 the lettuce biomass was comparable to that of the controls, whereas the unsterilized equivalents produced significantly lower lettuce biomass than the controls.

Figure 3.4 Lettuce growth trial prior to harvest

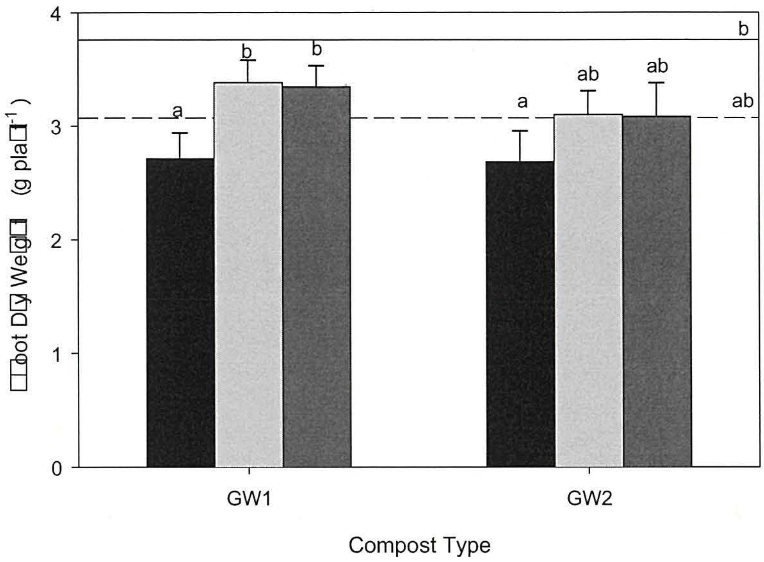


Figure 3.5 **Final above ground biomass of *Lactuca sativa* grown in two green waste composts (GW1 & GW2) that were either non-sterilized (NS), or sterilized via two methods (AC – autoclaved and M – microwaved). Solid and dotted lines represent the control values for the non-sterilized peat based (P) and peat-free (PF) composts respectively. Black bars represent the control – non-sterilised. Light grey bars represent composts microwaved and dark grey bars represent composts autoclaved. *Lactuca sativa* was grown in a complete randomized block design, with three blocks and n = 4 per block. Values represent mean ± SEM (n = 12). Columns with different letters are significantly different at the $P \leq 0.05$ level (LSD). The mean biomass (n = 12) for the peat-based control was 3.76g (± 0.4). The mean biomass (n = 12) for the peat-free control was 3.07g (± 0.2).**

3.5 Discussion

Sterilizing the composts significantly improved the seedling establishment and plant growth of *Lactuca sativa*. There was no significant difference between the sterilization methods – microwaving and autoclaving - with regard to plant biomass. The same could be said for percentage emergence rate, which was significantly improved in sterilized composts but with no real difference between the effectiveness of the two methods. The low rate of CO_2 evolution (Figure 3.1) following both methods of sterilization could be explained by (1) a small proportion of the microbial community survived, or (2) CO_2 was arising from abiotic sources (e.g. carbonate degassing) and (3) residual enzymatic activity (McNamara *et al.*, 2003; Lensi *et al.*, 1991; Powlson & Jenkinson, 1976).

Molecular analyses of the microbial component of the composts, post-sterilization, would have established whether the C was due to microbial respiration (e.g. stable isotope probing). The greater variance in CO_2 evolution for the microwaved treatments (see Figure 1) could have been due to the nature of the linear emission of microwaves resulting in hot and cold spots within the compost (Banik *et al.*, 2006). Indeed Wang *et al.* (2001) showed that the efficacy of sterilization via microwaving depended upon the position of samples in the oven, soil moisture; soil type and the irradiation rate, so it is entirely feasible that pockets of the microbial biomass survived the microwave irradiation. Studies have shown that repeated cycles of autoclaving are necessary to guarantee the complete eradication of the biota in soils (Bank *et al.,* 2008; Herbert *et al.,* 2005; Wolf *et al.*, 1989). Despite adopting every effort to maintain sterile conditions – microbial reinfection and recolonization cannot be ruled out. Stevenson *et al.* (2006) demonstrated that after soil sterilization, with the complete destruction of the biomass (via the addition of $HgCl_2$ or autoclaving), CO_2

flux still occurred, particularly in calcareous soils. The authors hypothesised that around 13 % of total soil respiration is from abiotic CO_2 production. Powlson & Jenkinson (1976) established that via a reduction in the partial pressure of CO_2 during autoclaving, decomposition of HCO_3^- occurred resulting in a flux of CO_2 and the increased precipitation of $CaCO_3$. There would be a concurrent rise in pH (see Table 3.1) with the decomposition of HCO_3^-, as the acid is removed and the reaction is proton-consuming. Residual enzymatic activities following sterilization would also incur a CO_2 efflux (Lensi *et al.*, 1991). However it is unlikely that the decomposition of HCO_3^- was a major cause for the residual CO_2 evolution from the composts and is more likely to be due to incomplete eradication of the microbial biomass or re-infection (e.g. by germination from heat resistant spores or via propagules present in the air which land on and colonize the compost).

A standardised method for sterilizing soils and composts is not available and the preferred methods have always varied and are commonly determined by the facilities available (McNamara *et al.*, 2003; Trevor, 1996). Both methods of sterilization had a significant effect on some of the physico-chemical properties of the compost but to varying degrees (Table 3.1). The fact that autoclaving tended to increase nutrient availability over microwaving, could be due to the combined effect of pressure and heat increasing the solvent ability of the water in the composts (Goto *et al.*, 2008).

The reduction in available P (Table 3.1) following sterilization could in part be explained by the concurrent rise in pH, as generally P is most available between pH 5.5 and 7.0 (Englestad & Terman, 1980). Other factors that could have influenced the solubility of P was the increase in Ca availability, the presence of carbonates and the changes in the structure and solubility of humic substances as a result of heating

(Anderson, 2005). Ultimately the low plant-available P in sterilized and non-sterilized green waste-derived composts could have been a limiting factor in the final shoot biomass, particularly when compared with the P availability in the peat-based control, which produced the greatest lettuce biomass.

The increased concentrations of Ca, Na and K following sterilization were assumed to be from the lysed microbial cells contents. In the autoclaving treatment, the increased solvent ability of the water caused by the combined effect of heat and pressure could have also cleaved the exchangeable cations from the humic substances, whereas this impact is limited in irradiation treatments, accounting for the greater concentration of the exchangeable cations in the autoclaved samples (Lensi *et al.*, 1991; Powlson & Jenkinson, 1976).

The increased availability of NO_3^- and NH_4^+ in all of the sterilized treatments could have played an important role in the improvement of *Lactuca sativa* biomass. The elevated levels of both NH_4^+ and NO_3^- were most likely due to the removal of the ammonium-oxidising and denitrifying bacteria and the release of N from the decomposition of the dead cells following sterilization (Tanaka *et al.*, 2003). The elevated concentrations of NH_4^+ would have also raised pH. The higher concentration of NO_3^- in the autoclaved composts was reflected by the results of the SPAD readings. Despite the arbitrary nature of the SPAD readings there is a positive linear correlation with leaf chlorophyll content and indirectly with the N availability to the plant, as they also correlate linearly with leaf NO_3^- content (Papasawas *et al.*, 2008). However with no significant differences in the end biomass between the two sterilization methods, the autoclaved composts probably experienced nitrogen saturation meaning that the elevated concentrations in the autoclaved sample were surplus to requirement and indeed could have reduced plant growth (Aber, 2004), potentially creating elemental allelopathic conditions (Morris *et al.*, 2009). The leaves of the lettuce grown in the autoclaved composts were observably darker green and these increased levels of chlorophyll content may have had an adverse affect on the leaf flavour. Future work should monitor the impact of higher chlorophyll content on the palatability of leaf crops.

The initial electrical conductivity of GW1 NS was significantly lower than GW2 NS, only serving to highlight the difficulty in applying generic treatment regimes for composts (Noble & Roberts, 2005; Christensen *et al.*, 2002; De Bertoldi, 1983). It is important to note that both GW1 and GW2 were produced at the same site with the same operational procedures and the considerable difference in soluble salt levels most likely pertains to the variability of the feedstock (Francou *et al.*, 2005). The very low percentage emergence (10 %) in the GW1 NS could not be readily explained by the basic physico-chemical parameters measured (table 3.1), as they were all within the acceptable limits for germination and plant growth (BSI, 2005). There is the possibility that the unsterilized compost contained residues of allelopathic chemicals produced by certain plants, particularly invasive species (Zhang *et al.*, 2008) which had the potential to inhibit seed germination and establishment (Prati & Bossdorf, 2004). The heat treatments would have activated the organic carbon and there is evidence that the allelopathic chemicals become immobilized by activated C in the substrate, possibly accounting for the significant rise in percentage emergence in GW1 M and GW1 AC (Prati & Bossdorf, 2004). The electrical conductivity was already particularly high in GW2 NS and so the elevated soluble salt content as a result of the sterilization treatments would generally be expected to have an inhibiting effect on germination (Cheng, 2007). GW2 AC had an average conductivity of 7.1 mS cm^{-1} (Table 3.1) and the percentage emergence for this treatment was comparatively low (Fig 3.2). Research shows that soluble salt concentrations are highest in the top and bottom 2 cms of the substrates, due to the impact of evaporation and leaching. Lettuce was sown in the top 2 cm and so the salt concentration would have been very high in the region where the seeds were sown, which is the most likely reason for the depressed percentage emergence in GW2 AC (Cheng, 2007). The failure of the autoclaved composts to support such a high rate of germination and emergence compared with the microwaved composts could have been due to the development of toxic compounds from the decomposition of organic compounds (Rubin, 1984).

Both GW1 M and GW1 AC saw a significant improvement on shoot biomass when compared with its non-sterilized equivalent. There were clear improvements for the sterilized GW2 treatments but the lack of significance in end biomass is likely to be due to the less suitable physico-chemical properties of this compost. The mechanisms for the improved biomass in the sterilized treatments are either the combined or individual effect of: (1) the reduced competition for nutrients, air and water due to the reduction or elimination of the biota, (2) the decomposition/immobilization of phytotoxic compounds or allelochemicals, (3) the elevated availability of key plant nutrients, and (4) the removal of plant pathogens. It is highly likely that responses to the changes in the composts would be variable from one plant species to the next, so growth trials using more than one plant species would verify whether the improvements are species-specific or not. It is likely that mature composts (> 12 months) would be more resistant to thermal degradation and so the sterilization treatments would have less impact on the chemical properties (Tilston *et al.,* 2002). This study has shown that sterilization of composts can improve the compost quality and could relieve operators of the high inputs required to meet the sanitization regulations. Sidhu *et al.* (2001) found that following prolonged storage sterilized composts had a negligible growth rate of seeded *Salmonella spp.* whereas the non-sterilized compost had significantly higher colonization. This again points to the change in substrate chemistry creating conditions less favourable to the growth of some pathogens and this is supported by other studies (Zmora-Nahum *et al.,* 2008).

Further work should determine whether the sterilization methods employed had destroyed or significantly reduced the microbial biomass. Finally, by harmonising the key macronutrient availability in the composts to reflect the controls (e.g. soluble salts), then a more accurate understanding of the plant growth improvements realised by sterilising greenwaste will be possible. By decreasing parameters, such as electrical conductivity, via wet sieving for example (Veekan *et al.*, 2005) then the green waste compost may demonstrate even higher rates of seedling establishment and plant growth as a result of sterilization.

3.6 Conclusion

The thermophilic phase in the composting process does not always ensure sanitization and ultimately microbes quickly repopulate any soil or compost once retuned back to ambient conditions. Casadei *et al.* (2001) conclude that a simple temperature and aeration regime is not enough to ensure sanitization throughout the compost when feedstock is so highly variable. Consumers may be reassured by the fact that biowaste-derived composts have been manually sterilized, post composting. Sterilizing biowaste-derived composts has the potential to alleviate compost operators of the high inputs required to maintain the regulatory temperatures during the thermophilic phase. Low and variable nutrient availability is a common problem in green waste composts - particularly those of undetermined quality (i.e. non- PAS: 100 certified) (BSI, 2005; Tognetti *et al.*, 2007), so if sterilization can make plant macronutrients more available then this is an attractive amelioration technique. Seedling establishment was significantly improved by sterilizing the compost and despite increased elemental conditions (EC) there was the potential that other allelochemicals, either from microbial exudates or plant decomposition were transformed/deactivated following sterilization.

This study has demonstrated that by sterilizing the green waste composts using microwaving, or autoclaving, both percentage emergence and plant growth for *Lactuca sativa* sp. were improved, matching the leading peat-based brand. Future work should use a greater variety of sterilized composts from different feedstocks and of different maturity. This would help to establish how variable the sterilization effects are in relation to the initial compost characteristics. Key compost properties should be adjusted prior to sterilization to see whether the positive impacts of sterilization on plant biomass are further enhanced.

CHAPTER 4: POTENTIAL PHYTOTOXIC AND ALLELOPATHIC EFFECTS OF COMPOST TEA TREATMENTS ON THE GERMINATION OF FOUR PLANT SPECIES

Pagella S.L., Quilliam R.S., Jones D.L.

4.1 Abstract

Compost tea is increasingly being used as a biocontrol media, providing an environmentally sustainable alternative to pesticides. However, research has shown that both immature and mature composts can exhibit either disease-suppressive or conducive properties. Further, composts may also induce allelopathic or phytotoxic effects. This study specifically investigates the effect of compost age on the potentially phytotoxic and allelopathic effect of two compost teas derived from mature and immature green waste compost. Our aim was to decouple the biological and chemical effects of the compost tea by either microfiltering or autoclaving the compost tea treatments prior to application. Four plants (tomato, cauliflower, wheat and alfalfa) were used in germination bioassays to serve as indicators of the levels of phytotoxicity or allelopathy. Additionally, seeds were either sterilised or non-sterilised to assess the importance of the microbial flora of the seed surface when treated with compost tea. Our results showed that seed responses to compost tea addition were both highly plant species specific and variable. Compost tea prepared from either stable or unstable composts elicited a germination response showing both a phytoxicity and stimulatory response depending upon plant species and pre-treatment conditions. Seed sterilisation had a negative effect on the germination of

tomato seeds in particular, exacerbating the negative effects of compost tea. The factors influencing seed germination in the presence of compost tea proved to be highly interactive and further work is required to understand the cause of potential phytotoxicity and allelopathic effects. Contrary to most accredited compost standards we have shown that using a single plant species fails to adequately reflect the behaviour of compost teas. Consequently, it will be a great challenge for the composting industry to develop quality standards for producing compost teas that are not phytotoxic to a range of plant species.

4.2 Introduction

Since the intensification of agriculture there has been a progressive loss of biodiversity that is unsustainable. Soil biodiversity in particular is seen as being beneficial in crop production for enhancing nutrient cycling and reducing disease incidence. It is not surprising therefore that persistent and recurrent soil borne diseases continue to pose a significant threat to crop production, and hence global food supply (Larkin, 2006). Organic amendments are known to possess disease suppressant properties and are increasingly being used for crop protection as part of an integrated management approach to disease control either instead of or alongside pesticides (Pelosi *et al.*, 2009). The production of composts with consistent and strong disease suppressive properties is a central goal in compost research and to date this has led to a variety of contradictory recommendations. A number of studies have shown that organic amendments for biocontrol purposes must be sufficiently mature to enable them to shift from being disease conducive to suppressive (Veekan *et al.*, 2005; McKellar & Nelson, 2003). In contrast, others have shown that mature composts have very little suppressive properties compared with the immature equivalent (Zmora-

Nahum *et al.,* 2008; Danon *et al.*, 2007). However, additional studies have shown that waste-derived compost that has not undergone prolonged curing is likely to be highly phytotoxic and so despite suppressing soil borne pathogens it could generate phytotoxic conditions, limiting plant growth and survival (Aslam *et al.*, 2008; Woods *et al.*, 2009).

Compost tea is a solution prepared by steeping compost in water for a given period of time (Hagaag & Saber, 2007). It can be prepared with additional aeration or non-aerated and can be used both as a nutrient feed for plant growth but also as a disease suppressant owing to the high level of microbial consortia present in the tea (Hargreaves *et al.*, 2009). Compost tea is increasingly used in agriculture as a biocontrol foliar spray to reduce the survival of fungal pathogens in certain crops (Hagaag & Saber, 2007; Yohalem *et al.*, 1994). There is a need for the composting industry to develop high value products if it is to become sustainable (Favoino & Hogg, 2008) and compost tea falls into this category. Research carried out on compost tea has focussed mainly on composts that are rich in nutrients, e.g. those produced from manures, although there is less understanding of the efficacy of compost tea produced from green waste materials (Noble *et al.*, 2006). Waste management companies are increasingly looking for diversification of compost markets to ensure financial sustainability, whilst adding value to their end product; therefore an understanding of the potential uses for compost tea is both timely and important.

4.2.1 Compost stability and maturity/phytotoxicity

Whilst some studies have identified that compost must reach sufficient maturity and stability to go from disease conducive to disease suppressive (Veekan *et al.*, 2005; McKellar & Nelson, 2003; El-Masry *et al.*, 2002; Tilston *et al.*, 2002), other studies demonstrate that the unstable, immature composts provide greater suppression of specific pathogens (Zmora-Nahum *et al.*, 2008; Danon *et al.*, 2008) owing to the more diverse microbial population inhibiting pathogens via antagonistic mechanisms (Sidhu *et al.*, 2001). However, the concept of compost stability and maturity continues to be a controversial issue in the composting industry and compost research (Gazi *et al.*, 2007). In the UK Quality Assurance Protocol, PAS:100 (BSI, 2005) stability and maturity are synonymous terms and are used interchangeably to describe microbial activity. In this study compost stability is used as an index of microbial activity alone, as defined by Bernal *et al.* (1998) and Komilis & Tziouvaras (2009); whereas maturity is taken to mean the degree of compost phytotoxicity and a variety of standard parameters are proffered as maturity indices (Bernal *et al.*, 1998; Said-Pullicino *et al.*, 2007; Komilis & Tziouvaras, 2009). Stability is generally assessed via respiration rate or the self-heating ability of the compost in standard conditions (Lasaridi & Stentiford, 1998). Phytotoxicity typically declines with composting time (Ward, 2005) and in the UK the standard test is a germination bioassay, but this alone will not identify the precise cause of phytotoxicity (Said-Pullicino *et al.*, 2007). The composting process and the initial characteristics of the parent material exert a considerable influence on the level of phytotoxic substances in both unstable and stable composts (Danon *et al.*, 2007) and so time of composting is not a useful measure of maturity. Phytotoxic substances can result in a reduction in the seed germination rate and plant growth and often increase plant vulnerability to infection (Danon *et al.*, 2007). Typically a variety of organic compounds, for example short and long-chain fatty acids, coexist in immature compost. In isolation the concentrations of these individual acids (e.g. formic, acetic, tannic) are usually lower than the minimum level required to have any negative effects on germination and plant growth but synergistically these volatile acids can create phytotoxic conditions (Wu & Ma, 2001).

Bailey & Lazarovits (2003) recognised that water soluble allelopathic chemicals can affect plant growth and germination. The release of secondary plant metabolites with allelopathic properties during a certain stage of a plant life cycle is an effective adaptive mechanism, as these allelopathic biocompounds can inhibit the germination, growth and survival of other plants, microbes and herbivorous predators (Wardle *et al.*, 1998). Conversely allelochemicals can promote germination and growth in some plant species (Singh *et al.*, 2010). If green waste feedstock contains plant species that contain allelopathic chemicals, then partially decomposed less stable compost would be more likely to contain allelochemicals than well decomposed stable compost as the chemicals are less likely to have been transformed as decomposition progresses. Allelochemicals are frequently secondary plant products such as phenolics, terpenoids or organic acids which are transformed into germination or growth inhibitory components.

Organic amendments are becoming increasingly integrated into techniques for the control of weeds and pathogens (Noble *et al.*, 2006) and so it is important for the composting industry to understand at what stage compost stability and maturity is attained, and the subsequent level of disease suppression at this stage. Borrero *et al.* (2006) demonstrated that microbial communities in more highly decomposed compost utilised more plant sugars and carbohydrates, whereas in less decomposed compost microbial communities mainly metabolised carboxylic acids and polymers. These authors demonstrated that compost containing microbial communities that tended to consume more complex C sources (i.e. those in the immature compost), were more able to suppress *Fusarium* wilts. Several reports have suggested that less stable composts offer greater disease suppression (Danon *et al.*, 2007; Zmora-Nahum *et al.*, 2005), therefore it is essential to identify whether the concomitant lack of maturity also generates phytotoxic conditions, which may negate any disease suppressive benefits. Some studies have shown that microbial-mediated disease suppression is likely to occur close to the plant seed surface and within hours of germination (Berg *et al.*, 2005; Chen & Nelson, 2008; Green *et al.*, 2007; McKellar *et al.*, 2003). However, the effect of compost tea on the microbial consortia of the seed coat has never before been studied.

While many studies have reported that compost stability and maturity are important factors in determining disease suppression, little or no research has considered the importance of compost stability or maturity prior to the production of compost tea. Previous studies have often focused on optimising the 'steepage' time for producing compost tea (Ingram & Millner, 2007; Wood *et al.*, 2009), however, as many of the organic compounds associated with phytotoxicity are in the dissolved fraction (Said-Pullicino *et al.*, 2007; Veekan *et al.*, 2005) it is conceivable that compost tea produced from unstable and/or immature composts will concentrate the phytotoxic chemicals.

In this study we have used an immature (biochemically unstable) green waste compost and a mature (highly stable) green waste compost to make produce two contrasting compost teas. A germination bioassay was used to assess the level of phyto-inhibition for four plant species, cauliflower, alfalfa, tomato and wheat. The industry standard germination bioassay for solid compost uses only tomato (BSI, 2005), therefore a selection of economically important plant species were chosen to provide as broad a range of responses to potential variables within the compost tea, as possible. These species contained examples of monocotyledons (wheat) and dicotyledons (tomato), mycorrhizal and non-mycorrhizal plants and a legume (alfalfa). Cauliflower is from the family Brassicaceae, and therefore represents the non-mycorrhizal species (Glenn *et al.*, 1985); brassicas are generally considered pioneer species and are therefore more likely to be tolerant of a wide range of environmental conditions, e.g. heavy metal contamination (Xiong, 1998). Tomato has expensive water and nutrient demands and is very susceptible to a wide range of diseases (De Curtis *et al.*, 2010); wheat was chosen as it is economically important worldwide (El-Wakeil *et al.*, 2010); a legume was chosen due to its N fixing potential (Aranjuelo *et al.,* 2011).

By microfiltering or autoclaving the compost tea treatments the biological and chemical effects on seed germination were assessed. In addition, by using sterilised and non-sterilised seeds, we have investigated the importance of the microbial community inhabiting the seed coat during the early stages of germination.

4.3 Methods and Materials

4.3.1 Compost Tea Production

Green waste compost for both teas was obtained from a commercial composting facility operated by Flintshire County Council, North Wales (53°17′14′′N 3°12′11′′W). The green waste was collected from civic amenity sites and municipal kerbside collection schemes within Flintshire. Both composts were produced aerobically in large windrows (30m long by 3 m high) and were turned approximately every 10 days. Stable compost was sampled from 4 bags to account for any differences between batches and samples were mixed prior to tea production. Stable compost was approximately 52 weeks old having been bagged at 20 weeks since composting began and then stored at 20 °C. Unstable compost was sampled from 4 random,spatially separate points in the windrow and mixed before tea production. Unstable compost was approximately 6 weeks old from shredding and was considered unstable due to the self heating ability of the compost in standard conditions (Lasardi & Stentiford, 1998). Temperatures generated throughout the windrow at the time of sampling were in excess of 60 °C. Respiration was analysed for 24 hours to assess the level of stability using a PP systems SR1 soil respirometer (Jones & Keiland, 2002). Both composts had a starting moisture content of 40 ± 5 %. Compost tea was produced by mixing a 50:50 v/v ratio of compost to distilled water and shaking at 200 rev min^{-1} for 7 days. Although this process allowed minimal aeration the product would not qualify as an aerated compost tea (Hagaag & Saber, 2007). Compost teas were then filtered through sterile muslin cloth. The teas were centrifuged for 10 minutes at 14,000 *g* and the supernatant filtered through Whatman No. 540 filter paper. The supernatant was then centrifuged and filtered again due to the high volume of organic precipitates retained after the first filtering. At this stage the tea was referred to as filtered (F) compost tea and still retained its microbial consortia. Aliquots of the filtered tea were aseptically microfiltered (M) through a 0.25 μm filter (Millipore Corporation) to remove all of the biota, or autoclaved (A) at a temperature of 120 °C (393 K) for 20 minutes (pressure of 150 kPa), which in addition to sterilising the tea had the added effect of perturbing the chemistry due to the combined effects of heat and pressure.

There were therefore 7 treatments in total, unstable filtered tea (CTuF) and stable filtered tea (CTsF), unstable microfiltered tea (CTuM) and stable microfiltered tea (CTsM), unstable autoclaved tea (CTuA) and stable autoclaved tea (CTsA), plus a control of autoclaved deionised water (C).

4.3.2. Germination Bioassay

Winter wheat, *Triticum aestivum* (var. Claire), cauliflower, *Brassica oleracea* (var. All the year round), alfalfa, *Medicago sativa*, and tomato *Lycopersicon esculentum* (var. Moneymaker) were selected to provide a wide range of plant types and responses. Seeds were germinated in Petri dishes that contained a 90 mm (diameter) disk of sterile filter paper (Whatman No. 2) soaked with 3 ml of compost tea (or autoclaved water control). Half of the seeds were sterilised by soaking in a 30% household bleach solution with 1 drop of Tween® 20 for 5 minutes (wheat for 20 minutes) and then rinsed 3 times with sterile DI water and allowed to air dry in sterile conditions. Ten seeds (20 for alfalfa) were placed on the soaked filter paper for each treatment and three replicate dishes were used for each treatment. Petri dishes were kept in the dark in a controlled growth cabinet, at 20 °C and 70% RH. To account for any potential temperature gradients, samples were fully randomised at each sampling stage.

The seed germination percentage and root growth were evaluated after all the control-treated seeds had germinated. Seeds were considered to have germinated once the radicle was ≥ 1 mm. Mean Germination Time (MGT) (Salehzade *et al.*, 2009), Radicle Vigour (RV) and the speed of Germination Index (GI) (Maguire, 1962) were calculated by using the following equations:

MGT = $\Sigma Dn / \Sigma n$

(D = days since germination trial began, *n* = number of seeds germinated on the day)

RV = % germination * main root (radicle) length (mm)

$$\textbf{GI} = \frac{\text{number of seeds germinated}}{\text{days since first count}} + \ldots\ldots + \frac{\text{number of seeds germinated}}{}$$

days on final count

(+……..+ represents the accumulative effect of calculating the number of seeds germinated/days since first count, for each day between the start and finish).

MGT is delayed when conditions are inhibitive to seed germination and so if compost tea conditions were phytotoxic or allelopathic then MGT would be higher (Salehzade *et al.*, 2009; Singh *et al.*, 2010). Conversely, if allelochemicals promoted germination in any of the plants then the MGT would be lowered (Singh *et al.*, 2010). Both radicle vigour and the rate of germination (GI) would be higher if compost tea treatments were non-phytotoxic (Singh *et al.*, 2010).

Statistical analysis (Two-way ANOVA with Tukey pairwise comparison and paired t-tests) was performed using the computer package Minitab 15 (Minitab Inc., State College, PA).

4.3.3 Compost Tea Analyses

Nutrients from the three compost tea treatments were analysed for key plant macronutrients. Phosphate was determined colorimetrically (Murphy & Riley, 1962). NO_3^- and NH_4^+ were determined using a Skalar SAN$^+$ segmented flow analyser (Skalar Analytical, Breda, Netherlands) and the method was adapted from Mulvaney (1996). K, Ca and Na were measured using Sherwood 410 flame photometer (Sherwood Scientific, Cambridge, UK). pH and electrical conductivity (EC) was determined according to the method of Smith and Doran (1996) using standard probes. Total soluble carbon (TC), total soluble organic carbon (TOC), and total soluble nitrogen (TN) was determined using a Shimadzu TOC-TN analyser (Shimadzu Corp., Kyoto, Japan). Total soluble organic nitrogen was established by subtracting the sum of NH_4^+ and NO_3^- from TN. Phenolic concentrations were analysed using the Folin-Ciocalteu reagent, calibrated with a phenol standard according to Swain and Hillis (1959). Differences between individual teas were analysed using a one-way ANOVA in SPSS (v.16) SPSS Inc., Chicago, IL, and significant differences were calculated using Tukey's post hoc test with $P < 0.05$ unless otherwise stated.

4.4 Results

4.4.1 Compost Stability

The young, unstable compost, (CTu) at the time of sampling was self generating temperatures of 60°C (± 5°C) and had a respiration rate of 28 (± 2.3) mg CO_2 g^{-1} VS d^{-1}. The older, stable compost, (CTs) was at an ambient temperature (18 °C) and had a respiration rate of 2.7 (± 0.6), CO_2 g^{-1} VS d^{-1}.

4.4.2 Compost tea physico-chemical properties

The combined factors, i.e. the level of stability of the teas (CTu and CTs) and the treatment type (F, M and A), had significant effects on the compost tea physico-chemical properties. Autoclaving caused the greatest changes compared with the filtered equivalents (Table 4.1). NO_3^- was significantly higher in all 3 compost tea treatments made from stable compost (CTs). Microfiltering significantly reduced NO_3^- concentration compared with CTsF and CTsA. There were negligible concentrations of NO_3^- in all three of the unstable compost teas (CTu). NH_4^+ was highest in the autoclaved mature tea CTsA, significantly lower in CTsF whilst CTsM, CTuF, CTuM and CTuA all had comparably low NH_4^+ concentrations. Levels of P were generally low although the unstable filtered tea (CTuF) was significantly higher than the rest. Autoclaving both the unstable and the stable compost teas resulted in significantly lower levels of P than in their filtered counterparts.

The starting pH was slightly lower in the unstable tea, CTu, but increased following both microfiltering and autoclaving. Autoclaving, but not microfiltering, also increased the pH of the stable compost tea. Electrical conductivity (EC) was almost twice as high in stable compared with the unstable compost teas. Following autoclaving of CTs there was a significant rise in the levels of soluble salts. Levels of K were significantly higher in the autoclaved unstable tea (CTuA) compared with all other teas apart from CTsF. Na and Ca were significantly higher in the stable compost tea compared with the unstable tea. Ca was also significantly lowered by microfiltering in both tea types. Total carbon was significantly higher in CTu compared with CTs. Microfiltering the CTu significantly lowered water soluble carbon (WSC) and water soluble organic carbon (WSOC). The main factor influencing TN was compost stability, with significantly higher concentrations in the stable tea (CTs). Total phenolics were highest in CTuA, followed by CTuF and then CTuM, all of which were significantly different from one another. The phenolics in the stable tea treatments were significantly lower than the unstable counterparts but there was no difference between the stable compost tea types.

Table 4.1 **Chemical properties of unstable (CTu) and stable (CTs) greenwaste compost teas. The compost tea was filtered (F) and then either microfiltered (M) or autoclaved (A). Statistical differences between CTuF, CTuM and CTuA, CTsF, CTsM, CTsA were calculated using a One-way ANOVA. Values with different letters within each row differ significantly from each other (one-way ANOVA and Tukey multiple comparison test, $P < 0.05$). Values are the mean ($n = 4$) ± SEM. Statistical differences between treatments are denoted by $^{a, b, c, d}$ and ^{e}.**

	Unstable Compost Tea (CTu)			Stable compost tea (CTs)		
	Filtered (F)	Microfiltered (M)	Autoclaved (A)	Filtered (F)	Microfiltered (M)	Autoclaved (A)
Available NO_3^-, mg N l^{-1}	0.9 ± 0.3^{c}	<0.01	0.5 ± 0.1^{c}	278 ± 6^{ab}	270 ± 2^{b}	287 ± 1^{a}
Available NH_4^+, mg N l^{-1}	11.4 ± 1.1^{c}	10.2 ± 0.2^{c}	9.9 ± 0.1^{c}	18.7 ± 1.6^{b}	10.3 ± 0.3^{c}	54.7 ± 0.5^{a}
Available P, mg l^{-1}	3.9 ± 0.1^{a}	2.9 ± 0.1^{b}	2.0 ± 0.1^{e}	2.6 ± 0.01^{c}	2.5 ± 0.03^{c}	2.3 ± 0.02^{d}
Conductivity, mS cm^{-1}	2.8 ± 0.1^{c}	2.7 ± 0.1^{c}	3.1 ± 0.1^{c}	4.8 ± 0.1^{b}	4.8 ± 0.1^{b}	5.3 ± 0.1^{a}
pH	7.44 ± 0.01^{e}	8.17 ± 0.03^{c}	9.06 ± 0.03^{a}	7.62 ± 0.03^{d}	7.56 ± 0.01^{d}	8.31 ± 0.01^{b}
Available K, mg l^{-1}	3302 ± 101^{bc}	3088 ± 150^{c}	4575 ± 106^{a}	3976 ± 116^{ab}	3358 ± 20^{bc}	3840 ± 224^{b}
Available Ca, mg l^{-1}	474 ± 4^{c}	361 ± 6^{d}	482 ± 9^{c}	991 ± 37^{a}	852 ± 10^{b}	1048 ± 18^{a}
Available Na, mg l^{-1}	28.7 ± 2.0^{ab}	24.9 ± 0.1^{a}	33.7 ± 1.0^{b}	45.5 ± 1.1^{c}	55.8 ± 1.6^{d}	59.9 ± 1.0^{d}
Water soluble C, mg l^{-1}	623 ± 9^{a}	458 ± 14^{b}	644 ± 15^{a}	220 ± 2^{c}	206 ± 23^{c}	236 ± 7^{c}
Water soluble organic C, mg l^{-1}	489 ± 8^{a}	343 ± 9^{b}	537 ± 13^{a}	192 ± 2^{c}	181 ± 20^{c}	212 ± 2^{c}
Water soluble N, mg l^{-1}	75.7 ± 0.9^{b}	42.8 ± 1.1^{b}	75.1 ± 2.7^{b}	478 ± 6.5^{a}	448 ± 50^{a}	501 ± 7.3^{a}
Total phenolics mg C l^{-1}	31.3 ± 0.6^{b}	22.4 ± 1.1^{c}	34.6 ± 1.1^{a}	6.9 ± 0.1^{d}	7.1 ± 0.1^{d}	9.1 ± 0.2^{d}

4.4.3 Phytotoxicity of stable and unstable compost teas

The mean germination time (MGT) for tomato and wheat germinated in the compost tea treatments was significantly longer than in the water control ($P < 0.001$), whereas the differences were negligible for alfalfa and cauliflower (Figures 4.1). There was a significant interaction between the main effects for tomato only ($P < 0.001$). The filtered stable tea (CTsF) had a significantly lower MGT for tomato, but there was no difference between the stable autoclaved (CTsA) and microfiltered teas (CTsM). In cauliflower the autoclaved stable treatment (CTsA) had a significantly lower MGT than CTsF, CTuF and CTsM.

There were significant differences in the speed of germination index (GI) between the water control and the compost tea treatments for wheat and tomato (except for CTsF) (Figure 4.2). GI was reduced in all treatments for wheat and tomato, apart from the stable filtered compost (CTsF) in tomato, which was not significantly different from the control. For cauliflower seeds the GI in autoclaved stable tea (CTsA) was significantly higher than the control. There were no significant interactions of the main effects for all of the plants, except tomato where the interaction was significant ($P < 0.001$). Cauliflower GI was significantly higher than the filtered and microfiltered equivalents in both the autoclaved unstable tea (CTuA) and the stable autoclaved tea (CTsA) the latter being more pronounced. The stable filtered tea (CTsF) and the autoclaved unstable tea (CTuA) both had significantly higher GI in tomato, conversely the unstable filtered tea (CTuF) and the autoclaved stable tea (CTsF) had significantly lower GI. Wheat and alfalfa saw no differences for GI amongst the compost tea treatments.

Radicle vigour (RV) of alfalfa seedlings was significantly higher in the control than all of the tea treatments (Figure 4.3). Similarly for tomato the RV in the control was higher than all but the stable filtered tea (CTsF). In wheat seedlings the RV was significantly lower in all of the teas compared to the water control, apart from the microfiltered unstable tea (CTuM). There was a significant interaction of the main effects for RV, for all four plant species. In alfalfa the RV for the unstable autoclaved tea (CTuA) was significantly lower than the other compost tea treatments. For tomato the stable filtered tea (CTsF) had a RV that was significantly higher than for the other compost tea treatments, however, the stable autoclaved tea (CTsA) had a significantly lower RV when compared with all the other teas. Wheat RV was significantly higher in the microfiltered unstable compost tea (CTuM) but significantly reduced in both autoclaved teas.

Figure 4.1a Alfalfa

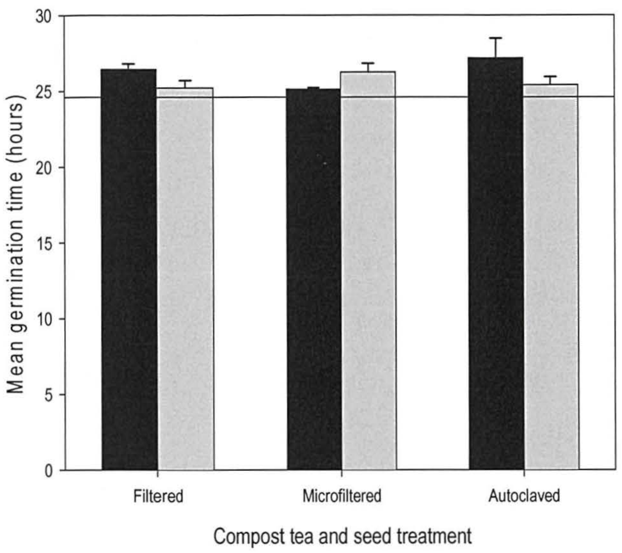


Figure 4.1b Cauliflower

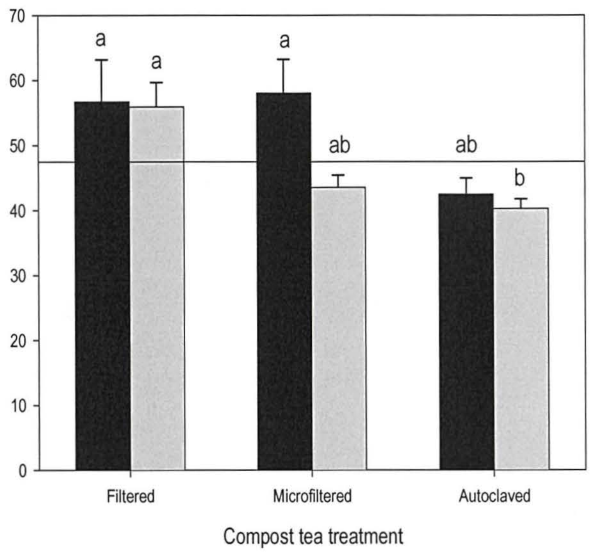


Figure 4.1c Tomato

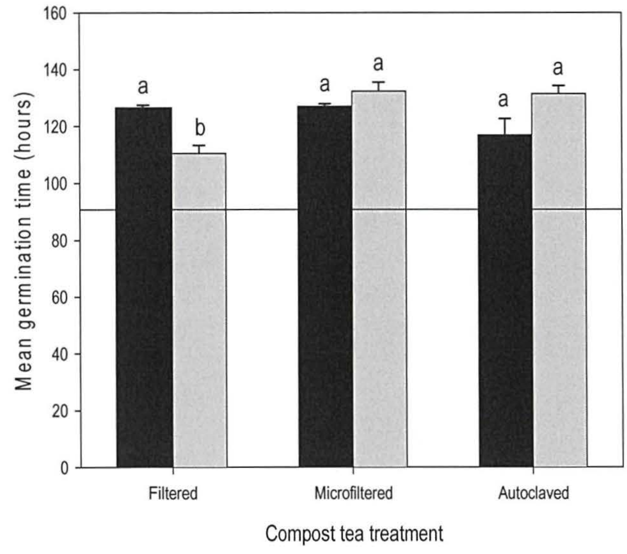
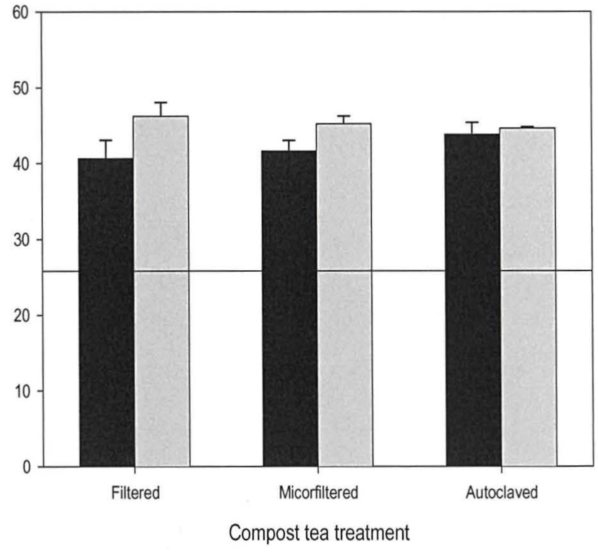


Figure 4.1d Wheat



Figures 4.1 Mean Germination Time (MGT) of alfalfa (a), cauliflower (b), tomato (c) and wheat (d) seeds germinated in compost tea made from immature (unstable) compost (black bars) or mature (stable) compost (grey bars). Solid line represents the water control. Bars with different letter codes differ significantly from each other (two-way ANOVA and Tukey multiple comparison test, $P < 0.05$. Data points are the mean of 3 replicates + SEM). Mean MGT for water controls were: alfalfa = 24.6 hours (± 0.4); cauliflower = 47.4 hours (± 3.9); tomato = 91 hours (± 0.6); wheat = 25.8 hours (± 0.4).

Figure 4.2a Alfalfa

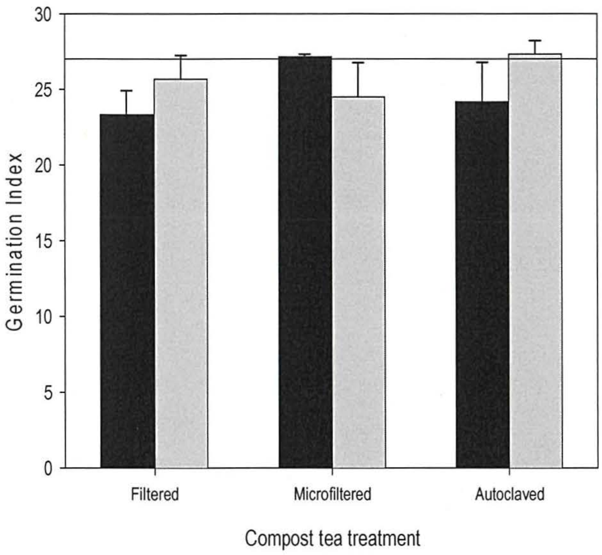


Figure 4.2b Cauliflower

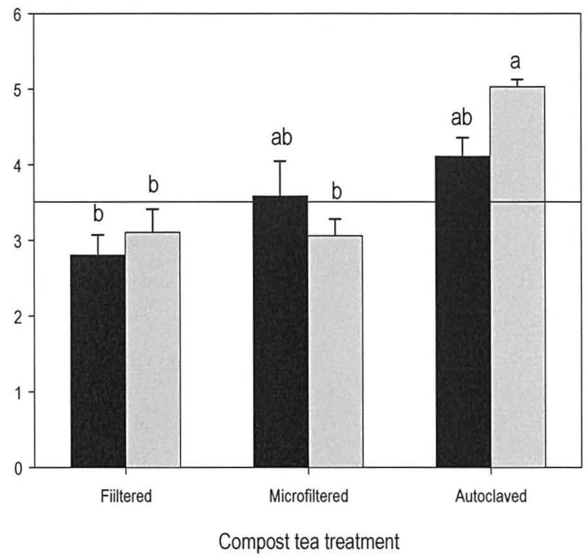


Figure 4.2c Tomato

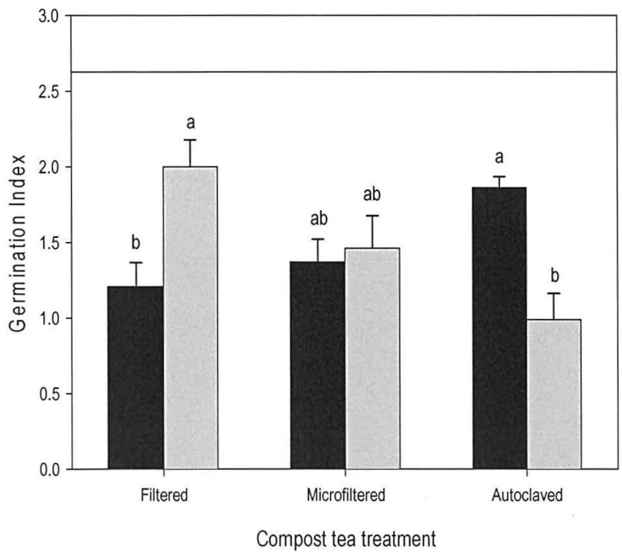
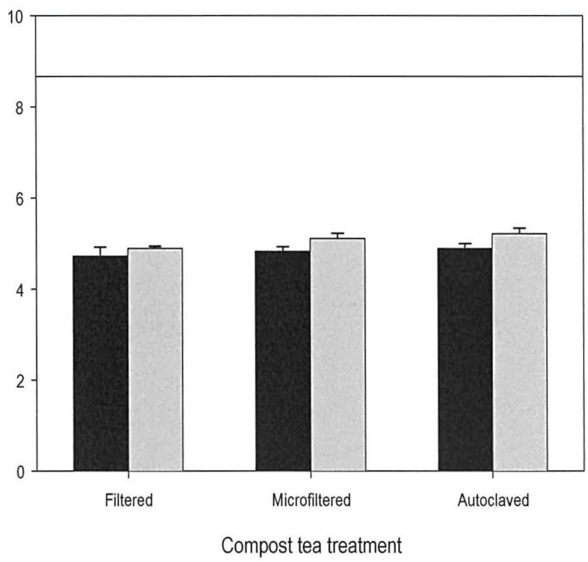


Figure 4.2d Wheat



Figures 4.2 Germination Index (GI) of alfalfa (a), cauliflower (b), tomato (c) and wheat (d) seeds germinated in compost tea made from immature (unstable) compost (black bars) or mature (stable) compost (grey bars). Solid line represents the water control. Bars with different letter codes differ significantly from each other (two-way ANOVA and Tukey multiple comparison test, $P < 0.05$. Data points are the mean of 3 replicates + SEM). Mean GI for water controls were: alfalfa = 27.1 (±0.4); cauliflower = 3.51 (± 0.5); tomato = 2.65 (± 0.04); wheat = 8.67 (± 0.3).

Figure 4.3a Alfalfa

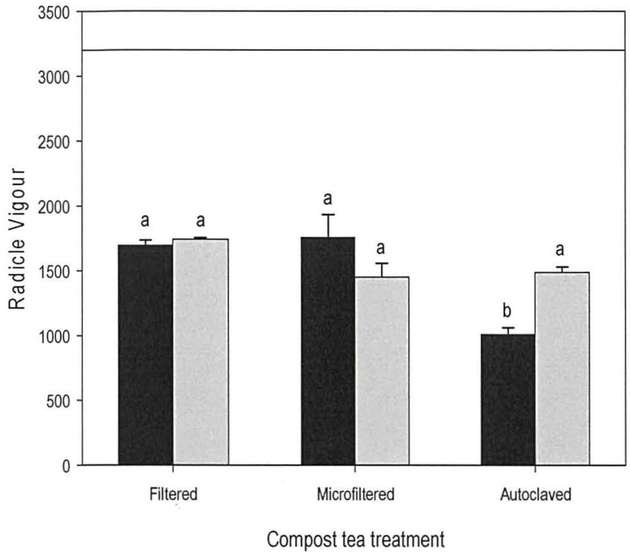


Figure 4.3b Cauliflower

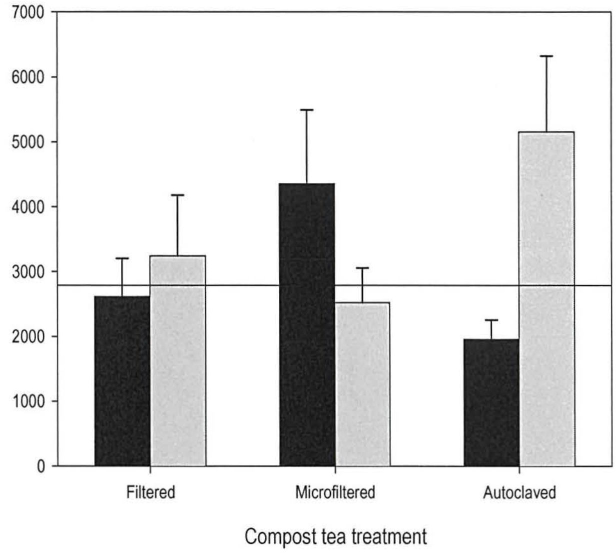


Figure 4.3c Tomato

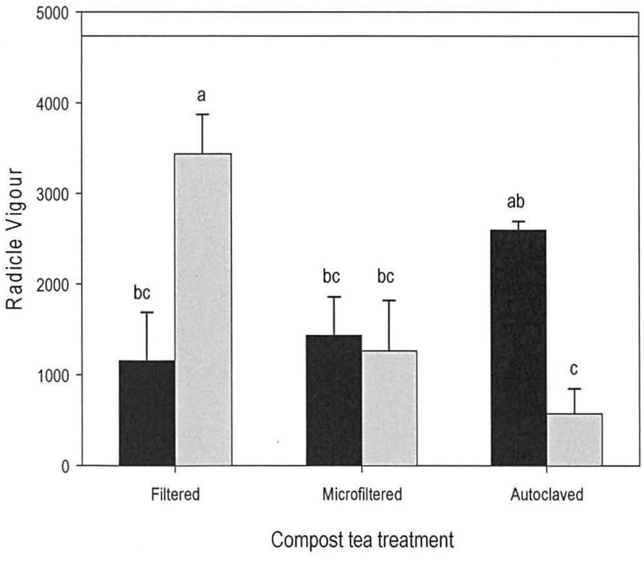
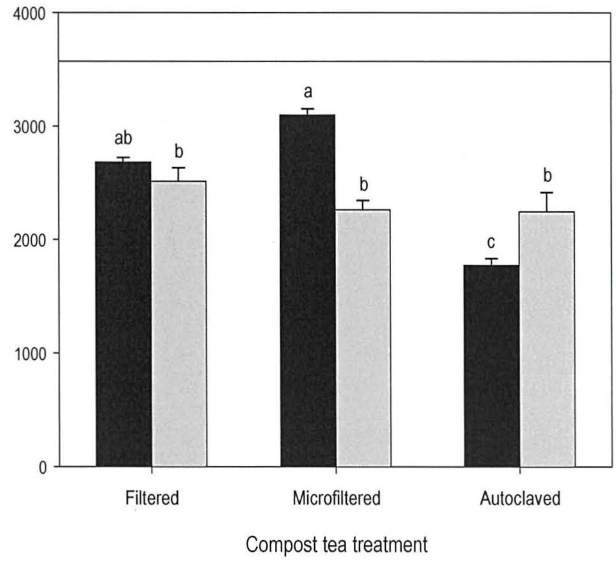


Figure 4.3d Wheat



Figures 4.3 Radicle Vigour (RV) of alfalfa (a), cauliflower (b), tomato (c) and wheat (d) seeds germinated in compost tea made from immature (unstable) compost (black bars) or mature (stable) compost (grey bars). Solid line represents the water control. Bars with different letter codes differ significantly from each other (two-way ANOVA and Tukey multiple comparison test, $P < 0.05$. Data points are the mean of 3 replicates + SE). Mean RV for water controls were: alfalfa = 3198 (± 304); cauliflower = 2787 (± 996); tomato = 4735 (± 210); wheat = 3570 (± 312).

4.4.4 *Seed Sterilisation*

For alfalfa, seed sterilisation significantly lowered the MGT for the unstable filtered and microfiltered teas (CTuF and CTuM) (Table 4.2).

Cauliflower MGT was improved by sterilising the seeds, with significant increases seen in the CTuM, CTsF and CTsM treatments. Conversely, sterilising seeds in the unstable autoclaved treatment (CTuA) significantly increased the MGT for cauliflower. Speed of germination (GI) for cauliflower seeds was significantly improved for sterilised seeds sown in CTsF and yet decreased for sterilised seeds sown in CTuF.

Sterilising tomato seeds germinated in the water control (not in the compost teas) had a highly significant, negative impact on all three measurements (MGT, GI and RV). For sterilised seeds in the compost teas-the GI was significantly reduced in all of the treatments apart from CTuF, where the negative effect of sterilising seeds on GI was more marginal. RV was higher in all of the compost tea treatments for unsterilised seeds with significant improvements in CTuM, CTuA, CTsF CTsM and CTsA compared with sterilised seeds; the greatest difference of effect between non-sterilised and sterilised tomato seeds, was in both the filtered compost teas (CTuF and CTsF).

Sterilised wheat seeds germinated in either CTuM or CTuA had significantly delayed MGT ($P < 0.05$). The GI was significantly reduced by treating sterilising seeds with all of the unstable tea types, and for CTsF and CTsA, whilst the RV tended to be greater for non-sterilised seeds following treatment with compost tea.

represent the mean of 4 replicates ± SE. Significant differences between sterilised (SS) and non-sterilised (NS) seeds for each treatment were calculated (t-test * = $P \leq 0.05$; ** = $P \leq 0.01$; *** = $P \leq 0.001$)

Plant	Compost tea	Mean Germination Time (MGT)		Germination Index (GI)		Radicle Vigour (RV)	
		SS	NS	SS	NS	SS	NS
ALFALFA	Control - water	24.7 ± 0.4	24.6 ± 0.4	18.5 ± 0.8	19.0 ± 0.3	2601 ± 73	3198 ± 304
	CTuF	24.8 ± 0.3*	26.4 ± 0.4	17.7 ± 0.7	16.2 ± 0.9	2157 ± 210	1696 ± 38
	CTuM	24.6 ± 0.2*	25.1 ± 0.1	17.2 ± 1.2	18.5 ± 0.0	1868 ± 211	1758 ± 174
	CTuA	26.2 ± 0.5	27.2 ± 1.3	17.3 ± 0.2	17.0 ± 1.3	1034 ± 114	1009 ± 52
	CTsF	26.0 ± 0.6	25.2 ± 0.5	17.3 ± 0.6	17.5 ± 0.9	1758 ± 142	1741 ± 13
	CTsM	24.7 ± 0.5	26.2 ± 0.6	19.5 ± 0.0	16.8 ± 1.3	1491 ± 146	1450 ± 106
	CTsA	24.8 ± 0.1	25.4 ± 0.5	18.5 ± 0.3	18.7 ± 0.4	1412 ± 54	1487 ± 44
CAULI-FLOWER	Control - water	46.1 ± 10	47.4 ± 3.9	5.40 ± 0.8	3.51 ± 0.5	3388 ± 654	2419 ± 996
	CTuF	60.2 ± 9.8	56.7 ± 6.5	2.25 ± 0.5*	2.81 ± 0.3	2146 ± 1091	2614 ± 592
	CTuM	32.4 ± 3.8*	58.0 ± 5.2	5.00 ± 0.7	3.58 ± 0.5	3653 ± 982	4357 ± 1136
	CTuA	58.5 ± 1.4**	42.5 ± 2.5	2.97 ± 0.4	4.11 ± 0.2	1814 ± 477	1958 ± 295
	CTsF	39.4 ± 2.6*	55.9 ± 3.8	5.39 ± 0.2*	3.27 ± 0.5	3778 ± 324	3241 ± 939
	CTsM	31.2 ± 1.4**	43.5 ± 1.9	4.11 ± 0.7	3.00 ± 0.3	2941 ± 829	2521 ± 534
	CTsA	45.7 ± 5.6	40.2 ± 1.5	4.48 ± 0.5	5.03 ± 0.1	3865 ± 984	5156 ± 1167
TOMATO	Control -water	116 ± 4.6**	90.8 ± 0.7	1.84 ± 0.1***	2.63 ± 0.1	1974 ± 268***	4735 ± 210
	CTuF	127 ± 3.1	127 ± 0.9	1.28 ± 0.2	1.27 ± 0.2	762 ± 77	1155 ± 533
	CTuM	134 ± 3.6	127 ± 0.9	0.64 ± 0.1*	1.37 ± 0.2	187 ± 63*	1436 ± 423
	CTuA	133 ± 3.3	117 ± 5.7	0.89 ± 0.1**	1.86 ± 0.1	186 ± 48***	2595 ± 97
	CTsF	119 ± 1.9	110 ± 2.9	1.23 ± 0.2*	2.00 ± 0.2	1124 ± 140**	3436 ± 435
	CTsM	138 ± 3.5	132 ± 3.2	0.33 ± 0.1**	1.46 ± 0.2	40.7 ± 30*	1265 ± 555
	CTsA	135 ± 5.0	132 ± 2.8	0.11 ± 0.1**	0.99 ± 0.2	8.51 ± 4.8*	808 ± 245
WHEAT	Control - water	29.4 ± 1.8	25.8 ± 0.4	7.78 ± 0.4	8.67 ± 0.3	2693 ± 163	3570 ± 312
	CTuF	43.1 ± 3.2	40.6 ± 2.4	3.50 ± 0.3*	4.72 ± 0.2	1360 ± 276**	2685 ± 40
	CTuM	47.4 ± 0.8*	41.6 ± 1.4	3.67 ± 0.4*	4.83 ± 0.1	1176 ± 182***	3100 ± 52
	CTuA	41.5 ± 0.7	43.8 ± 1.6	3.67 ± 0.3*	4.89 ± 0.1	995 ± 28*	1775 ± 59
	CTsF	40.7 ± 3.3	46.2 ± 1.8	2.89 ± 0.2***	4.89 ± 0.1	953 ± 142***	2515 ± 118
	CTsM	43.2 ± 1.5	45.2 ± 1.0	4.44 ± 0.2	5.11 ± 0.1	1724 ± 73*	2265 ± 80
	CTsA	30.2 ± 1.3***	44.6 ± 0.2	4.67 ± 0.2*	5.22 ± 0.1	2304 ± 160	2247 ± 169

4.5 Discussion

The aim of this study was to examine the potentially phytotoxic effects of compost teas prepared from both stable (mature) and unstable (immature) compost. Various studies have shown that raw, partially composted, and fully composted organic amendments (solid state) can be either disease suppressive or conducive depending on the pathogen being targeted. Consequently there is an increasing interest in developing compost teas for use as a sustainable form of biocontrol in crop production and horticulture, e.g. as foliar and soil borne pathogen suppressants in addition to being used as nutrient feeds. However, relatively little research has focused on the maturity or stability of the compost used to produce compost tea, in relation to either phytotoxicity or disease-suppression.

4.5.1 Compost maturity and stability

In this study we have made a clear distinction between the terms stability and maturity. Traditionally, the time since composting began, was taken as a sufficient indicator of the level of stability (Wood *et al.*, 2009) but in this study we followed the standard test for stability, being rate of respiration (BSI, 2005). Our results established that the old mature compost could be classed as stable (CTs) and the younger immature one as unstable (CTu).

The germination bioassay aimed to establish the level of maturity of the compost in relation to phytotoxicity; however the specific cause of phytotoxicity cannot be determined from a bioassay alone (Said-Pullicino *et al.*, 2007). Firstly, the impact of the 6 compost tea treatments on mean germination time (MGT), speed of germination index (GI) and radicle vigour (RV) differed quite markedly for each plant, indicating that the level of seed sensitivity to the compost tea physico-chemical properties and the microbial consortia was highly species specific, or random. This brings into question whether the bioassay in the PAS 100 (BSI, 2005) for maturity/phytotoxicity is robust, as only tomato is tested. Coincidentally the tomato response to the stable filtered compost tea (CTsF) demonstrated the least harmful conditions. Compared with all other compost tea treatments CTsF was not significantly different to the water control in all measures apart from the MGT. The PAS: 100 maturity test states that "any significant negative response in the compost-amended medium shown during the four-week test period is taken to indicate phytotoxic factors in the compost which are not indicated in other tests", (p33, BSI, 2005). It is important to note that this standard test has been developed for solid substrates and so a direct comparison with tea is not possible. However, following this logic, the germination bioassays for both the stable and unstable compost teas indicated both were phytotoxic, owing to significant differences in some plant responses compared with the water control. However, the results were not uniform and no clear trend was apparent.

Examining the compost tea properties may indicate potential parameters for phytotoxicity. Maturity is linked (but not directly correlated) with factors such as a low respiration rate (< 8 mg CO_2 g^{-1} VS d^{-1}), and increased nitrification (Wood *et al.*, 2009), both of which were characteristic of the stable compost tea used in this study. The high level of inorganic N in CTs was indicative of highly mature compost with significantly greater concentrations of NO_3^- than in CTu and a much more favourable NH_4^+: NO_3^- ratio in terms of plant growth. NO_3^- availability can also have a significant effect on the germination and growth of seedlings and is often used as a germination promoter (Salisbury & Ross, 1992). Even so, it is arguable that NO_3^- availability was very high in the CTs and in the long term this could have had an elemental allelopathic effect in terms of indirectly inhibiting plant growth or by favouring the growth of plant pathogens (Morris *et al.*, 2009). Elemental allelopathy refers to allelopathic conditions owing to too high concentrations of chemicals such as nitrates (Morris *et al.*, 2009)). NO_3^- availability has also been shown to reduce infection and lower the MGT which contributes to the disease resistance in seedlings (Salisbury & Ross, 1992) although without further analyses there is not clear evidence of this in the data.

Soluble salts (> 2.0 mS cm^{-1}) can inhibit germination either by generating an osmotic potential and preventing water uptake by the seed, or via the toxic effects of the salt ions on the germinating seed (Khajeh-Hosseini *et al.*, 2003). In the stable compost CTs, the EC was ≥ 4.8 mS cm^{-1} for all three compost teas, with the highest being the autoclaved tea, CTsA, at 5.3 mS cm^{-1}. Therefore it is logical to question whether the very high EC of the stable compost tea was phytotoxic or immature (given that in this study these terms are used synonymously, whereas stability does not necessarily mean it will be mature/non-phytotoxic). With this evidence in mind, seeds treated with the stable compost tea may have suffered reduced germination rates and a negative impact on seedling growth due to the relatively high levels of salt. However, as the results for MGT, GI and RV were so varied within each plant species, it is unclear whether the soluble salt content in CTs was responsible for inhibiting germination. Owing to the interactive and dynamic processes, which influence seed germination, it is likely that while the high EC in CTs had some negative impacts on some of the plants species, this was counteracted by other parameters that were more favourable to germination.

Significant differences in the radicle vigour of germinating tomato seedlings demonstrate the complexity of the biological and chemical mechanisms occurring within the compost tea. The microfiltered stable tea (CTsM) incurred a reduction in tomato radicle vigour compared with the filtered equivalent (CTsF) even though this tea had a comparable EC to CTsF. This suggests that the removal of the biota had a negative effect on the seedling response, which was increased by autoclaving-where both the EC rose and the microbial biomass was destroyed. So both the removal of the compost tea microbial biota, coupled with changes in the chemistry had the greatest negative impact on the root growth for tomato. Cauliflower seeds however, had the highest radicle vigour in CTsA, which had the highest EC and was comparable with the radicle vigour of seedlings in the control treatment. In general, the cauliflower germination response in the compost teas was better than the control particularly in CTsA-where the GI was significantly higher than the water control. Both the removal of the compost tea biota and the change to the chemistry following autoclaving had a positive effect on cauliflower germination although the opposite was seen for tomato. It would appear that conditions in the stable compost tea (CTsF) that were phytotoxic to cauliflower, were transformed during autoclaving, rendering them less phytotoxic. Autoclaving CTuF increased the GI and MGT for cauliflower but had a detrimental impact on the RV, which could feasibly have been due to the significant rise in pH. Rapid radicle emergence may not necessarily infer that emergence is more successful, but retarded germination time is often either an indication of phytotoxicity or the presence of allelochemicals (Demir & Mavi, 2004). It is possible that the high soluble salt content could have lowered the margin of difference between CTu and CTs in relation to other phytotoxic properties.

4.5.2 Compost tea treatment effects

Importantly microfiltering the compost tea treatments significantly lowered the WSC and WSOC. Apart from the decline in NH_4^+ concentrations microfiltering the stable tea had no other significant effects on the nutrient composition. Subsequently the main effect of microfiltering may not be solely the removal of the microbial consortia, and the resultant drop in WSOC may relate to a reduction in allelopathic conditions (Aslam *et al.*, 2008). Therefore it is hard to determine whether the removal of the compost microbes was responsible for the reduced MGT of cauliflower seeds treated with CTsM or whether it was due to a reduction in WSOC.

4.5.3 Plant Response

The response of tomato seeds supported the hypothesis that stable compost is more likely to be mature (i.e. not phytotoxic) (Wood *et al.*, 2009), which was demonstrated by the germination response in CTsF not being significantly different to the water control. Whereas the response of tomato seeds in the unstable equivalent CTuF, was indicative of phytotoxic conditions and so could be considered immature. Interestingly, when CTuF was autoclaved (CTuA) MGT, RV and GI were all improved. The microfiltered CTuM saw no improvement demonstrating that the immature tea CTuF was positively affected by transforming the chemical conditions via heating but not by the removal of the microbial biomass or lowering the WSOC alone. Slawiński *et al.* (2004) found that UV irradiation and oxygen transformed toxic allelopathic phenolics into low molecular weight water soluble products and it is likely that the combined effect of steam and pressure would also produce similar effects, thus potentially remediating allelopathic effects. Microfiltering CTuF did not improve GI and RV for tomato and so the phytotoxicity in the unstable compost is likely due to the chemical conditions of the tea for tomato and not due to any negative effects of the biota, e.g., nutrient competition (McNamara *et al.*, 2003) and predation of seed colonizers that mediate/trigger germination (Chen & Nelson, 2008).

The response of cauliflower seeds however, indicated that compost tea produced from unstable compost was not necessarily as detrimental to seed germination and growth. Cauliflower response measured via MGT in CTsM, CTuA and CTuA were comparable to the control, implying that if we apply the standards for assessing maturity with solid state compost (BSI, 2005) these tea treatments would be considered mature, i.e. not phytotoxic. MGT for cauliflower was highest for CTuF, CTsF and CTuM. As MGT was high in CTuM but not CTuA, it is fair to conclude that the removal of the biota was not the defining mechanism for the improvements seen in the autoclaved teas, rather some change to the chemistry rendering them non-phytotoxic. Removing the microbes was not enough for the unstable tea to reach mature conditions for germination. However, long-term studies are needed to test these allelopathic effects under natural conditions especially since phytochemical characters of the soil and the microbial activity can mitigate or intensify this effect.

4.5.4 Seed Sterilisation and Seed Colonizers

Tomato seeds were most affected by sterilisation. Seeds germinated in the distilled water control performed significantly better (MGT GI and RV) using non-sterilised seeds supporting the claim that seed colonizing bacteria play a key role in the rate and success of germination (Berg *et al.,* 2006). In all of the compost tea treatments tomato growth was affected by removing microbes from the seed surface; the germination response was slower and less vigorous, with the exception of CTuF.

This is most likely due to the high numbers and diversity of microbial populations in the untreated CTu competing and predating the seed colonizers.

Wheat seeds also largely demonstrated that sterilising seeds had a negative effect in the bioassay however MGT was longer for the non-sterilised seeds sown in the stable teas (CTsM), which could be explained by the high level of NO_3^- availability lowering the microbial reliance on seed exudates thus slowing the time taken for seeds to germinate. The negative effects of seed sterilisation were most pronounced for the compost tea treatments, compared with the control, with significant differences for RV in all treatments, except CTsA.

Cauliflower seeds saw a variety of responses but significant improvements were seen for the sterilised seeds apart from for CTuF, CTuA and CTsA. Webb *et al.* (2009) found that imbibing wetland sawgrass seeds with hypochlorite tended to increase the germination rate owing to speeding up the decomposition of the seed husk. They found that the phytohormone abscisic acid (ABA) was present in the husk and the allelopathic impact of this chemical was repressed by imbibing with water or hypochlorite. The fact that sterilised cauliflower seeds sown in CTuF saw a decline in the germination response (GI) could be due to the seed coating and associated microbes being an important element in remediating negative effects of the microbial consortia contained in this compost tea produced from unstable compost. Ultimately CTuF, CTuA and CTsA were consistently more favourable towards non-sterilised cauliflower seeds, which indicate that the microbes coating the seed surface were important in facilitating germination in these compost tea treatments. Clearly further work is required to understand the mechanisms behind these responses.

4.6 Conclusions

Compost maturity and stability are important indicators of the quality of waste-derived composts. This study has highlighted the need for greater clarity concerning the use of these key terms related to compost quality as stable compost and compost tea can clearly produce phytotoxic conditions. There were several key conclusions from this study:

- With highly variable responses to the compost tea produced from stable and unstable composts it corroborates the findings of Wood *et al.* (2009) that the length of time for composting and maturation is not sufficient as a single parameter, to guarantee the transformation of phytotoxic compounds or removal of allelopathic conditions.
- Using one plant species as a predictive test for maturity/phytotoxicity is insufficient, as this study has shown that assessing phytotoxicity via a bioassay will elicit different responses from different plant species. The PAS: 100 (BSI, 2005) stipulates that tomato is used as a bioassay to assess the maturity of composts. This study has shown that tomato is sensitive to the level of stability of composts used to make compost tea but that plant response is highly species specific with regard to maturity.
- By autoclaving the unstable compost tea the germination results were significantly improved, suggesting the presence of allelochemicals that were transformed by heating. Autoclaving increases the availability of salts and so the initial salt concentrations of the compost used to make the tea should be $<$ 2.0 mS cm^{-1} to ensure that autoclaving doesn't increase the salt concentrations to phytotoxic levels. However, increasing the compost curing time could also

stimulate elemental allelopathic conditions, in a similar way to the increased levels of EC and NO_3^-, which may be inhibitory to some plant species.

- The attempt to decouple the effect of the biota and the chemistry on germination was only partially achieved by microfiltering and autoclaving the compost tea treatments but the results were highly variable among plant species. Microfiltering the unstable compost tea resulted in a reduction in the levels of water soluble carbon (WSOC). In contrast, autoclaving the bioactive teas resulted in a rise in WSOC that could have been phytotoxic to some plants (tomato, wheat and alfalfa) particularly in the CTuA treatment, as WSOC was already initially high. In some cases removing the biota from the unstable compost tea improved germination rates but microfiltering significantly lowered WSOC and this result cannot be ignored as phytoxicity has been linked to this fraction in other studies (Wu & Ma, 2001; Said-Pallucino *et al.*, 2007).
- Seed sterilisation had a negative impact on tomato germination, most particularly in the control treatments confirming that the seed coating and colonizing microbes are an important part of the germination process. The unstable filtered tea was least affected by sterilisation of tomato seeds indicating that microbes in this highly bioactive tea dominated the seed surface and initial seed microbes were less important in this medium.

Overall, this study has demonstrated the highly interactive effects of the biological and chemical conditions of compost teas on germination responses. This highlights the difficulty in the providing quality assurance for green waste compost derived products and with developing predictive tests for phytotoxicity and allelopathy. A systems approach for investigating this topic is imperative.

CHAPTER 5: WETTING AND DRYING EFFECTS IN PEAT, GREEN WASTE COMPOST AND VERMICOMPOSTS

Pagella S L. and Jones D L.

5.1 Abstract

Understanding the water dynamics of the plant-soil system is a key part of managing irrigation, fertilisation and developing sustainable peat replacements in horticulture. Generally, soilless plant growth media are less resistant to changes in temperature, water content and solute concentration, than field soil owing to the limited volume of substrate in containers (Michel, 2010; Naasz *et al.*, 2005). The primary aim of our study was to determine the rate of drying from saturation for peat, green waste-derived compost and vermicompost, followed by an assessment of any subsequent changes in their re-wettability and hydrophobicity. We investigated whether the vermicomposted green waste (V) amended with different feedstocks (sewage sludge and paper pulp), had greater physical stability compared with aerobically composted green waste (G) and the extent to which this was due to differences in drying/wetting effects. Composts were dried to varying percentages of container capacity and shrinkage was recorded. Re-wettability of the composts was monitored following water addition from above, and capillary rise from beneath. The level of hydrophobicity was measured at each drying stage using the 'Water Drop Penetration Method' (WDPT). The water release curve (WRC) and the level of hysteresis between wetting and drying cycles were ascertained using a Dewpoint (WP4) water potentiometer. Our results showed that feedstock significantly

influenced the wetting and drying effects in vermicompost. Overall, the vermicomposts made from green waste and green waste and sewage sludge (70: 30 v/v) had greater resilience to shrinkage than peat and green waste compost, when dried to < 5% of container capacity. Quasi-irreversible changes to the physical structure were most pronounced in the vermicompost made from paper pulp and green waste. Water repellency in peat was more persistent as samples were increasingly dried. Hydrophobicity in all three vermicomposts was much less persistent than in green waste compost and peat. Improving green waste-derived compost via vermicomposting appears to have positive results when considering the water characteristics of growing media, but feedstock has a significant effect. Overall, our results suggest that difference in decomposition processes exerts a strong influence on the wettability, WRC and physical stability of composts. The transformation of organic matter via vermicomposting results in composts that are less likely to become hydrophobic, particularly when dry, whereas peat and standard aerobic compost become increasingly more hydrophobic with drying. Lastly the physical stability of vermicomposts is strongly influenced by the initial feedstock.

5.2 Introduction

Owing to increased ecological and climate change concerns, large quantities of waste-derived compost are being produced throughout the UK and are progressively replacing peat as soilless growing media (Zaller, 2007). Further, maximising water efficiency in agricultural and horticultural systems is increasingly important as ambient temperatures rise. Approximately 70% of water withdrawals throughout the world are accounted for by crop irrigation (IPCC, 2007) and as water supply is becoming limited in a growing number of regions in the world, it is increasingly subject to tighter regulations and higher costs (Bastida *et al.*, 2007; Boudreau *et al.*, 2009). The water dynamics in soilless cultures are particularly interesting as the limited amount of substrate volume often results in large fluctuations in oxygen and nutrient availability and wetting and drying effects can be magnified (Naasz *et al.*, 2005).

Growing media are typically selected for their aeration or water retention properties and peat tends to be favourable in both these respects (Michel, 2010; Raviv *et al.,* 2005). However, compared with the body of literature on the biological and chemical properties of composts, there is comparatively little information available on the water characteristics of various peat-free growing media (Boudreau *et al.*, 2009). Added to this, water retention and physical properties of composts receive comparatively little attention in the UK PAS: 100 Compost Standard (BSI, 2005) used for assessing compost quality. However, there is raised awareness that the physical environment is an important determinant and regulator of biological activity in soils (Young & Crawford, 2004) and soilless media. As peat is still largely viewed as a superior growing media compared with waste-derived composts (Michel, 2010; Raviv, 2005) efforts to convince professional and amateur horticulturalists that green waste-derived compost can become a viable peat replacement have only gained marginal success in recent years (Walker *et al.*, 2006).

Vermicomposting waste is becoming an increasingly popular way of adding value to waste–derived composts (Roberts *et al.*, 2007). Research has shown it can result in a high quality plant growth media (Atiyeh *et al.*, 2000; Arancon *et al.*, 2004) owing to the presence of biologically active growth promoting substances, although this is not a consistent finding and its effects can be plant species-specific (Roberts *et al.*, 2007). However, there are no studies available looking at the re-wettability of vermicomposts following desiccation. Vermicompost is formed via the ingestion and rapid breakdown of organic matter due to the action of the earthworm gut and associated mesophilic micro-organisms (Atiyeh *et al.*, 2000). Unlike standard compost, the waste does not undergo a thermophilic phase during decomposition (Farrell & Jones, 2009). This means the high lignin content contained in green waste feedstocks is less likely to be subject to mineralization, as with composting, which can reduce porosity and increase bulk density (Raviv *et al.*, 2005). Another added benefit to compost operators is that waste transformation via vermicomposting can be significantly quicker than with conventional composting (Ndegwa & Thompson, 2001. Vermicompost tends to have a lower bulk density and higher porosity than aerobic composts and greater homogeneity in the finished product (Zaller, 2007). A wide range of biodegradable waste can be vermicomposted (Roberts *et al.*, 2007) and as feedstock exerts an influence on the end quality of compost (Ward, 2005) the effect of feedstock was studied.

5.2.1 Water Repellency (WR)

Peat and green waste compost are prone to hydrophobicity (Rainbow & Nelson, 1998) and this phenomenon has affected their effectiveness as a growing media and consequently their marketability (Urrestarazu *et al.*, 2008; Rainbow & Wilson, 1998). The wettability of growing media describes how easily and evenly water infiltrates the porous materials, once they have dried to a certain degree. How a drop of liquid configures itself on a solid surface depends upon the relationship between the intermolecular surface tensions of the solid and the liquid (cohesion), and the interfacial tension between them both (adhesion) (Doerr, 1998).

Although several mechanisms explaining water repellency in a wide range of soils have been studied (Oostindie *et al.*, 2008) it is widely accepted that an hydrophobic organic coating of soil particles frequently results in a level of water repellency (Oostindie *et al.*, 2008; Dekker, 2005; Hallet *et al.*, 2001 Czarnes *et al.*, 2000). The level of desiccation at the surface of soils is often an important factor influencing the persistence of hydrophobicity (Doerr, 2000) and as dryness increases, water repellency tends to extend to the sub-surface (Dekker, 1996). Mineral materials tend to be hydrophilic (Michel, 2010) and so the greater mineral content found in composts and vermicomposts could result in a less persistent hydrophobicity, compared with highly organic peat (Jones *et al.*, 2009). Tracking the level of wettability over a range of drying levels is an important aspect of quality assurance in soilless media. Hydrophobicity in containerised media naturally incurs an increase in water use and often results in irreversible damage to the physical properties of the media and subsequent plant growth (Michel, 2010).

5.2.2 Physical Stability

In this study physical stability relates to the substrates' resilience to shrinkage and swelling. Besides having suitable physical properties to start with, growing media need to provide a stable physical environment for plant growth (Michel, 2010). Repeated cycles of drying and rewetting, or excessive desiccation (Oostindie *et al.*, 2008) can have a significant effect on the physical stability of substrates owing to shrinkage and swelling (Gerbhardt *et al.*, 2010). Sphagnum peat has typically been favoured for its ability to return to its original state following drying (Michel, 2010; Caron *et al.*, 2005) but this may only be the case in less well decomposed peats (Michel *et al.*, 2002). Conversely, a lack of maturity in waste-derived composts can incur significant changes to physical properties over time due to the effects of further decomposition (Michel, 2010; Komilis & Tziouvaras, 2009). Irreversible changes to the volume of substrate due to shrinkage, can lead to a loss of aeration or water holding capacity and as such the material should be considered physically unstable (Michel, 2010) and unsuitable as containerised media without appropriate remediation.

This overall aim of this study was to establish whether vermicompost would behave more favourably than green waste compost when subjected to increasing drying/wetting. Peat was used as a positive control media to asses the comparative performance of other treatments. A secondary aim was to assess whether the feedstocks used influenced the wettability and physical stability of the vermicompost following different levels of desiccation.

5.3 Methods and Materials

5.3.1 Composts

There were 3 media treatments in total, namely (1) horticultural peat, (2) vermicompost and (3) green waste-derived compost. Three distinct batches of peat were used - peat 1 (P1), peat 2 (P2) and peat 3 (P3) were sourced from lowland peat lands in Ireland and were supplied by Murphy's Scotts, Ltd. The commercial extraction of all three peat composts occurred between 6 and 12 months prior to the start of the study. Peat composts were formed from sphagnum moss, ensuring good hydraulic conductivity and capillary rise properties compared with sedge peat (Caron *et al.*, 2005). Peat particle size was fine (< 10 mm) and no nutrients or wetting agents were added. Three green waste-derived composts were used; G1, G2 and G3 – all of which originated from an in-vessel aerobic composting plant operated by Wormtech

Ltd., Monmouthshire (51°37'N, 2°46'W). Each batch was fully stable and mature (12 months old each, ± 1 month). Particle size was screened to < 20 mm. Vermicomposts were produced using *Dendrobaena veneta* in controlled conditions using the same worm stocking densities and operation according to the method outlined by Roberts *et al.* (2007). V1 was made from green waste (finished compost particle size < 10 mm), V2 was produced from a 70-to-30 (v/v) mix of green waste and paper pulp (finished compost particle size > 0.05 mm) and V3 was produced from a 70-to-30 (v/v) mix of green waste and biosolids (finished compost particle size < 10 mm). For the vermicompost, no physical processing occurred prior to use in this study meaning that any observed differences were due to the effect of the biodegradation incurred by the worms and associated microbial consortia and the feedstock. Composts were used as received – i.e. particle size were not homogenised as grinding the material would have altered the biochemical composition of peat compost (Flaig *et al.*, 1975). Each compost type (peat, green waste and vermicompost) was replicated by 12, with four random samples taken from 3 distinct batches. Each batch was considered a separate treatment owing to the spatially separate nature of the composts. Waste-derived compost are typically heterogeneous in their properties and so this study was interesting in ascertaining whether there were significant differences within treatments of the same types of compost (green waste, peat and vermicompost), as well as between different types.

5.3.2 Container Capacity

Composts were packed into 600 cm^3 polypropylene plant pots possessing ten 4 mm diameter drainage holes in the bottom. Composts were wet up to container capacity (CC) following an adapted method from Cassel & Nielsen (1987). Pots were sub sampled and dried in an 80° C oven for 24 hours to obtain gravimetric and volumetric water content. CC is the equilibrium water content after complete wetting followed by free drainage via the holes at the bottom of the container (Cassell & Nielsen, 1987) and differs from field capacity (FC). In CC, water freely drains from the macropores into air at the base of the pot where the soil water pressure is 0 kPa.

5.3.3 Compost Characteristics

pH and electrical conductivity (EC) were determined from a compost/distilled water suspension (1:1, v/v), according to the method of Smith & Doran (1996) using standard probes. Water content was determined by drying in an oven at 80 °C for 24 hours. Organic matter content (OM) was determined by loss-on-ignition at 430 °C for 24 hours (Navarro *et al.*, 1991). Total carbon and nitrogen was analysed using a LECO CHN 2000 analyser (LECO Corp., St Joseph, MI). The bulk density was measured using an adapted method of Blake & Hartge (1986). The composts were placed in a 1000 cm^3 container and a force of 260 Pa was applied by way of a steel disc, so as to ensure a uniform packing pressure. It was not in the interest of this study to homogenise the particle size or bulk density between treatments, as one of the key differences between peat-based and green waste composts at the point of purchasing, is bulk density (Papfotiou *et al.*, 2005). Compost physical properties are summarised in Table 5.1.

5.3.4 Drying and Rewetting Cycles

The composts at container capacity in the 600 cm^3 polypropylene plant pots were placed in an environment-controlled growth room, with a constant relative humidity of 70 % and temperature of 20 ° C (± 2 ° C) and no light. The gravimetric water content was calculated for each individual pot at container capacity – from this, the fraction of the mass that was water in each pot was calculated over time. As mass was lost, the percentage of water remaining from the original mass of water at container capacity was tracked. There were 5 sub-sets each containing 4 replicates of the 9 treatments (n = 36 per sub-set).

Set 1 – pots dried to 80% of the total water held at container capacity

Set 2 – pots dried to 60% of the total water held at container capacity

Set 3 – pots dried to 40% of the water held at container capacity

Set 4 – pots dried to 20% of the water held at container capacity

Set 5 – pots dried to 5% of the water held at container capacity (this tends to be when mass loss via drying reaches equilibrium – i.e. no further mass loss unless dried in an 80ºC oven).

Using the same pots for each drying point would have effectively measured the effects of successive wetting and drying cycles, which was not the objective of this study. Pot position was randomised at each measurement to account for any temperature gradients in the growth room. The rate at which the composts dried and the various differences in effect that drying rate had on both rewetting and hydrophobicity were of key interest in this study.

As pots reached the specific level of mass loss depending on the calculated percentage of container capacity lost, they were rewet by rapid addition of DI water to the top of the composts (Fig. 5.1). High infiltration rates were generally observed due to a combination of hydrophobicity and preferential flow pathways, resulting in the water flushing straight through the media and exiting at the base of the pots at which point it was captured by individual basal containers. The water level in the basal container was retained, until no more of the water could become re-absorbed into the compost. Following the first application of water, 20 ml of leachate was removed from the basal trays for physicochemical analyses (stored at -10 °C) and this accounted for by replacement with an additional 20 ml of water. Pots were loosely covered with cling film to prevent water loss via evaporation. The rate of rewetting was calculated as the percentage of water re-absorbed. Rewetting was terminated once there was no significant mass gain (< 1 g) for a minimum of two consecutive hours.

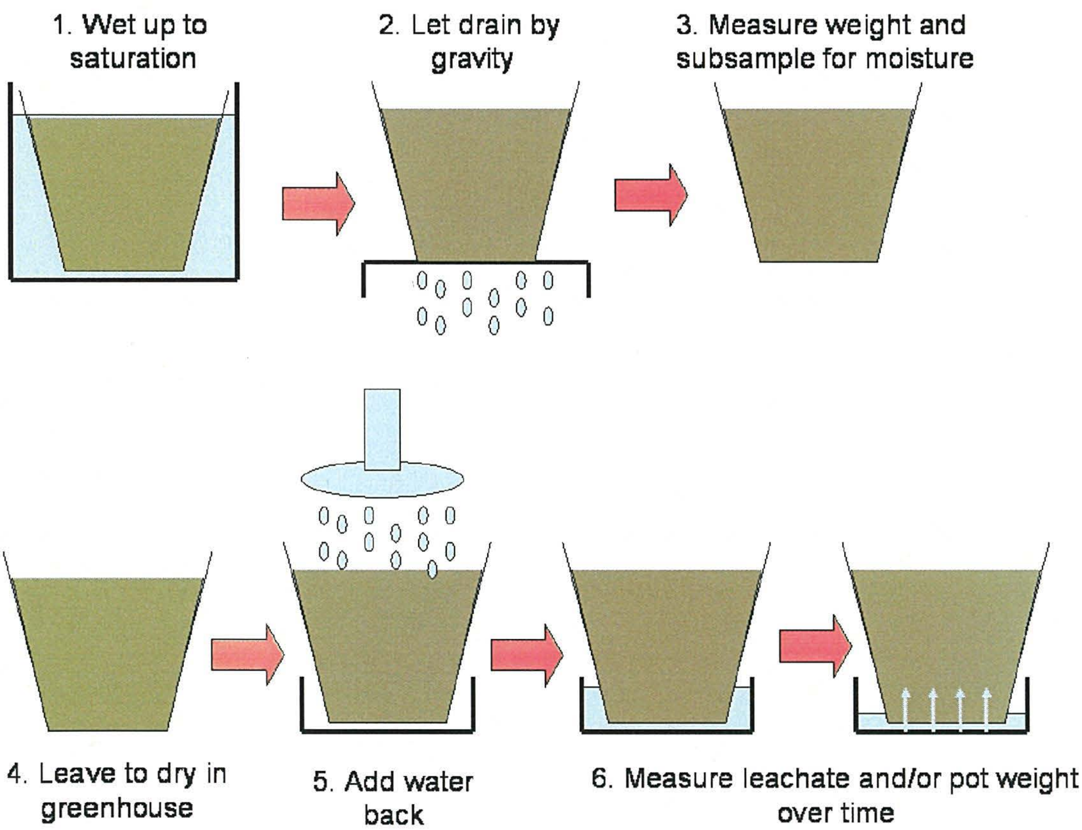


Figure 5.1 Schematic representation of the method used for obtaining the drying and wetting cycles for peat, green waste and vermicompost. Briefly, saturated samples were progressively dried in a controlled environment growth cabinet at 20°C at 70% RH and in the absence of light, the pots re-watered and the time for water re-absorption to occur recorded.

5.3.5 Water Repellency/Hydrophobicity

Samples were carefully removed from the compost surface and from the sub-surface (5 cm below surface). The reason for this was that typically hydrophobicity tends to decrease with distance from the surface (Doerr, 1998), although Doerr *et al.* (2000) showed that in some cases the reverse is seen. However, these findings relate to soils measured *in situ* and not composts.

Samples were taken from the sides of the pot to limit the destruction of the core, so as to limit creating preferential pathways that might influence the rewetting process. However, this incurred a limitation to the results owing to the spatial variation of hydrophobicity (Doerr *et al.*, 2000). The overall mass of pots with regard to the % of water lost did not account for the spatial variation in moisture throughout the whole pot. Drying is rarely uniform and some parts of the compost (central) were wetter than the compost sampled at the top and sides of the pot. The wettability of the compost as a whole, was assessed during the rewetting process.

The persistence of water repellency was measured using the 'water drop penetration time' (WDPT) method according to Doerr (1998) and Letey (1969) (Figure 5.2). WDPT was taken as the mean of 3 drops per sample. Observations of droplets were terminated after 5 hours, at which point the sample was considered to be persistently hydrophobic. Figure 5.2 demonstrates how if the drop does not penetrate immediately it indicates that the water surface tension is above that of the soil surface and so the soil – water contact angle (θ) is $\geq 90°$. Since water enters a porous surface if θ is $\leq 90°$ then this procedure measures the time taken for θ to alter from ≥ 90 ° to $\leq$ 90 °. In summary, the contact angle of water on a plane surface therefore provides an index of water repellency (Doerr, 1998) (See section 2.3.2.6.4 – 2.3.2.6.5 for further explanation).

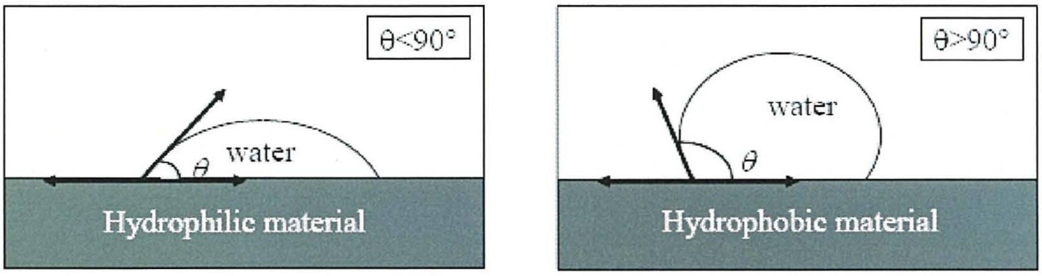


Figure 5.2 Contact angle of water drop on hydrophobic and hydrophilic materials, (Michel, 2010).

5.3.6 Compost Shrinkage

Shrinkage was calculated by measuring the percentage (%) of volume lost in terms of the pot space occupied by the compost. Three random measurements were taken horizontally from the sides of the pot, three vertically in a central position and three vertically from the outer edge of the compost core to the sides of the pot. The percentage reduction in volume was calculated using the formula for the volume of a cone (Eq. 1)

$$\text{Volume (cm}^3\text{)} = 1/3\ \pi\ r^2\ h \qquad (1)$$

5.3.7 Rewetting Leachate

Leachate emerging from the bottom of the pots during re-wetting was collected for the 100%, 40% and 5% drying treatments at the beginning of the rewetting process. Leachates were analysed for electrical conductivity and pH Smith & Doran (1996) and UV absorption at 400 nm and 254 nm (UVA) to assess the level of soluble humic substances (Weishaar *et al.*, 2003). Humic substances are known to influence organic matter breakdown and hydrophobicity in soils.

5.3.8 Water Retention/Release Curves

Water potential was calculated by using a dew-point potentiometer, WP4 (Decagon Devices, Pullman, WA). Briefly, 2 g of compost from the dried pots were sub-sampled below the surface. Samples were removed close to the sides of the pot so as not to create large fissures in the compost, as this would then affect the infiltration process during rewetting. To ascertain the compost water potential following rewetting, pots were wet up (as in section 5.3.2) for differing lengths of time to vary the volumetric water content in samples and then approximately 2 g of compost was sub sampled. All samples were sealed in the WP4 psychrometer cups and left to equilibrate at room temperature for 12 hours before analysing. Once 3 water pressure potential readings were gained within 10% of each other the samples were then dried in an 80°C oven to calculate the compost volumetric water content (θ).

5.3.9 Data Analysis

Data were subjected to factorial ANOVAs using SPSS v.16, (SPSS Inc., Chicago, IL) and Minitab (Minitab Inc., State College, PA). Statistical differences were calculated using Post Hoc analyses were Tukey's pairwise comparisons. Means and standard errors were calculated using Microsoft Excel 2003 (Microsoft Corp., Redmond, WA) and all graphical figures were produced in SigmaPlot v10.0 (Systat Software Inc., San Jose, CA) with the exception of the Van Grenuchten models which were done in Microsoft Excel 2007 (Microsoft Corp., Redmond, WA). The WDPT was measured using a macro-timer in Microsoft Excel 2003 (Microsoft Corp., Redmond, WA).

Rate of compost drying was described by a first order exponential decay model (Eq. 2):

$$D = a \times \exp(-b \times t) \tag{2}$$

Where D = is the moisture remaining in the compost, b is the exponential coefficient describing the rate of water leaving the compost and a is the exponential coefficient related to the moisture content associated with the exponential coefficient b at time (t). The half-life ($t½$) of a was then calculated, giving the theoretical time taken for half of the volume of water to be lost, and using the following equation (Eq. 3):

$$t½ = \ln(2)/b \tag{3}$$

Rewetting data was inverted (100 - % recapture) and curves were evaluated using a first order exponential rise to maximum (three parameter) model (Eq. 4):

$$R = \mathrm{y0} + a \times (1 - \exp(-b \times t)) \tag{4}$$

Where R was the moisture recaptured, b is the exponential coefficient describing the rate of water recaptured, a is the exponential coefficient related to the moisture content associated with the exponential coefficient b at a point in time (t) and y0 is the asymptote. The asymptotic curve was used to calculate the theoretical rise to maximum (of water retention) and this value was used to analyse any significant differences between treatments within each drying group.

The van Genuchten (1980) retention model was used to describe the water retention characteristics of the 3 unsaturated porous media, (Eq. 5):

$$\theta(\psi) = \theta r + (\theta r - \theta s)/[1 + (\psi/\alpha/)^{m}]^{n} \qquad (5)$$

Where θr is the residual volumetric water content ($cm^3\ cm^{-3}$), θs is the volumetric water content at saturation ($cm^3\ cm^{-3}$), α is the curve fitting parameter, often described as the inverse of the air entry value (1/ ψe) (cm^{-3}). Both *m* and *n* are fitting constants representing the steepness of the curve.

5.4 Results

5.4.1 Compost Properties

Peat was significantly different to green waste compost and vermicompost for all parameters measured, apart from the conductivity (EC). The EC for vermicompost was comparable to peat (Table 5.1). Green waste compost was significantly higher ($P < 0.001$, n = 12) than both vermicompost and peat for EC. The pH for peat was typically low, with vermicompost and green waste compost significantly different ($P < 0.001$, n = 12) – green waste compost being the highest pH (7.85). Bulk density followed a similar pattern – peat significantly lower than the rest ($P < 0.01$, n = 12), then vermicompost and finally green waste compost with a significantly higher bulk density than the rest ($P < 0.01$, n = 12). Peat was significantly higher than the other composts with 91% organic material ($P < 0.001$), with vermicompost significantly lower ($P < 0.01$, n = 12) than green waste compost - 39% and 44% respectively. The volumetric water content at container capacity (θ) for green waste and vermicompost was comparable but peat was significantly higher (P < 0.01, n = 60). Gravimetric water content at container capacity were significantly different for all three with the highest in peat ($P < 0.001$, n = 60), then vermicompost ($P < 0.001$, n = 60) and finally green waste compost ($P < 0.001$, n = 60).

For the 3 batches of vermicomposts (V1, V2 and V3) there was considerable variation in the parameters. Bulk density differed for all three ($P < 0.001$, n = 4), with the lowest being V2 (paper pulp amended) and the highest in the pure green waste vermicompost V1. Interestingly, the bulk density value for V1 did not differ markedly from that of the green waste compost (0.42 g cm^{-3}). V2 had a significantly higher gravimetric water content at container capacity ($P < 0.001$, n = 20) compared with V2 and V1. The paper pulp amended vermicompost (V2) also had a higher volumetric water content compared with the pure green waste vermicompost (V1) but V3 (sewage sludge amended vermicompost) was similar to both V1 and V2 ($P > 0.05$).

5.4.2 Compost Shrinkage

Drying induced shrinkage was presented for each individual batch of the three composts types (Table 5.2). This was undertaken to highlight any variability between the 3 different compost types during different stages of drying and between the three different vermicomposts feedstocks. The 3 green waste batches exhibited no significant differences between each other at any stage of drying ($P > 0.05$, n = 4), all 3 experiencing minimal shrinkage, even at 5% of container capacity (CC). The paper pulp amended vermicompost (V2) experienced the greatest shrinkage at every drying stage. V2 shrinkage was significantly higher than all of the composts ($P < 0.001$, n = 4, per drying stage) apart from at 20% CC where P1 was not significantly different from V2 ($P > 0.05$, n = 4). There were differences between the three horticultural peats, where P1 and P2 shrinkage was higher at 40% CC and 20% CC than the P3 equivalent ($P < 0.01$, n = 4). However, by the final stage (5% CC) all peats were very similar having had a reduction in substrate volume ranging from 23 – 25%. The green waste compost batches were all comparable with the pure green waste vermicompost V1, all having minimal shrinkage.

Table 5.1. **Properties of green waste-derived composts, horticultural peat composts and vermicompost. Results for vermicomposts were presented separately, owing to the difference in the feedstocks used - vermicompost 1 (V1) was green waste-derived, vermicompost 2 (V2) was green waste and paper pulp (70:30) and vermicompost 3 (V3) was green waste and sewage sludge (70:30). Values for composts are the mean (n = 12) ± SEM. Statistical differences between peat, green waste compost and vermicompost were calculated using a One-way ANOVA and also for the 3 vermicomposts to see the effect of feedstock. Values for vermicomposts are the mean (n = 4) ± SEM. Statistical differences (*P* < 0.01) between treatments within each compost type are denoted by superscript letters ($^{a, b}$ and ^{c}). CC indicates container capacity.**

	Peat	Green waste	Vermicompost	Vermicompost		
				V1	V2	V3
pH	3.7 ± 0.1^{a}	7.9 ± 0.1^{c}	6.8 ± 0.2^{b}	7.2 ± 0.1^{a}	7.2 ± 0.1^{a}	6.1 ± 0.1^{b}
Bulk density, g cm^{-3}	0.19 ± 0.00^{a}	0.42 ± 0.00^{c}	0.38 ± 0.00^{b}	0.42 ± 0.00^{a}	0.34 ± 0.00^{c}	0.37 ± 0.00^{b}
Conductivity, mS cm^{-1}	0.6 ± 0.0^{a}	4.6 ± 0.6^{b}	2.4 ± 0.8^{a}	3.2 ± 0.1^{a}	0.9 ± 0.1^{a}	5.6 ± 0.5^{b}
Organic matter, %	91 ± 0.7^{a}	44 ± 1.1^{c}	39 ± 1.9^{b}	36 ± 0.7^{a}	36 ± 0.8^{a}	47 ± 3.0^{b}
Gravimetric water content at CC, g g^{-1} compost	3.6 ± 0.1^{a}	1.6 ± 0.0^{c}	1.8 ± 0.0^{b}	1.5 ± 0.1^{a}	2.1 ± 0.0^{c}	1.8 ± 0.0^{ab}
Volumetric water content as percentage (%)	71 ± 1.1^{a}	66 ± 1.2^{b}	69 ± 1.2^{b}	65 ± 2.8^{a}	73 ± 1.2^{b}	67 ± 1.6^{ab}

Table 5.2 Shrinkage of compost incurred as a result of successive periods of drying – values refer to the percentage of container capacity remaining in composts with 5% being the least moisture remaining. P1 – 3 are the three horticultural peat composts, G1 – 3 are 3 green waste-derived composts and V1 – 3 are 3 vermicomposts using different feedstocks (V1 = green waste, V2 green waste & paper pulp (70:30 v/v), V3 = green waste & biosolids (70:30 v/v)). Differences were calculated between each compost type within each drying group using a one-way ANOVA and are denoted by $^{a\,b\,c}$ and ^{d}.

	Level of drying – as percentage of water content at container capacity			
	60 %	40 %	20%	5 %
Compost	Percentage of volume lost (%)	Percentage of volume lost (%)	Percentage of volume lost (%)	Percentage of volume lost (%)
P1	0.0 ± 0.0^{a}	10 ± 2.4^{bc}	24 ± 3.6^{bc}	25 ± 2.7^{b}
P 2	1.5 ± 0.4^{a}	13 ± 1.7^{c}	15 ± 3.1^{ab}	26 ± 2.7^{b}
P 3	0.0 ± 0.0^{a}	1.7 ± 0.9^{ab}	6.8 ± 2.1^{a}	23 ± 2.2^{b}
G 1	0.0 ± 0.0^{a}	3.1 ± 0.5^{ab}	6.8 ± 2.9^{a}	5.3 ± 1.5^{a}
G 2	0.0 ± 0.0^{a}	0.3 ± 0.2^{a}	6.3 ± 1.9^{a}	7.8 ± 0.1^{a}
G 3	0.0 ± 0.0^{a}	1.3 ± 0.5^{ab}	11 ± 1.7^{a}	11 ± 0.2^{a}
V1	0.0 ± 0.0^{a}	3.3 ± 0.3^{abc}	7.2 ± 2.2^{a}	6.7 ± 1.5^{a}
V2	14 ± 2.8^{b}	24 ± 3.6^{d}	32 ± 3.5^{c}	55 ± 2.5^{c}
V3	2.3 ± 2.1^{a}	8.7 ± 0.9^{abc}	14 ± 2.3^{ab}	13 ± 1.3^{a}

5.4.3 Water Repellency

The interaction of the independent variables – compost type, sample position in the compost profile (surface, sub-surface) and drying level was significant (P < 0.001) meaning that their main effects were not additive (see Table 5.3). Compost type had a highly significant effect on hydrophobicity (P < 0.001). Data for 80% CC dried composts was not presented as hydrophobicity was not persistent for any samples at this wetter end of the scale. There were no significant differences between the different vermicomposts (P > 0.05, n = 9) and so the data is not presented.

Peat became progressively more hydrophobic as the level of desiccation increased. Water repellency in the 5% CC peat treatment compared with the 60% CC peat treatment was very different for both surface and sub surface samples. Water took more than 20 times longer in surface samples and 800 times longer in sub surface samples to infiltrate the matrix in the 5% of CC samples (P < 0.001, n = 27) compared with 60% CC samples. The level of drying had the least effect in vermicompost – particularly at the surface where there were no differences between each drying group (P > 0.05, n = 27). Green waste compost WDPT followed a less consistent pattern but hydrophobicity tended to be more persistent in the drier samples (5 and 20% CC). However, the sub sample at 20% was the most hydrophobic.

Surface samples tended to be more hydrophobic than sub-surface samples in the wetter treatment (60% CC) in all composts. The difference between the hydrophobicity of surface and sub-surface samples tended to get less, as drying progressed. However, at no point were surface samples significantly more hydrophobic than sub surface ones for peat and vermicompost (P > 0.05, n = 108).

Overall, vermicompost demonstrated the least water repellency compared with peat and green waste-derived composts. By 20% CC, vermicompost was significantly less hydrophobic than both the peat and green waste-derived composts. Sub-surface green waste compost was significantly more hydrophobic than peat sampled at sub-surface ($P < 0.01$, n = 27) at 20% CC. By 5% CC the largest differences between the composts were apparent. Vermicompost was mildly hydrophobic compared with green waste compost ($P < 0.05$, n = 54) and peat was highly hydrophobic compared with both green waste compost ($P < 0.01$, n = 54) and particularly vermicompost ($P < 0.001$, n = 54).

Table 5.3 Water Drop Penetration Times (WDPT) as a measure of the persistence of water repellency for peat, green waste and vermicompost that have been dried down to 60, 40, 20 and 5% of water content at container capacity. Samples were taken from the surface of the pots (surface) and 5 cm below the surface (sub surface) (n = 12 for each compost at both sampling points). Statistical differences were calculated using a 3-way ANOVA with pairwise comparison in Minitab (v.15) with the main factors being the level of drying, profile (surface, sub surface) and compost type. Values are time taken in seconds (ss) presented as the mean ± SEM (n = 27) and significant differences between treatments denoted by $^{a\,b\,c\,d\,e}$ and ^{f}.

Compost	Drying Level – Percentage of container capacity							
	60 %		40 %		20 %		5 %	
	WDPT (ss) Surface	WDPT (ss) Sub surface	WDPT (ss) Surface	WDPT(ss) Sub surface	WDPT (ss) Surface	WDPT(ss) Sub surface	WDPT (ss) Surface	WDPT(ss) Sub surface
Peat	191 ± 78^{ab}	3.8 ± 0.3^{a}	55 ± 23^{ab}	14 ± 4^{ab}	2232 ± 286^{e}	1878 ± 221^{e}	4313 ± 348^{f}	4220 ± 313^{f}
Green waste	209 ± 63^{ab}	3.4 ± 0.1^{a}	1373±162^{cde}	401 ± 117^{ab}	2020 ± 300^{e}	4475 ± 372^{f}	1682 ± 132^{ed}	1862 ± 176^{e}
Vermicompost	50 ± 19^{ab}	3.4 ± 0.1^{a}	581 ± 65^{bc}	218 ± 18^{ab}	460 ± 44^{ab}	880 ± 107^{bcd}	506 ± 46^{b}	603 ± 53^{bc}

5.4.4 Compost Leachates

On the whole for vermicompost and green waste-derived compost, the pH and EC values for their leachate, steadily increased with the level of drying (Table 5.4). UV absorbance at 254 and 400 nm steadily rose for peat. In the green waste compost treatment, the UV absorbance at 254 nm was significantly higher at 5% CC compared with 40% and 100% CC. For the vermicompost treatment, leachate collected from rewetting the 5% CC pots had a significantly higher absorbance value 400 nm than 100% and 40% CC leachates.

5.4.5 Drying curves

At each drying stage the green waste-derived composts lost water faster than both the peat and vermicompost treatments. Initially, water loss in vermicompost was slowest presenting a significantly different rate than the green waste composts ($P <$ 0.001, n = 12) and peats ($P < 0.05$, n = 12). As the composts became drier, water loss in the peat was slowest. Consequently, the time taken to reach 20% and 5% CC for peat was significantly greater in comparison with both the vermicompost and green waste compost treatments (20% = $P < 0.01$, n = 12) (5% = $P < 0.001$, n = 12).

Table 5.4 Analysis of compost leachate collected during rewetting, following different degrees of drying - no drying (100%), drying to 40% and 5% of container capacity. Leachates were then analysed for pH, EC and UV absorbency at 254 nm and 400 nm in the visible range to assess humic substances. Statistical differences were calculated using a 1-way ANOVA in SPSS (v.16) for each compost type with the main factor being the level of drying (100%, 40% or 5%). Values are the mean of samples (n = 4) ± SEM. Significant differences between treatments at the *P* < 0.01 level are denoted by ^{a} and ^{b}.

	Peat			Green waste compost			Vermicompost		
	100%	40%	5%	100%	40%	5%	100%	40%	
pH	4.16 ± 0.09	4.38 ± 0.18	4.20 ± 0.10	7.75 ± 0.12^{a}	8.48 ± 0.08^{b}	8.33 ± 0.14^{b}	6.58 ± 0.11^{a}	6.46 ± 0.10^{a}	7
Conductivity, mS cm^{-1}	0.14 ± 0.01^{a}	0.23 ± 0.03^{b}	0.14 ± 0.02^{a}	2.00 ± 0.20^{a}	2.00 ± 0.21^{a}	4.12 ± 0.56^{a}	2.40 ± 0.40	2.02 ± 0.19	3
Absorbency 254 nm	1.09 ± 0.21^{a}	1.34 ± 0.13^{a}	2.31 ± 0.16^{b}	11.09 ± 0.49^{a}	9.37 ± 1.20^{a}	5.32 ± 0.34^{b}	1.26 ± 0.19	1.34 ± 0.07	0
Absorbency 400 nm	0.14 ± 0.03^{a}	0.14 ± 0.01^{a}	0.31 ± 0.02^{b}	2.04 ± 0.14	1.34 ± 0.07	1.51 ± 0.12	0.11 ± 0.02^{a}	0.11 ± 0.01^{a}	0

Figure 5.3a

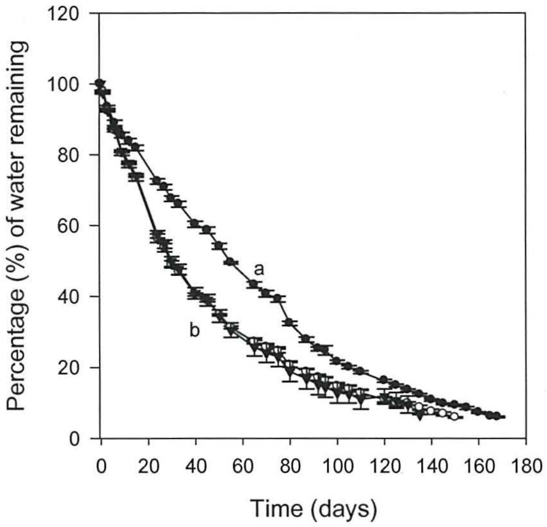


Figure 5.3b

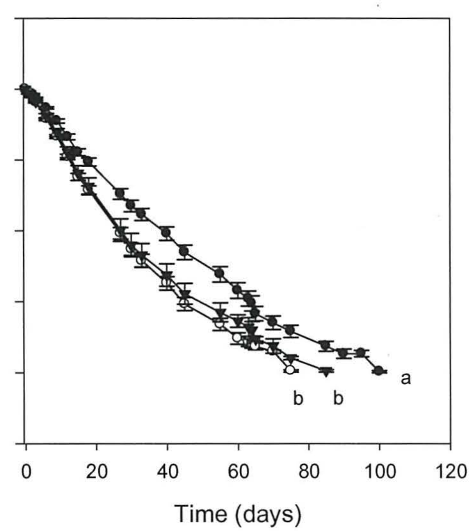


Figure 5.3c

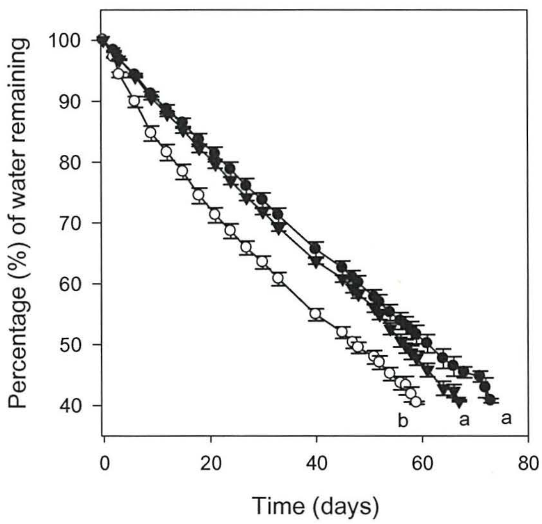


Figure 5.3d

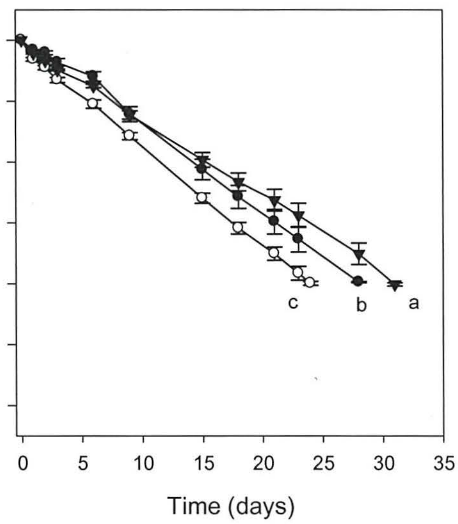
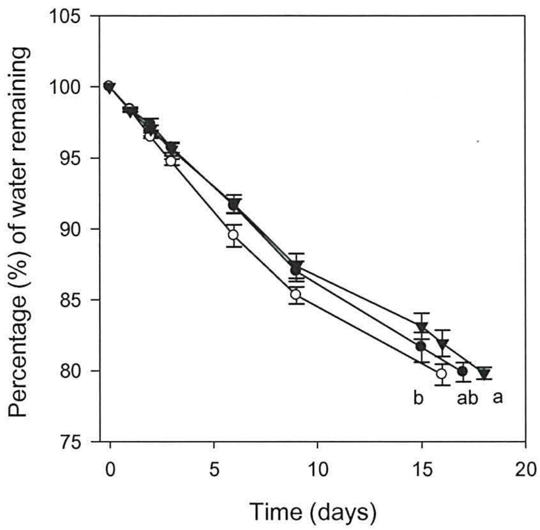


Figure 5.3e



Figures 5.3a, 5.3b 5.3c, 5.3d and 5.3e are the drying curves for peat, vermicompost and green waste having been dried to 5, 20, 40, 60 and 80 percent (%) of container capacity (CC) respectively. Lines with black circles ● represent peat, clear circles ○ represent green waste compost and black triangles ▼ represent vermicompost. Statistical differences between compost types were calculated using the exponential decay rate coefficient within each drying group, using a One-Way ANOVA (SPSS v.16) with compost type as the main factor. Values represent the mean of composts (n = 12) ± SEM.

Table 5.5a & 5.5b Half-time (t½) of the time taken for composts to dry to 50% of container capacity. Statistical differences were calculated using a 1-way ANOVA in SPSS (v.16) with the main factor being compost type. Values represent the mean for composts ± SEM (n = 12). Significant differences at the *P* < 0.01 level are denoted by [a] and [b].

Table 5.5a

Compost	Half life (days)
Peat	50 ± 2.3^{a}
Green waste	40 ± 2.1^{b}
Vermicompost	46 ± 3.1^{ab}

Table 5.5b

Compost	Half life (days)
V1	42 ± 2.8
V2	48 ± 3.1
V3	48 ± 1.7

The *t*½for composts drying to 50% of CC (Table 5.5a), demonstrates that peat-based composts were significantly slower at losing moisture compared with the green waste-derived composts, whereas the vermicomposts were comparable to peat ($P > 0.05$, n = 12). The effect of feedstock in the three vermicompost treatments was not significant in terms of drying rate ($P > 0.05$). However, the *t*½for green waste vermicompost (V1) was very similar to the green waste compost, whereas the time taken for half of the water held to exit the substrate for V2 (+paper pulp) and V3 (+ sewage sludge) were more comparable with peat.

5.4.6 Rewetting Curves

At 5% of CC (Figure 5.4a), when composts were severely desiccated, the vermicomposts recaptured a significantly greater amount of water during rewetting than both the peat ($P < 0.05$, n = 12) and green waste composts ($P < 0.001$, n = 12). Green waste compost was the most compromised in terms of rewetting, absorbing only 34% of water held at CC - being significantly lower than peat as well as vermicompost ($P < 0.01$, n = 12).

In the 20% CC treatment (Figure 5.4b), vermicompost rewet to a level comparable with that previously held at 5% (approx 60% CC at rewetting equilibrium), whereas the green waste composts recaptured around 62% of water added having doubled its volume of water uptake when at 5% CC. At 20% CC desiccation, peat recaptured the greatest amount of water lost (78%) and this was a significantly greater percentage than both green waste compost and vermicompost ($P < 0.01$, n = 12).

At 40% CC (Figure 5.4c) the vermicompost rewetting rate remained fairly constant, recapturing 65% of water. However, both the peat and green waste compost recaptured a comparable amount of water (81% and 79% respectively) and were significantly better than the vermicompost ($P < 0.001$, n = 12).

There were no significant differences between composts at 60% CC (Figure 5.4d). The results from the composts dried to 80% CC (Figure 5.3e), were similar to 40% CC, where peat and green waste compost were recapturing a similar amount of water (92% and 91% respectively). Vermicompost only recaptured 73% of water during rewetting and this was significantly less than the peat and green waste compost treatments ($P < 0.001$, n = 12).

The influence of vermicompost feedstock was most pronounced when monitoring the rewetting behaviour (Table 5.6). Essentially, V2 (the vermicompost amended with paper pulp), was the most compromised by the effects of drying. This was particularly pronounced at 5% CC, where V2 recaptured significantly less water during rewetting than both V1 and V3 ($P < 0.001$, n = 4). Results were similar for the 20% CC drying stage. In the 40% CC treatment, V3 (vermicompost and sewage sludge) recaptured a significantly greater percentage of the water lost than both V1 ($P < 0.05$, n = 4) and V2 ($P < 0.001$, n = 4). V1 recaptured almost 20% more water than V2 ($P < 0.001$, n = 4). For the wetter samples (60% and 80% CC) V1 recaptured the highest amount of water lost, although V3 was comparable. V2 still recaptured less water than both V1 ($P < 0.001$, n = 4) and V3 ($P < 0.05$, n = 4) in the wetter composts (60% and 80% CC).

Figure 5.4a

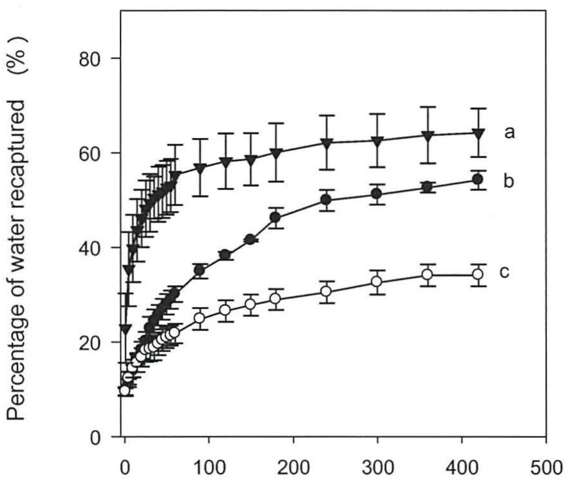


Figure 5.4b

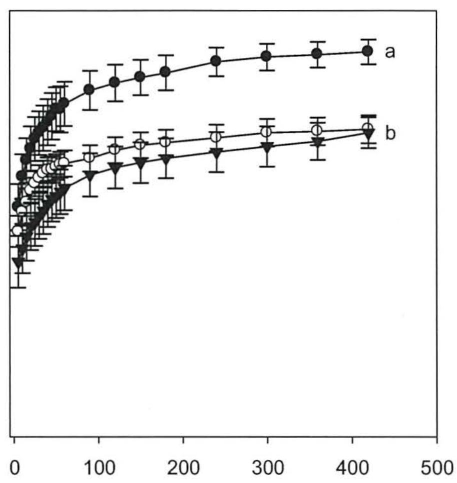


Figure 5.4c

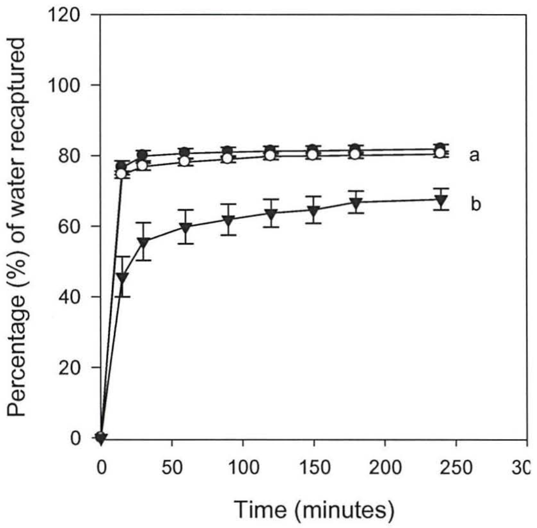
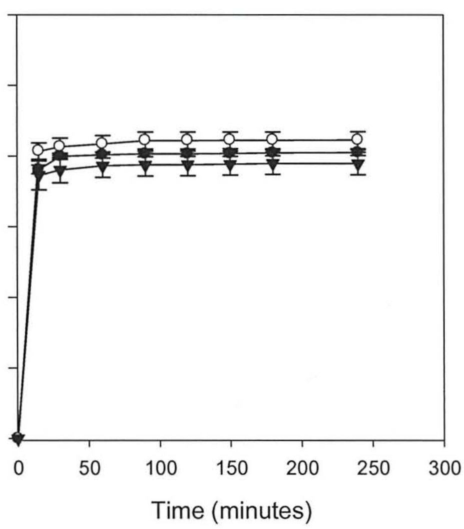
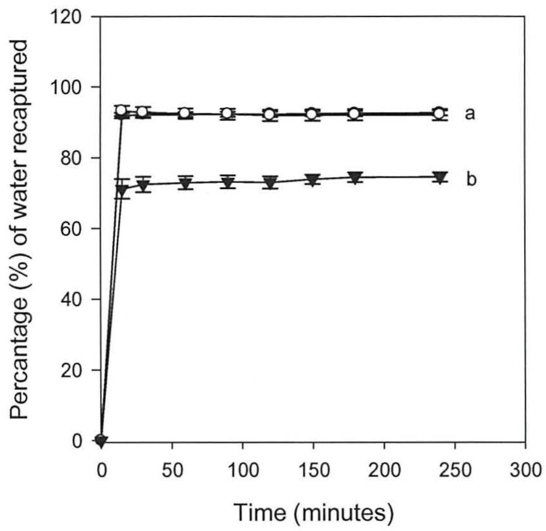


Figure 5.4d





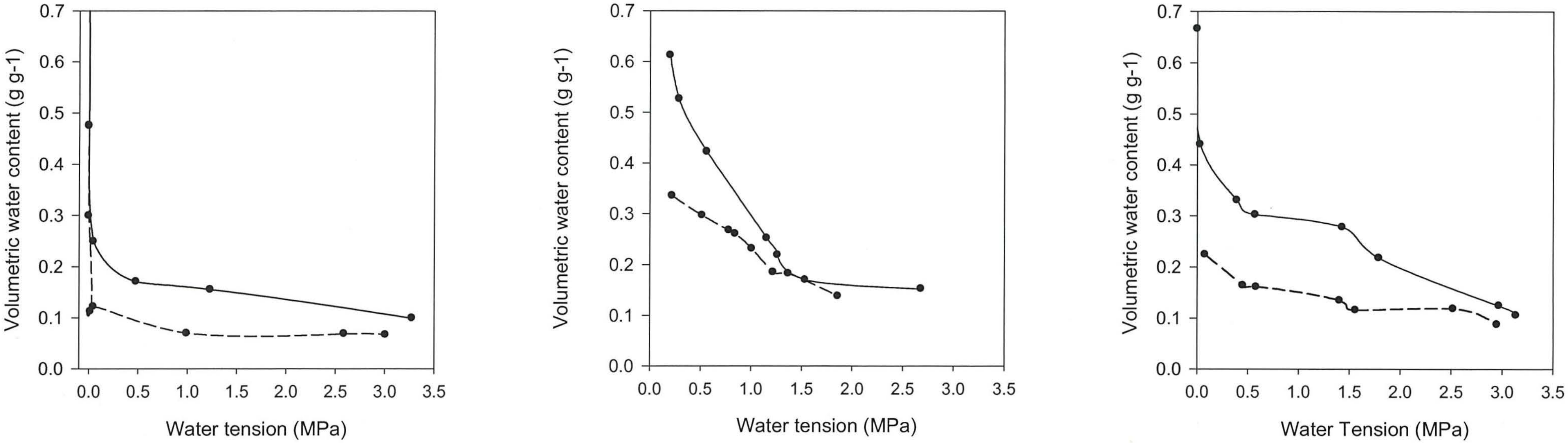
Figures 5.4a, 5.4b 5.4c, 5.4d and 5.4e are the percentage of water recaptured for peat, vermicompost and green waste having been dried to 5%, 20 %, 40%, 60% and 80% of container capacity (CC) respectively. Values are the mean of rewetting rate presented as the percentage of water recaptured at a point in time ± SEM (n = 12). Lines with transparent circles ○ represent green waste compost, black circles ● represent peat and black triangles ▼ represent vermicompost. Standard errors for VC were high owing to VC2 all of which were severely compromised in their ability to rewet following extensive drying (Figure 5.5a). Statistical differences between compost types were calculated using the asymptote for each replicate, within each drying group using a One-Way ANOVA (SPSS v.16) with compost type as the main factor

Table 5.6 Theoretical rise to maximum percentage of water recaptured for 3 vermicomposts following drying to different levels of container capacity (CC). V1 = green waste vermicompost, V2 = green waste and paper pulp vermicompost (70: 30) and V3 = green waste and sewage sludge vermicompost (70: 30). Values represent the mean of asymptotes at each drying level ± SEM (n = 4) Statistical differences are denoted by [a], [b] and [c] and were calculated using a one-way ANOVA within each drying group, with vermicompost as the main factor

Vermicompost	Percentage of water recaptured (%)				
	80% CC	60% CC	40% CC	20% CC	5% CC
V1	77 ± 1.9^{a}	87 ± 2.3^{a}	69 ± 2.1^{c}	69 ± 0.9^{a}	69 ± 1.6^{a}
V2	67 ± 2.7^{b}	66 ± 2.3^{b}	50 ± 0.8^{b}	43 ± 1.0^{b}	48 ± 0.9^{c}
V3	75 ± 1.4^{a}	78 ± 3.3^{a}	75 ± 0.8^{a}	68 ± 1.1^{a}	63 ± 1.1^{b}

5.4.7 Water Release Curves (WRC)

Figures 5.5a&b presents the initial data for the water availability in the three compost types for desorption and adsorption. Fitting the data to the van Genuchten model had varied success in the three compost types. This model assumes a rigid matrix and so with the level of shrinkage experienced in the peat and V2 and to a lesser degree - V3, fitting the data was only marginally successful. Figure 5.5a is the WRC for the three peat composts. Having fitted the date to the van Genuchten model an envelope between 0.47 and 0.8 volumetric water contents exists where the first negative pressure reading could have fallen. This means that drawing any firm conclusions about air entry point and the hysteresis between desorption and sorption in the wetter end of the scale is not possible. Figure 5.5b is the WRC for green waste compost, which experienced minimal shrinkage. The hysteresis between desorption and sorption is small. However, air entry point cannot be reliably calculated based on this data owing to the lack of pressure readings in the wetter end of the scale. The van Genuchten model was most successfully fitted to the green waste compost water potential data. This was largely due to the lack of shrinkage in this media. Again there is a lack of data at the wetter end of the spectrum meaning that assessing the parameters for the point of air entry was not done, as more data was required for conclusions to be drawn from the graphs.



Figures 5.5a, 5.5b and 5.5c are the water release curves for each peat, green waste and vermicompost respectively. The solid line — represents desorption where the water tension versus volumetric water content are for composts dried to various water contents and the dashed line - - - - represents adsorption where the water tension versus volumetric water content are for composts wet-up to various water contents. The area between the lines represents the lag phase or hysteresis between wetting and drying. A model representing exponential decay was fitted to each replicate (n= 12 per compost). Values represent the individual reading for random samples at different volumetric water contents.

5.5 Discussion

5.5.1 Drying composts

This study confirms that peat, green waste compost and vermicompost are affected differently by drying and rewetting events. For peat and green waste derived composts we had 4 replicates from 3 distinct batches and there was very little variation between the batches for each compost type. The vermicomposts varied in the feedstock and differences in the drying and re-wettability were seen. Drying rate was initially slowest in all of the three vermicomposts but as the composts became drier, the peat composts had the slowest rate of water loss. This could possibly be explained via two mechanisms: i) as the peat became drier it formed a seal at the surface, with increased strength (Whalley *et al.*, 2001) and decreased porosity (Gerbhardt *et al.*, 2010) slowing water loss via evaporation as the percentage area of surface connected pores decreased, and/or ii) rate of water loss from the larger and active pores was high but as the water stored in smaller non-connected or inactive pores was evaporating, the rate slowed significantly. This is most likely owing to the lack of connectivity between the smaller pores or increased tortuosity in the smaller pore structure (Rezanezhad *et al.* (2009). Peat also experienced increased shrinkage which would have entrapped water in the smaller pore space. Green waste compost is typically characterised by a large pore structure (Papafotiou *et al.* 2005), owing to large particle size and in this study all three green waste composts lost water at the fastest rate at all stages of drying down from CC. The ability for compost to withstand drying is an important property of growing media for the ease of management and environmental (IPCC, 2007) and economic considerations.

The three vermicomposts amended with different feedstocks did vary in the rate that water that was lost but not significantly. Both V2 and V3, which had paper pulp and sewage sludge amendments respectively, retained water for longer compared with the pure green waste vermicompost (V1). The feedstock therefore exerted an influence on pore formation and structure. V2 had the highest water content at container capacity and the volumetric water content was higher for V2 than the peat (73% c.f. 71%). Based upon this water parameter alone V2 looked like a favourable waste-derived alternative to peat.

5.5.2 Compost Shrinkage

The nature, size and stability of the pore space in compost, as with soil, depend upon the texture (proportions of various sized particles) but also by the level of aggregation (structure). The shrinkage incurred by drying was minimal in the green waste compost possibly owing to the lack of fine micro pores that tend to be most affected by drying (Michel, 2010). Green waste compared with peat has a much higher mineral content and this has been shown to stabilise structure in soils (Gebhardt *et al.*, 2010). Stabilising structure refers to the ability of the matrix to resist shrinkage and swelling. Highly organic materials have a dominance of smaller pore spaces meaning they are characteristically more prone to shrinkage, accounting for the comparatively higher shrinkage incurred in all three batches of the horticultural peat (Michel *et al.*, 2002). The variation in initial shrinkage in the three peat batches was indicative of slightly different aged peats, as less decomposed peat is less likely to shrink or have reduced wettability as a result of quasi-irreversible changes to the pore structure (Michel *et al.*, 2002). All three peats were produced from sphagnum peatland and the effect of desiccation could easily have incurred quasi-irreversible damage to the unique water transport and storage occurring within the remnants of the sphagnum plant debris (Caron *et al.*, 2005). Interestingly once the composts were rewet it was observed that P3, which had significantly more resilience to shrinking at the wetter end of the scale, returned to its original volume whereas swelling in the P1 and P2 substrates never quite occupied the same volume of space again. Valat *et al.* (1991) report that the elasticity of sphagnum peats compared with more woody herbaceous peat, coupled with the lack of decomposition, typically ensures that the effects of shrinkage are reversible and this study supports this finding.

Figure 5.8 Shrinkage in drying vermicompost (V2) made from green waste and paper pulp

The shrinkage in V2 was the most extreme of all the media (Figure 5.8), with over 50% reduction in substrate volume at 5% CC. The finer more silt like particles in this vermicompost and the kaolinite clay content originating from the paper production process (Chantigny *et al.*, 1999) are likely explanations for this lack of physical stability (resistance to shrink and swell). The shrinkage in the most desiccated pots for V2, exceeded 50% and was irreversible beyond the 20% drying stage. This means the original volume of the substrate was not regained following rewetting and in most cases did not alter at all. This has serious implications for using a vermicompost amended with paper pulp as a containerised media. All vermicomposts were produced in controlled conditions at the same time, receiving the same maturation phase and storage conditions. The differences therefore, can be assumed to be as a result of feedstock amendments.

5.5.3 Water Repellency

Overall the most persistent hydrophobicity was seen in the horticultural peat. Doerr *et al.* (2000) highlight that research has shown that a slow natural decomposition regime, as in peat bogs, tends to have an additive effect in terms of hydrophobicity compared with substrates that undergo faster rates of decay. The decomposition regime is clearly an important determinant of the level of water repellency for the composts used in this study. Valat *et al.* (1991) found that less decomposed peat was significantly more hydrophilic than more decomposed peat. Added to this, Valat *et al.* (1991) and Caron *et al.* (2005) state that peat containing sphagnum moss tends to be more wettable than peat sourced from woody herbaceous peat lands. Sphagnum peat tends to be hydrophilic and elastic in character meaning it tends to recover its loss of volume once it has been rewet. This study supported this claim. However, the results from this study show that persistent hydrophobic spots in the peat composts were more severe than either green waste or vermicomposts demonstrating a clear hydrophobic nature. The key factor influencing the severity of WR in peat was moisture content. For composts produced from thermophilic aerobic processes the reverse is true – whereby increased humification (increased composting time) tends to reduce the presence WR compounds (Valat *et al.*, 1991).

Vermicompost is produced from an entirely different decomposition method. Interestingly, regardless of feedstock, the levels of WR in the vermicomposts was significantly less than in peat and green waste compost – particularly at the drier end of the scale, meaning the stability of WR was not as dependent on actual water content for vermicompost. This is an attractive property in growing media – whereby allowing the vermicompost to dry out – will not induce severe WR, meaning rewetting is not as difficult. When the green waste feedstock was vermicomposted, WR was significantly less than in the standard green waste compost equivalent and this effect was magnified as the actual water content decreased. Indeed the vermicomposts' WDPT at each drying level (80% - 5%) did not differ significantly. This finding is unusual but not unique (Doerr *et al.*, 2000). Clearly the process of decomposition using worms resulted in the reduction of hydrophobic compounds. Doerr *et al.*, (2000) state that the aliphatic and aromatic C compounds are generally found in hydrophobic coatings. Vincelas-akpa & Loquet (1997) found that standard green waste compost had higher levels of aliphatic and aromatic C compared with vermicompost. In vermicomposting the mutual relationship between the compost microflora, ingested microbes and the intestinal mucilages means that the end product has different carbon fractions to the compost produced via the thermophic aerobic composting process.

This study did not look at the transformations of lignincellulosic compounds in the various media but further work would benefit from focusing on this, for a range of different wastes. According to the findings of Doerr *et al.* (2002) where high ambient relative humidity (ARH) was shown to induce more severe WR than in low

ARH conditions, the growth room conditions were controlled at ARH of 70%. This was because greenhouses typically have high ARH but this study acknowledges that WR may be less in environments of lower relative humidity, such as households. In environments with an ARH < 70% the composts may not manifest the same level of WR.

5.5.4 Compost Rewetting

V2 (green waste and paper pulp vermicompost) saw a large re-organisation of the solid phase due to shrinkage (Naasz *et al.*, 2005), which negatively affected the wettability overall. However, the water repellency results did not support this finding, where the reduced infiltration during rewetting was not reflected by increased WDPT. WDPT for V1 and V3, although not significantly different to V2, tended to be less than in V2, indicating more persistent hydrophobicity from surface and sub-surface samples. The independent samples removed from V2 were not very hydrophobic but the lack of rewetting could be explained by the overall loss of pore structure and connectivity resulting in a lack of hydraulic conductivity (Omuto, 2009). After rewetting the V2 that was dried to 5% CC and 20% CC, the original volume was never recovered; indeed there was very little regain of the compost volume that was lost during drying. This indicates a lack of elasticity (Michel, 2010), where quasi-irreversible changes to the physical properties of this vermicompost occurred with increased desiccation. The fact that V2 did not rewet, (largely via capillary rise), means that grouping it with the other two vermicomposts resulted in high SEMs (figures 5.4a and 5.4b).

At 80 – 20% of CC peat recaptured the closest amount of water originally to that held at CC but once it had been dried to 5% of container capacity then it was significantly poorer at rewetting than the vermicomposts. At 5% CC the peat and greenwaste compost were both very hydrophobic and unable to rewet to anywhere near the original amount of water held at CC. This could be due to a loss of pore connectivity between the macro air-filled pores and the micro pore, impeding unsaturated hydraulic conductivity (Horgan & Ball, 2005). Pires *et al.* (2008) recorded an increase in the total number of larger pores (> 500 μm) owing to wetting and drying cycles. However, this study was looking at soil and not compost. Pores of this size tend to allow root penetration and water transport through the matrix. Whereas a decrease in pores of a medium size range (50 – 500 μm) is common following cycles of wetting and drying (Pires *et al.*, 2008). These pores are typically classed as transmission pores whereby water is transported through the matrix. So a loss in transmission pores could be the underlying reason for the lack of rewetting in the very dry green waste compost (5% CC) and peat – along with the increased hydrophobicity reducing even wetting via capillary rise.

5.5.5 Water release curves (WRC)

The main forces responsible for holding water in the soil are adhesion and capillarity. Adhesion is the intermolecular forces between the soil particle surfaces and water while capillarity describes how water is held in pores spaces between particles (Omuto, 2009). The WRC is based upon the combined effect of these forces and soil moisture. When soils/composts shrink during drying, the rewetting process often reveals the hysteresis phenomena (Nassz *et al.*, 2001) and reduced wettability (Michel *et al.*, 2002). In this context hysteresis relates to the difference in volumetric water content at a range of suctions depending on whether the substrate is drying or wetting. On the whole soils/compost hold less water at the same suction during adsorption (Horgan & Ball, 2005). All of the composts in this study demonstrated a level of hysteresis between desorption and adsorption cycles. The green waste compost experienced minimal shrinkage compared with peat and V2 and Figure 5.7 demonstrates very little hysteresis between desorption and adsorption for green waste compost. However, Horgan and Ball (2005) state that peat-free media tend to experience greater hysteresis between desorption and adsorption compared with peat-based media.

Ultimately, owing to the lack of measures for matric potential at the wetter end of the spectrum further analysis of the WRCs was curtailed. The WP4 is well known for not analysing suction of samples at the wetter end of the scale (> -1 kPa) as the relative humidity is too close to saturation (Curtis & Classen, 2008). Equally alternative methods commonly employed to obtain the WRC are subject to similar limitations. The pressure-plate method frequently results in an under-estimation of the plant available water in many soils and the issue of shrinkage in highly organic substrates adds to the disequilibrium that causes this lack of reliability in the data (Curtis & Claasen, 2008). Standard mathematical functions in hydraulic models (van Genuchten, 1980) could not be used to fully describe the substrates properties during wetting and drying episodes. This is also partly due to the fact that the model selected depicts the inverse relationship between soil suction and moisture based upon mineral soils. But it may also be due to important physical parameters that are not considered such as texture and the structure of pores (Omuto, 2009). Future studies should employ a range of methods for acquiring the WRC for composts and new models are being proposed to find parameters such as the air-entry point (Omuto, 2009) by considering factors such as texture and structural components of the pores.

5.5.6 Limitations due to preparation of materials and methods

The variation caused by compaction when first filling the pots would have an uneven effect on porosity and permeability (Michel *et al.*, 2002). Despite attempts to harmonise the packing densities this argument could explain some of the variability in rewetting rates within the same compost types. Media was packed using the same method for assessing the bulk density of samples (see Methods 5.3.3), so as to homogenise the compaction of material. However owing to the variability in compost particle size the effect of the compression would be greater in some compost compared with others.

By assessing the physical stability following increased desiccation and infiltration this study has not distinguished amongst the possible mechanisms for the effects seen in each compost and further work isolating potential mechanisms is required to illuminate them. In the vermicompost it is clear that feedstock exerts an important influence on physical stability in drying episodes. Characterising the wetting properties of media will generally be limited by the fact that wettability varies spatially and temporally (Hallet and Young, 2004).

Once plants are introduced to the system, owing to the complex relationship between the structural properties of the soil and the biota (Young & Crawford, 2004) the response of the various composts to wetting and drying effects could change from those witnessed in this trial. Importantly this study did not include plants in the trial and there is growing evidence to suggest that the impact of plant roots on soil stability is of more importance than any other factors, with fungi and other biota enhancing this effect (Hallett *et al.*, 2009). Roots provide stability through the effects of hydrophobicity and soil binding (Hallet *et al.*, 2009). So there is every chance that the stability of V2 would increase with the incorporation of plants in the growing media.

Cycles of wetting have been shown to further increase the effect of binding agents (Doerr *et al.*, 2000) increasing the proportion of water-stable aggregates. Plant roots, saprophytic organisms and earthworms have all been shown to exert an important influence on soil structure but their relative importance remains elusive (Hallet *et al.*, 2009). But again comparisons are limited by the fact that the majority of studies on this topic are for soil not compost.

Lastly with the WRC it was clear that the equations used to ascertain important water retention parameters are not suitable for highly organic materials. Ideally we would have monitored hydraulic conductivity in saturated and unsaturated conditions to provide a complete characterisation of the hydraulic properties of the media. The WP4 and pressure plate method are subject to disequilibrium with coarse grained media and substrates that are prone to shrinkage and swelling. The WP4 in particular does not accurately calculate potential head in wetter samples.

5.6 Conclusions

Research has shown that blending peat with waste-derived compost can result in a growing media with acceptable properties although a reduction in porosity and water-holding capacity (Clemmensen, 2004) along with an increase in bulk density (Papafotiou *et al.*, 2005) is usual. However, as peat is being phased out as a growing media (Wallace *et al.*, 2005) finding alternatives that do not require any peat amendment, is increasingly important. This study has shown that vermicompost tends to result in a growing media that does not get as hydrophobic as peat and standard green waste compost when subjected to increasing desiccation. However, the feedstock used in the initial feedstock can exert a significant influence on the physical stability of vermicomposts. Paper pulp amended green waste results in a vermicompost that had quasi-irreversible changes to the physical structure following drying. In order to determine an index of the available moisture in soil and to classify the water retention and drainage properties of organic media, new models and methodologies are required.

CHAPTER 6: USING NON-IONIC SURFACTANTS TO ALLEVIATE HYDROPHOBICITY IN PEAT AND GREEN WASTE COMPOST

6.1 Abstract

Ensuring the wettability of media is an important consideration when developing peat replacements for horticulture. This study aimed to evaluate the effect of two non-ionic surfactants – Ultraflo (UF) and Fifty90 (FN) on the wettability of horticultural peat and green waste-derived compost. The persistence of potential hydrophobicity was assessed using the water drop penetration time (WDPT) test. Surfactants were assessed for any potentially deleterious effects on the biota, by monitoring compost respiration and via a growth trial using wheat (*Triticum aestivum* L.). We determined the substrate leachate properties and volume following four successive wettings. Results showed that wettability was significantly improved by surfactant addition but the results varied for both composts. Of the two surfactants, FN was a more effective wetting agent than UF at lower application rates (0.2 ml l^{-1}). Electrical conductivity and pH of leachates, along with the emission of humic substances were mainly unaffected by the addition of surfactants. Water retention, taken as the volume of water not leached, was greatest for the highest application rate of FN. There were no significant differences in the above-ground biomass between plants grown in control (no surfactant) compost and those in surfactant-amended composts. However, wheat root dry weight was higher in the green waste control,

than in those treated with surfactants. Respiration in peat was significantly lower in samples treated with UF compared with the control and FN. Preliminary results indicate that non-ionic surfactants successfully alleviate hydrophobicity in the green waste and peat based composts, with FN being more effective than UF. Both surfactants showed very little or no deleterious effects on plant growth and microbial activity. However, further work is required to assess potential effects of leaching surfactants into the wider environment and the potential health risk associated with surfactant treated crops grown for human consumption.

6.2 Introduction

Substrate water repellency is common in soilless media and is often exacerbated by increased cycles of wetting and drying during the growing season (Urrestarazu *et al.*, 2008). Maximising efficiency in irrigation and the use of agrochemicals is inextricably linked to the wettability of growing media and this is of particular importance in containerised media owing to the small amount of substrate used (Naasz *et al.*, 2005). The wettability of growing media can be significantly improved by the application of surfactants. Applying surfactants to soilless media alleviates hydrophobicity by ensuring even wetting and reducing the overall infiltration rate (Olszewski *et al.*, 2008). However, there is a paucity of literature regarding the types of wetting agents used in horticulture (Olszewski *et al.*, 2008) and the effects of wetting agents in waste-derived media (Urrestarazu, 2008). Conversely, there is a rise in research studying the effects of non-ionic surfactants on soil wettability in turf science (Doerr & Ritsema, 2005). As hydrophobicity is common in coarse grained soils (Doerr *et al.*, 2000) – golf courses have become particularly susceptible to water repellency and concurrent loss of turf quality due to dry spots.

Initially, water retention properties were improved by amendment with organic matter. However, this had the unwanted side effect of exacerbating hydrophobicity, with an increase in incidence and severity (Doerr & Ritsema, 2005) due to the coating of the soil particles with hydrophobic organic compounds (Doerr *et al.*, 2000). Consequently, in recent years a range of non-ionic surfactants have been developed to improve soil wettability and turf quality in golf greens (Aamlid *et al.*, 2009).

Surfactants are surface-active agents that occur naturally or can be artificially synthesised and are commonly used in a wide range of applications such as detergents, cosmetics and as an adjuvant in pesticides (Hepworth, 2006). Surfactants have a polar, water soluble head group and a non-polar hydrocarbon tail group which is insoluble in water (Ivankovic *et al.*, 2010). Essentially at a certain concentration surfactant molecules aggregate into micelles, reducing the free energy in the system and this is known as the critical micelle concentration (CMC). At concentrations above the CMC the solubilisation of hydrophobic organic compounds occurs (Ying, 2006). Non-ionic surfactants tend to have a lower CMC than cationic and anionic surfactants (Ying, 2006) and are typically less active (Hepworth, 2006). This is because, unlike the cationic and anionic surfactants, they do not disassociate into ions when in solution. The polar head functional group in the non-ionic surfactants solubilises hydrophobic compounds (Ivankovic *et al.*, 2010; Hurrab & Schaumann, 2006). Ultimately, surfactants decrease the liquid-solid contact angle and reduce the surface tension of the infiltrating liquid (Darboux *et al.*, 2008)

Owing to the exponential rise in the use of surfactants in every day life there has been a growing need to consider their effect on the wider environment. Wetting agents are already present in waste waters and the effects of releasing them into natural ecosystems are a concern, particularly with regard to aquatic organisms

(Ivankovic *et al.*, 2010). Cationic surfactants in particular, have been shown to be damaging to mammalian cells and consequently tend only to be used in topical applications such as mouth washes and hair conditioner (Ivankovic *et al.*, 2010). The advantage of using the non-ionic surfactants is that less is used to acquire the same level of solubilisation and that they are less reactive. However, studies have shown that surfactant application in soils has resulted in traces of toxic elements being found in plants such as tobacco (*Nicotiana tabacum*), sugar beet (*Beta vulgaris*) and spiderwort (*Tradescantia albiflora*) (Ivankovic *et al.*, 2010). Although non-ionic surfactants are considered less active – they can still interfere with microbial activity by binding to the proteins and phospholipid membranes (Ahlstrom *et al.*, 1997). This can result in damage and death to microbes, owing to increased permeability of the cell walls (Ahlstrom *et al.*, 1997). There are concerns therefore, with promoting the increased use of surfactants for alleviating hydrophobicity in soils and growing media, and further research into the long term effects of applications on ecosystem functioning is required. Another area for consideration with organic media is the likelihood that there will be a greater concentration of dissolved organic carbon (DOC) released into the environment which has implications for water treatment (as it is costly to remove the DOC colouration from drinking water).

However, apart from monitoring their impact on respiration and on plant growth, the wider and potentially deleterious effects of surfactants were not considered in this study. The main aim of this study was to assess the optimal application rate of two non-ionic surfactants for alleviating hydrophobicity in peat and green waste-derived compost. The effect of the surfactants on the EC, pH and humic acid content of drainage water were monitored and the impact on surfactant treated composts on plant growth and microbial respiration determined.

6.3 Materials and Methods

6.3.1 Composts and Growth Trial

Individual plastic plant growth containers (1000 cm^3) were filled with either 3 distinct batches of green waste-derived compost (n = 3) or unamended horticultural peat (n = 3). The three peats were sourced from lowland sphagnum peat sites in the Ireland, and were distributed by Murphy's Scotts, Ltd. The commercial extraction of all three peat composts occurred at approximately 12 months prior to the start of the study (exact geographical source locations not available). The three green waste composts were sourced from Wormtech Ltd, a commercial large-scale in-vessel aerobic composting site, located in Monmouthshire, UK (51°37'N, 2°46'W). Each compost batch was distinct but of a similar age and maturity (approximately 12 months old). Basic characteristics of the media used in this study are described in Table 6.1.

Nutrients from the compost media and the controls were extracted using distilled water at a 1:6 w/v ratio (compost-to-distilled water). Samples were shaken at 250 rev min^{-1}, centrifuged for 10 minutes at 14,000 *g* and the supernatant was recovered after filtering through a Whatman No. 40 filter paper. Phosphate was determined colorimetrically (Murphy & Riley, 1962). NO_3^- and NH_4^+ were determined using a Skalar SAN^+ segmented flow analyser (Skalar Analytical, Breda, Netherlands) and the method was adapted from Mulvaney (1996). pH and electrical conductivity (EC) were determined from a compost/distilled water suspension (1:1, v/v), according to the method of Smith & Doran (1996) using standard probes. Water content was determined by drying in an oven at 80 °C for 24 hours. Organic matter content (OM) was determined by loss-on-ignition at 430 °C for 24 hours (Navarro *et al.*, 1991).

Two non-ionic surfactants were used- Ultraflo (Vitax Ultraflo, Ltd) and Fifty/90 (Aquatrols Vitax, Ltd). These were added to individual pots at the highest recommended application rate of 2 ml l^{-1}. Control pots had the equivalent volume of distilled water added (no surfactant). Leachate exiting the base of the pots was collected and re-poured back onto the top of the compost for a maximum of 5 times, to ensure that the surfactant was absorbed into the media. The pots were covered to negate evaporation and left for 12 hours at 20°C to equilibrate. Pots were then left to equilibrate on the wet bench in greenhouse for 48 hours prior to planting seedlings. The greenhouse was maintained at 20°C with natural daylight set to model the daytime growing season of England and Wales (May-September, 18.8 ± 1.0 °C; Met Office, Exeter, UK) according to Jones *et al.* (2009). Water was supplied by an automated irrigation system and capillary matting. Wheat seeds (*Triticum aestivum* L. cv 'Atlas') were soaked in distilled water for 12 hours and then germinated on filter paper (Whatman No.2) for 3 days until the main shoot was approximately 1 cm in length. Composts were sub-sampled (5 g) prior to planting seedlings to ascertain the moisture content, by drying in an 80 °C oven for 24 hours. Four seedlings were sown in each pot and after two weeks of growth, the smallest two in each pot were thinned out and discarded. Pots were placed in a randomised block design. Shoot height, shoot biomass and root biomass were monitored. Despite the low nutrient content of the peat composts no nutrients were added, as we were interested only in differences between the surfactant treatments compared with the control.

Table 6.1 Major characteristics of the peat and green waste composts used in the experiments. Values represent the mean ± SEM (n=6).

	Peat compost	Greenwaste compost
pH	3.7 ± 0.07	7.9 ± 0.1
Bulk density, g cm^{-3}	0.19 ± 0.01	0.42 ± 0.03
Electrical conductivity, mS cm^{-1}	0.6 ± 0.03	4.6 ± 0.6
Organic matter, %	91 ± 1	41 ± 1
Available NH_4^+, mg N kg^{-1}	0.93 ± 0.04	28.2 ± 8.6
Available NO_3^-, mg N kg^{-1}	0.14 ± 0.04	69.7 ± 2.0
Available P, mg kg^{-1}	<0.01	4.9 ± 0.3

6.3.2 Wettability

To assess wettability, composts (4 g) had 80 μl of surfactant solution added to the compost surface at different recommended surfactant concentrations: i) 0.02 ml l^{-1}, ii) 0.2 ml l^{-1} and iii) 2 ml l^{-1}. The control treatment was compost with water added containing no surfactant (n=9) (c. 80 μl distilled water). Samples were air dried at 20°C until a stable equilibrium weight had been achieved. Wettability was assessed by the water drop penetration time (WDPT) test as outlined by Doerr (1998) which is an assessment of the persistence or stability of hydrophobicity (Doerr *et al.*, 2006). Briefly, 4 drops of distilled water were placed on the surface of each sample (n=9). The median penetration time class for the drops per sample were recorded, using the 11 time classes outlined by Doerr *et al.*, (2006) (see table 6.2). Tests were terminated after 5 hours, as considerable evaporation of droplets occurs after this time.

Table 6.2 **Water Drop Penetration Time (WDPT) classes – descriptive ratings to classify the persistence of compost water repellency (adapted from Doerr *et al.*, 2006). WDPT interval SI units 'ss' is time in 'seconds'**

WDPT class	0	1	2	3	4	5	6	7	8	9	10
WDPT interval/ss	≤ 5	6-10	11-30	31-60	61-180	181-300	301-600	601-900	901-3600	2601-18000	> 18000
Persistence rating	None (wettable)	---------Slight-----------			----------Moderate------------			--------Severe---------		----------Extreme----------	

6.3.3 *Effect of Surfactants on Microbial Activity*

To assess the impact of surfactants on the microbial consortia a preliminary examination of biological activity by assessing CO_2 evolution from composts. Composts were pre-treated with the highest concentration of the two surfactants (2 ml l^{-1}), whilst the control had the equivalent amount of distilled water added (n = 3). Baseline respiration was assessed using a PP systems SR1 soil respirometer according to Jones & Kielland (2002). Composts were then wet up to container capacity (CC) following an adapted method from Cassel and Nielsen (1987). Samples were air-dried to quasi-constant weight and then all were wet with 100 ml distilled water. Respiration was monitored again for 24 hours to assess whether microbial activity differed from the water controls.

6.3.4 *Leachate Analysis*

Compost was packed into 300 cm^3 polypropylene pots with 5 drainage holes at the base of the pot (n = 3). Surfactants Ultraflo (UF) and Fifty/90 (FN) were used at the two highest recommended concentrations, namely 0.2 ml l^{-1} and 2 ml l^{-1}. Subsequently, 80 ml of surfactant solution was added to the top of the pots. The control treatment was distilled water with no surfactant added (n = 3). Pots were left to equilibrate for an hour and covered to prevent evaporation. Composts were then air dried in a fan assisted oven at 28°C. Once the pots had reached a quasi-constant weight, 100 ml of distilled water was added to the top and leachate was collected from the base of the pots. The wetting process was repeated 3 times, once gravity-free draining had ceased. The volume of water exiting was recorded each time. Each leachate was analysed for electrical conductivity (EC) and pH (Smith & Doran (1996) and for humic acid content via the analysis of UV absorbance at 254 nm (Weishaar *et* *al.*, 2003) to assess whether surfactant amendment affected the biochemical conditions of the compost and resulted in more labile organic fractions.

6.4 Results & Discussion

6.4.1 Wettability of Composts

The WDPTs for composts are presented in Table 6.3. Despite low level hydrophobicity in the control peat compost, both surfactants were effective at reducing peat water repellency. Peat was significantly more wettable at the highest application rate ($P < 0.05$, n = 9) for both surfactant types. Green waste compost controls were severely hydrophobic after air drying. UF reduced hydrophobicity to 0 (= none) only at the highest application rate ($P < 0.001$, n = 9), whereas FN achieved this at the lower concentration of 0.2 ml l^{-1} ($P < 0.001$, n = 9). For severely hydrophobic green waste compost, the FN surfactant was more effective than UF. This could mean that compared with UF, FN is a more economically viable product for horticulturalists, as more dilute applications can be used to eliminate hydrophobicity.

Table 6.3 Water Drop Penetration Time (WDPT) for peat and green waste compost treated with two non-ionic surfactants (UF = Ultraflo and FN = Fifty90) at 3 different application rates. 1 = 0.02 ml l^{-1}, 2 = 0. 2 ml l^{-1} and 3 = 2 ml in 120 ml waterl^{-1}. 'Class' relates to the qualitative assessment of water repellency (see table 6.2). Values represent the mean ± SEM (n = 9). Statistical differences are calculated using a 1-way ANOVA within each compost type and differences are denoted by ^{a}, ^{b} and ^{c}

	Peat	Peat class	GW compost	GW compost class
Control	18.8 ± 2.6^{ab}	2	320 ± 37^{a}	6
UF 1	11.1 ± 2.4^{bc}	2	159 ± 28^{b}	4
UF 2	7.9 ± 1.0^{bc}	1	14 ± 2.7^{c}	2
UF 3	2.3 ± 0.3^{c}	0	1.5 ± 0.2^{c}	0
FN 1	16.3 ± 1.9^{ab}	2	166 ± 36^{b}	4
FN 2	10.4 ± 1.4^{bc}	2	4.3 ± 0.9^{c}	0
FN 3	1.9 ± 0.1^{c}	0	0.5 ± 0.1^{c}	0

6.4.2 Effect of surfactants on plant growth

Figures 6.1a and 6.1b show the effect of surfactants on wheat shoot height over a period of six weeks. As one would expect the results differ markedly between composts, with plants grown in green waste compost shoot heights close to 30 cm, whereas the plants grown in peat averaged around half of this. This was largely due to the low nutrient content and relatively low pH of the peat media. Shoot height results do not indicate any significant differences between the treatments, meaning that from this preliminary investigation the surfactants did not appear to inhibit or promote plant growth compared with the control treatment.

Figure 6.1a Wheat shoot height - peat

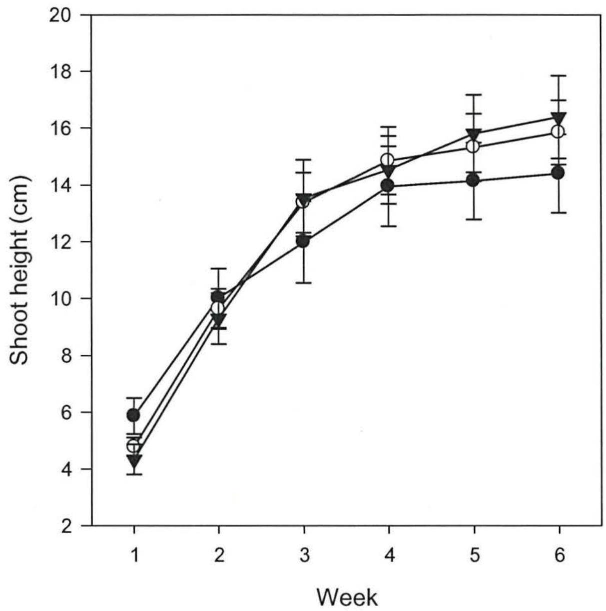


Figure 6.1b Wheat shoot height - green waste compost

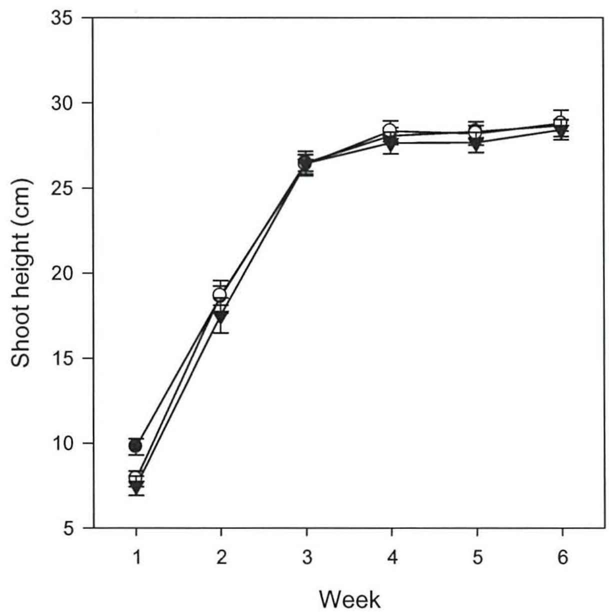


Figure 6.1a & b **Wheat shoot height during 6 weeks of growth for peat (figure 6.1a) and green waste compost (figure 6.1b) treated with two different non-ionic surfactants (2 ml l^{-1}). Control treatments are represented by the line with black circles ● and received only distilled water. The Ultraflo (UF) surfactant is the line with white circles ○ and the line with black triangles ▼ is the compost treated with Fifty/90 (FN) surfactant. Values represent the mean ± SEM (n = 9). Following a 1-way ANOVA there were no statistical differences between treatments**

Figure 6.2a Root dry weight

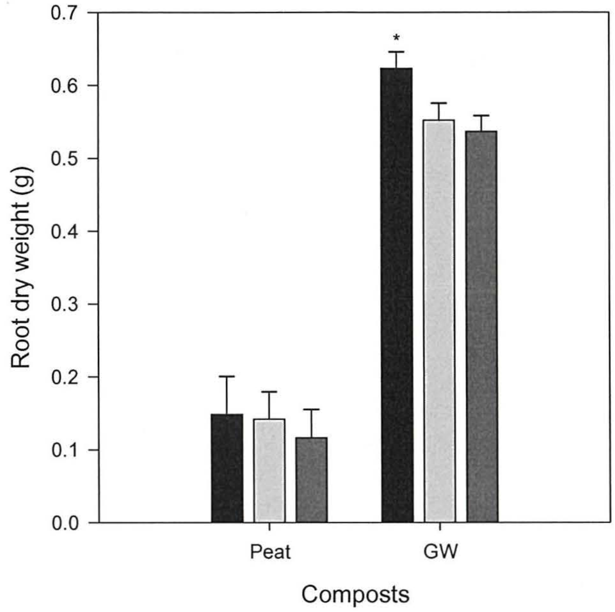


Figure 6.2b Shoot dry weight

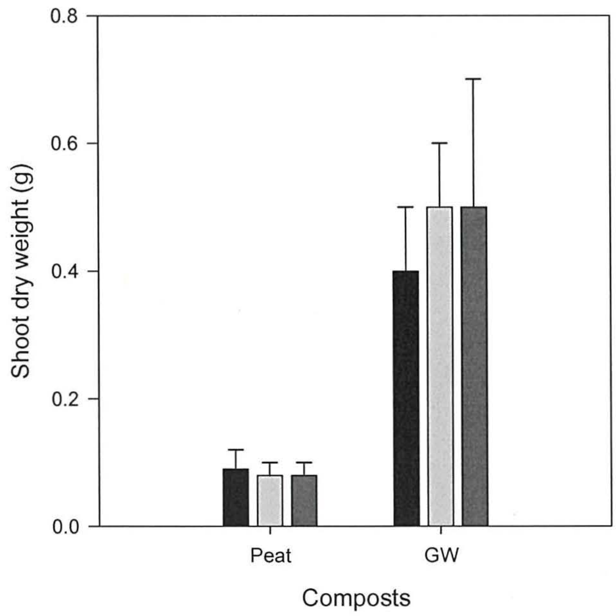


Figure 6.2a & b Root dry weight and shoot dry weight respectively for wheat grown in peat and greenwaste compost treated with two different surfactants. Black bars □ represent the control – no surfactant. Light grey bars □ represent composts treated with Ultraflo and dark grey bars □ represent composts treated with Fifity90. Values represent the mean ± SEM (n = 9). Statistical differences between treatments were calculated using a 1-way ANOVA for both peat and greenwaste compost and are denoted by * (P < 0.05)

Figure 6.2a shows the root biomass (dry weight) following the final harvest at 6 weeks. Again plants grown in the peat substrate did not grow as well as in the green waste compost but this was not a consideration for this study as we were interested in the comparison between the surfactant treated composts and distilled water controls. Root biomass was not affected in the peat. However, in the green waste compost the control root biomass was significantly higher than in both UF and FN treated media. There were no significant differences between the two surfactants. Again this could be due to a number of factors which were not realised by this study. There could have been some interference with the functioning of microbes in the rhizosphere, either through direct damage to the microbial membranes (Ahlstrom *et al.*, 1997) or through an increase of water in the pore space causing water logging and the concomitant reduction in air-filled porosity.

Figure 6.2b indicates that in both compost types, adding the surfactants had no significant effect on the shoot biomass (dry weight). This is not the case in studies using surfactants in turf systems where researchers frequently report an improvement in above ground biomass (Oostindie *et al.,* 2008; Darboux *et al.*, 2008; Doerr *et al.*, 2007). However, this tends to be evidence from trials conducted at a landscape scale and the water and solute dynamics of growing media are very different.

Further investigation is required to elucidate the mechanisms underpinning the results from the growth trial. Future growth trials with surfactants should include water stress as a variable, as this is likely to result in differences, with surfactant treated media being more resilient to the effects of drought (Guillan *et al.*, 2005; Urrestarazu *et al.*, 2008). Added to this, it would be more useful to monitor water availability than water content as this provides a more accurate indication of the water available to plants and microbes (Wallach & Raviv, 2005) and studies have shown surfactants can significantly increase the water availability in soilless media such as rock wool (Urrestarazu *et al.*, 2008) but there are none that have looked at water availability in surfactant-treated green waste compost. Despite favourable results indicating that overall the wheat was not unduly affected by the addition of surfactants to the growing media, this study did not analyse any toxicological effects of the surfactants on the wheat plants beyond tracking the impact on growth and survival.

6.4.3 Leachate Analysis

The volumes of leachate exiting the air dried media are presented in Table 6.4 for the first two wetting events. The results show that the highest concentration of FN (2 ml l^{-1}) in the green waste compost resulted in a significant decline in leachate volume ($P < 0.001$), indicating that the surfactant increased water retention. This is likely to be due to a reduction in hydrophobic spots that generally result in an increase in uneven matrix wetting owing to the presence of preferential water flow pathways (i.e. bypass flow). UF often resulted in greater volumes of leachate than in the control in both peat and green waste compost although the results were not significant.

Table 6.4 Volume of leachate exiting dried media amended with surfactants. Values represent the mean ± SEM (n = 3). Statistical differences were calculated using a one-way ANOVA for each compost type and leachate and differences are denoted by superscript letters (^{a,} and ^{b}). UF is the Ultraflo surfactant and FN is the Fifty/90 surfactant. UF/FN 2 = 0.02 ml l^{-1} and UF/FN 3 = 0.2 ml l^{-1}. The control had no surfactant added.

	Peat		Green waste compost	
Surfactant	Volume Leachate 1 (ml)	Volume Leachate 2 (ml)	Volume Leachate 1 (ml)	Volume Leachate 2 (ml)
Control	85 ± 3	83 ± 5	87 ± 3^{a}	80 ± 2^{a}
UF 2	72 ± 9	90 ± 3	75 ± 4^{a}	93 ± 1^{a}
UF 3	82 ± 4	91 ± 2	82 ± 1^{a}	92 ± 1^{a}
FN 2	85 ± 2	73 ± 6	80 ± 3^{a}	84 ± 2^{a}
FN 3	65 ± 8	78 ± 10	55 ± 4^{b}	57 ± 6^{b}

The EC, pH and UV absorbance (254 nm) for the drainage water from the first two wetting events are presented in Tables 6.5 and 6.6 as there were no differences in results for the third and fourth leachates. Leachate EC results varied but the surfactant treatments were not significantly different from the controls. pH again varied but on the whole the surfactant treatments remained similar to the control. The last wetting in the green waste compost did differ from the control for the UF 0.2 treatments with regard to pH (P < 0.5, n = 3). Studies have shown that many humic substances become solublised at pH > 6.5 (Hurrab & Schaumann, 2006). As humic substances reduce the surface tension of liquids, then a decrease in hydrophobicity at a higher pH could partly be due to this mechanism (Hurrab & Schaumann, 2006). The last green waste compost leachate (L4) for FN 0.2 – the highest concentration application rate - saw the most significant decrease in water repellency (Table 6.3) and a concurrent rise in pH (Table 6.6). It is possible then, that this effect was owing to the increased solubilisation of humic substances and the concomitant lowering of the surface tension of the water. However, the leaching of humic substances did not correlate with this result.

Overall there were no clear trends with the UV254 absorbance results and on the whole treated composts did not differ significantly from the control. Only in the third leachate in the green waste compost (L3) were results significantly different to the control – with the UF 0.2 treatment seeing the largest absorbance value. It is worth noting that NO_3^- content can interfere with UV absorbance readings at 254 nm (Weishaar *et al.*, 2003) and the green waste compost solution contained comparatively high levels of NO_3^- (69.7 mg N kg^{-1}) which could have affected the absorbency values to a small extent throughout (ca. 10% of reading).

Table 6.5 **Table of pH and EC (mS cm^{-1}) values for leachate (L1 to L4) from peat and green waste compost treated with surfactants Ultraflo (UF) and Fifty/90 (FN). Surfactants were applied at two rates (0.2 ml and 0.02 ml in 120 ml distilled water). Control had no surfactant added. Values represent the mean ± SEM (n = 3). Statistical differences were calculated using a one-way ANOVA within each compost type and leachate number. Differences are denoted by $^{a, b \text{ and } c}$**

Peat treatment	**pH**				**EC**			
	L 1	L 2	L 3	L 4	L 1	L 2	L 3	L 4
Control	4.64 ± 0.10	4.21 ± 0.08	4.14 ± 0.07	4.42 ± 0.13	0.114 ± 0.04	0.195 ± 0.05	0.227 ± 0.07	0.092 ± 0.01
UF 0.02	5.06 ± 0.07	4.31 ± 0.10	4.30 ± 0.24	4.78 ± 0.29	0.099 ± 0.03	0.199 ± 0.07	0.315 ± 0.18	0.125 ± 0.03
UF 0.2	4.64 ± 0.05	3.99 ± 0.13	4.09 ± 0.12	4.80 ± 0.19	0.195 ± 0.09	0.428 ± 0.19	0.370 ± 0.14	0.058 ± 0.01
FN 0.02	4.73 ± 0.09	3.86 ± 0.16	3.77 ± 0.15	4.27 ± 0.09	0.098 ± 0.02	0.256 ± 0.01	0.286 ± 0.09	0.086 ± 0.03
FN 0.2	4.50 ± 0.20	4.20 ± 0.15	4.19 ± 0.13	4.21 ± 0.19	0.437 ± 0.28	0.139 ± 0.05	0.119 ± 0.05	0.088 ± 0.03
Green waste compost								
Control	7.31 ± 0.08	7.67 ± 0.19	8.12 ± 0.19	8.17 ± 0.14^{ab}	0.975 ± 0.27	2.517 ± 0.91	2.892 ± 0.82	1.622 ± 0.18
UF 0.02	7.18 ± 0.08	7.47 ± 0.18	8.00 ± 0.16	7.94 ± 0.15^{a}	1.018 ± 0.30	3.274 ± 1.47	3.880 ± 0.61	1.628 ± 0.23
UF 0.2	7.59 ± 0.05	7.61 ± 0.17	8.04 ± 0.11	8.14 ± 0.12^{abc}	0.733 ± 0.04	5.470 ± 0.92	6.193 ± 0.69	1.152 ± 0.16
FN 0.02	7.26 ± 0.28	7.89 ± 0.24	8.11 ± 0.27	8.71 ± 0.09^{bc}	0.515 ± 0.18	3.282 ± 0.78	4.823 ± 1.31	2.001 ± 0.27
FN 0.2	7.68 ± 0.19	7.78 ± 0.30	8.09 ± 0.23	8.75 ± 0.08^{c}	1.041 ± 0.23	1.679 ± 0.76	2.388 ± 1.27	1.203 ± 0.06

Table 6.6 **Analysis of compost leachate (L1 to L4) collected during rewetting air dried peat and green waste. Leachates were then analysed for UV absorbency at 254 nm to assess level of humic substances. Statistical differences were calculated using a 1-way ANOVA in SPSS (v.16) for each compost type with the main factor being surfactant content. Values are the mean of samples (n = 3) ± SEM. Significant differences between treatments at the $P < 0.05$ level are denoted by ^{a} and ^{b}.**

Surfactant application	UV Absorbency 254 nm							
	Peat				Green waste compost			
	L 1	L 2	L 3	L 4	L 1	L 2	L 3	L 4
Control	0.56 ± 0.30	1.07 ± 0.36^{ab}	1.81 ± 0.60	1.13 ± 0.20	1.71 ± 0.37	3.22 ± 0.55	5.47 ± 0.96^{a}	7.79 ± 2.24
UF 0.02	0.39 ± 0.02	1.02 ± 0.27^{ab}	2.07 ± 0.79	1.19 ± 0.52	2.30 ± 0.58	9.41 ± 4.37	6.77 ± 1.25^{a}	5.85 ± 1.07
UF 0.2	0.62 ± 0.20	2.63 ± 0.60^{b}	3.39 ± 0.74	1.25 ± 0.52	2.29 ± 0.59	10.6 ± 3.92	20.6 ± 4.38^{b}	7.98 ± 1.81
FN 0.02	0.20 ± 0.05	1.07 ± 0.27^{ab}	1.86 ± 0.49	1.01 ± 0.27	0.82 ± 0.05	6.13 ± 1.54	12.7± 2.35^{ab}	9.76 ± 1.86
FN 0.2	0.37 ± 0.20	0.48 ± 0.07^{a}	1.05 ± 0.52	1.59 ± 0.53	1.10 ± 0.10	2.61 ± 0.05	8.07 ± 4.26^{a}	6.93 ± 1.86

6.4.2 *Effect of Surfactant on Microbial Activity*

Non-ionic surfactants are biodegradable (Zhao *et al.*, 1998) and so could have provided a readily available substrate for microbial activity. This would have been evident by a rise in respiration (substrate induced respiration). Equally there is evidence to suggest that non-ionic surfactants can damage or destroy the cell wall constituents of microbes (Ahlstrom *et al.*, 1997) and this could have been evident by a decline in microbial activity. Figure 6.3a monitors the respiration following a cycle of drying and rewetting in peat treated with surfactants. Figure 6.3b is the respiration for green waste composts subjected to the same regime. Respiration rate for the green waste composts treated with the two surfactants did not differ significantly to the control (no surfactant) at any point during the 24 hours period. Conversely, peat treated with UF had significantly lower rates of CO_2 evolution than the control ($P <$ 0.05, n = 3) between 5 hours and 12 hours ($P < 0.05$, n = 3). There were no significant differences between the two surfactants despite FN maintaining a higher rate of respiration throughout the 24 hours in both composts. Interestingly, for the control peat compost, respiration began to decline at around 15 hours and at this point the two surfactants followed a very similar trend to each other, except that UF was at a lower rate than FN. This would have been due to ambient temperature fluctuations during analysis. However, in the green waste compost treated with FN there was overall greater biological activity but still comparable with the control, providing an early indication that the green waste compost treated with both UF and FN resulted in little change to the functioning of the biota. Although the respiration rate indicates some demise owing to the treatment with UF in the green waste compost, it was more pronounced in the peat. It is possible that UF incurred higher retention of water in the micropores of the peat media resulting in a decline in air filled porosity and a decline in microbial activity. Overall in both composts FN appeared to have a more favourable effect on microbial activity but the reasons for this are unclear.

Figure 6.3a

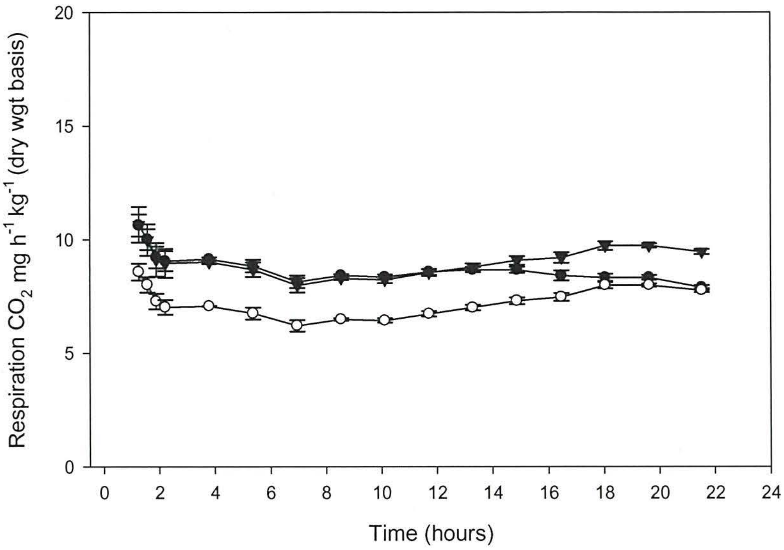
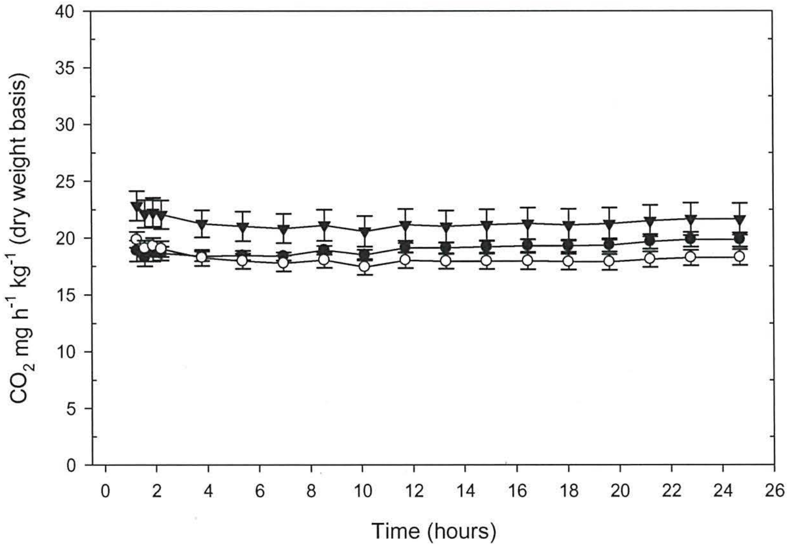


Figure 6.3b



Figures 6.3a & b Respiration of peat (Fig 6.3a) and green waste compost (Fig 6.3b) following rewetting. Composts were treated with surfactants prior to desiccation and rewetting. Lines with black circles ● represent the control (no surfactant), clear circles ○ represent compost with Ultraflo (UF) surfactant and black triangles ▼ represent composts treated with Fifty/90 (FN) surfactant. Surfactants were applied at the highest rate (0.2 ml in 120 ml of distilled water). Values represent the mean (n = 3) ± SEM.

6.5 Conclusion

This was a comparative study firstly to assess the most efficient application rate for two non-ionic surfactants to reduce the hydrophobicity of green waste and peat compost. It was clear that FN was more effective at achieving this in the green waste compost – where the control was severely hydrophobic – at the medium concentration of 0.2 ml l^{-1}. UF achieved the same total reduction in hydrophobicity at the highest concentration of 2 ml l^{-1}. This means that FN would be a more competitive wetting agent both on an economic and environmental basis, owing to smaller amounts achieving similar results to UF at lower concentrations. The impact of the surfactants on plant growth was also addressed by a bioassay with wheat. Plants grown in the peat compost treated with the surfactants, appeared to be unaffected compared with the control for root and shoot growth. Green waste compost treatments were comparable with the control for above ground biomass. However, the control had significantly higher biomass for root growth and this effect warrants further investigation before recommending their use.

The volume of leachate draining after wetting indicated that FN was more effective at retaining water added providing further support for favouring this surfactant over UF. Water shortages are becoming common place and methods for lowering consumption are increasingly important in agriculture, horticulture and amateur gardening. Overall, initial results are that FN would be a suitable wetting agent for green waste compost in containerised media. Three distinct batches of green waste compost were providing a level of confidence that this result was not batch specific and issues such as feedstock variation may not affect the effectiveness of the surfactants.

This study did not measure the impact on water availability for plants and microbes following surfactant application – i.e. water potential in the substrates. We also did not concentrate on the ability of surfactant treatments to sustain plants at times of water stress. Finally, it is important to assess the effects on the biota and wider ecosystem of the widespread use of surfactants.

Ultimately the FN was the most effective at reducing WR in GW to 0 i.e. none, at the median application rate of 0.02 ml in 120 ml. UF reached this state at the higher rate of 0.2 ml in 120 ml.

CHAPTER 7: GENERAL DISCUSSION

7.1 Discussion of results

This thesis has attempted to achieve two key objectives.

1. Further the understanding of the parameters influencing green waste quality for use as a peat replacement in containerised media, with a particular focus on the wettability of composts.
2. To find ways of improving the end quality of green waste compost, with a view to adding value and diversifying end uses.

The company partially funding this research project were processing 2500 tonnes of green waste compost but there was a lack of consumer interest in the end product except for high volumes of compost, sold in low cost sales (e.g. for restoration work). Heyworth (2000) identified as far back as 2000 that over production of waste-derived compost was likely to out strip demand as consumer's struggled to see value in waste-derived compost. Coupled with the fact that the predominant fraction being composted was green waste – being relatively low in plant and microbial nutrients and unpredictably prone to phytotoxicity, improving compost end quality was seen as a priority by the company. Wormtech were keen to develop composts that could compete in the high value end of the horticultural market, filling the projected gap left by the successive phasing out of peat-based composts. Sterilizing the green waste compost saw improvements in germination rate and plant growth, making the composts comparable to the leading peat-based control (chapter 3). Sterilization is a traditional technique used by gardeners in home composting and is generally employed to remove harmful pathogens and weed propagules. However, sterilising the composts, particularly autoclaving, brought about changes to the chemical properties of the compost. One main effect was the increased availability of macronutrients such as NO_3^-. As green waste compost typically has low plant available N, this result has interesting implications concerning its use as containerised media. Sterilizing the composts is likely to remove any need for amendment with artificial fertilisers, saving additional cost. It is clear though that some of these chemical changes could result in phytotoxic conditions and the effects of sterilization are unlikely to be easily standardised. The initial characteristics of the feedstock will exert a key influence on these changes and feedstock variation is an ongoing problem affecting quality in green waste composting. The germination part of this trial indicated an almost complete eradication of phytotoxic conditions via microwaving, although the precise changes responsible for this result were not uncovered.

At the time of this experiment the company funding the research were adhering to the strict turning regime required to gain their PAS: 100 certification (BSI, 2005). This incurred high inputs in terms of time and energy associated with this regime. Research shows that despite reaching the required thermophilic temperatures, sanitisation is not always guaranteed nor sustained and the re-colonisation of composts by pathogens is likely. This poses the question as to whether the stipulations regarding sanitisation in the PAS: 100 are sufficiently evidence-based. It is unlikely that the temperature regulations will change in the near future but there is scope for individual operators to sterilize compost prior to bagging to increase the availability of nutrients and reduce phytotoxicity. This would be particularly relevant when developing a range of high quality peat replacements marketed for plants with high nutrient demand.

In order to broaden the variety of marketable products from green waste, Wormtech were very interested in understanding more about the possibility of developing a range of compost teas (chapter 4), both as plant nutrient feeds and for biocontrol purposes. Further, there was a need for research into the efficacy of waste-derived compost teas, particularly green waste compost teas as studies examining this feedstock were relatively absent in the literature. Our initial trials (Appendix 1) demonstrated that non-aerated tea derived from relatively immature green waste compost suppressed three economically important soil borne plant pathogens (*Rhizoctonia solani, Aphanomyces euticheS* and *Gaeumannomyces graminis*). However, growth suppression was only successfully studied *in vitro*, rather than in a soil-plant system. As suppression *in vitro* does not equate to suppression in the soil system, without subsequent research at this scale the results had limited applicability. However, this preliminary trial did serve to support the hypothesis that the microbial consortia of the compost tea can suppress fungal growth, most probably via antagonism and/or competition for resources (see Appendix 1). Most studies on the suppression of soil borne pathogens disagree over whether immature or mature solid state composts are disease suppressive of conducive and very few have examined whether compost tea should be prepared from mature/immature composts. Any phytotoxic effects in immature solid state compost are likely to be amplified in compost teas, as allelopathic chemicals are often water soluble and more prevalent in immature composts. The work presented in chapter 4 demonstrates the highly interactive effects of the microbial biota and chemistry on germinating seeds, and importantly that this response is entirely species specific, therefore addressing a potential shortcoming of the PAS: 100 Quality Assurance Protocol (BSI, 2005). Phytotoxicity/maturity is assessed as part of this 'industry standard' by way of a germination bioassay using tomatoes as the bio-indicator. However, we have used four different plants (cauliflower, tomato, alfalfa and wheat), and the results from our bioassays clearly indicated that conditions in the tea affected each species differently and that compost tea made from mature and immature composts can both be phytotoxic or growth promoting.

For the compost industry, developing reliable compost teas is likely to be an even greater challenge than quality assurance in compost. Owing to factors such as the variation in feedstock and the potential inclusion of decaying plants releasing water-soluble allelochemicals, guaranteeing a tea that does not exhibit phytoxicity across a broad range of plants is unlikely. Equally, until a reliable test for assessing the suppressive nature of composts and compost teas is found then sufficient standards for ensuring a consistent product can be produced and marketed. Composting plant material that releases allelopathic chemicals during decomposition warrants further work, particularly tracking the stage at which the prohibitive effect of allelochemicals subsides.

The lack of wettability in green waste composts once the water content was low was one main reason for lack of re-purchase of Wormtech's green waste compost marketed as containerised media. Chapter 5 examined the effect of drying and rewetting on green waste compost, peat and vermicompost hydrophobicity and physical properties. As each compost type was the result of entirely different decomposition processes, it was likely that there would be differences in the wettability and physical stability. As vermicomposts are reported as having a greater percentage of polysaccharides and a more homogenous texture than green waste compost, we were interested to see if it was more comparable to peat than green waste compost in response to increased drying. By amending green waste vermicomposting raw waste with paper pulp and sewage sludge, we were able to see the effect of feedstock.

Clearly amending the green waste vermicompost with 30% paper pulp (V2) resulted in vermicompost with a highly homogenous peat-like appearance and the highest volumetric water content at container capacity, out of all the composts. Initial results therefore indicated that V2 might be a superior peat replacement compared with the other composts. However, once V2 was subjected to drying there were quasi-irreversible changes to the structural stability of this compost. Based on the parameters set out in the PAS: 100 protocol, this fact would not have been indentified at the quality assessment stage. Regardless of feedstock amendment, the vermicompost was far more wettable than green waste compost and peat at low water contents. This had implications for the company as to whether vermicompost was a cost-effective operation to ensure a more wettable compost and whether it would improve and sustain sales. The different results concerning wettability in peat, green waste compost and vermicomposts produced using different feedstocks indicates that water repellency was governed by the decomposition process rather than the feedstock used. Some studies support this view but add to this, the issue of maturity. So for peat, less decomposed peat is both less repellent and more elastic and the reverse seems to be the case for green waste compost. More recent work is beginning to show that certain organic compounds are more prevalent in water repellent soils than in hydrophilic ones (Atanassova & Doerr, 2010). As both the decomposition process and the feedstock exert a key influence over the carbon fractions in composts, the potential for soil hydrophobicity studies to elucidate compost science in this regard, should not be overlooked. It has been shown that the carbon fractions in compost can determine the level of hydrophobic compounds in composts. Owing to the limitations of the models used to predict important parameters influencing water retention in the compost only a preliminary understanding was acquired. It would appear that the hysteresis between wetting and drying cycles is less in green waste compost compared with vermicompost and peat but water retention is lower. More work on the hydraulic properties of the three composts was required to yield a comprehensive understanding of the mechanisms controlling water transport.

Finally it was clear that the non-ionic surfactants increasingly used in fair ways to alleviate hydrophobicity and dry spots, worked well in this regard for the green waste compost. One brand behaved more favourably in terms of wettability, microbial respiration and plant growth (Fifty90). Further work is required to assess the cost/benefit if using wetting agents in green waste compost, over amending with vermicomposts to increase wettability and water retention. Subsequent research has shown that compost hydrolysates can be isolated from composts and act as biosurfactants in growth media (Paradelo *et al.*, 2009). If biosurfactants can be isolated from green waste composts, this could represent further diversification of end markets. The question of whether biosurfactants are likely to be less toxic than using chemical surfactants is unanswered.

The self organisation of the soil/compost system describes how the soil ecosystem actively modifies itself over relatively small time scales. A key mechanism for this modification is the response of the biota to cycles of wetting and drying (chapter 5, 6 & appendix 1). Both microbial and root exudates and decaying litter result in the release of hydrophobic compounds which influence the level of physical stability on a soil/compost. The internal perturbations caused by wetting and drying and the activity of the biota can cause and determine changes in the physical structure – porosity, aggregate formation and pore size distributions. The physical structure of soils and growing media is important in determining moisture conditions, biological activity and the capacity of soils/composts to store carbon. In growing media more work is required to understand the relative importance of the type of decomposition process, whether it be vermicomposting (chapter 5), aerobic composting or raw waste to land (appendix 2), on the ability of the system to modify the physical habitat without compromising its effectiveness as a plant growth substrate. The same is true for the impact of different feedstocks on physical characteristics and the development of phenomenon such as hysteresis between desorption and adsorption and water repellency. The interaction of feedstocks and decomposition process in relation to physical properties is an interesting area that requires further work, both on waste-derived peat replacements but also on the effects of various organic inputs to soils.

7.2 Further Work

There is considerable scope for an interaction of the themes in chapters 3 – 6. For example, the physical structure of soils/composts could play an important role in the level of disease suppression or indeed disease incidence in soils (chapter 4 linking in with chapter 5 and appendix 1). Sterilizing composts clearly has an effect on the biota and chemical properties but the effects on porosity and water availability for example are unknown. The research conducted in this project established various methods for improving the end quality of green waste and researching a range of end uses but further work on the relative merits and cost effectiveness of each is required.

7.2.1 Assessing how effective the sanitisation regime is in eradicating pathogens from waste-derived composts and whether the sterilization of composts represents a sensible option for adding value to green waste compost as containerised media. This should be studied over several seasons to assess the impact of variable green waste feedstocks. Vermicompost sterilization trials would be interesting as there are clear differences between it and aerobically produced compost. Greater analysis of the effects of sterilization on physical properties and water characteristics is required

7.2.2 As the composting industry is underpinned by the need to reduce greenhouse gases there is an urgent need to conduct life cycle assessments (LCA). Sites have become increasingly centralised, incurring greater transportations along with an increasing on-site reliance on fossil fuel based technologies. The embodied carbon of infrastructure is also worth studying. A comparison with alternatives waste management options such as energy from waste and applying raw waste to land would be valuable.

7.2.3 Spatial dynamics of hyphal movement through compost. Does compost structure and matric potential of the compost affect the movement of pathogens from one seedling to the next?

7.2.4 This project identified compost maturity as a key element of compost quality and this parameter is of particular importance in containerised media where phytotoxic effects are often amplified. Finding reliable predictive tests for assessing compost phytotoxicity/maturity are still required.

7.2.5 Although the possibility is remote, a predictive test for assessing the disease suppressive potential of composts and compost teas would be a key development for securing markets for this product.

7.2.6 Re-evaluation of the parameters selected to provide compost quality assurance in the PAS: 100, particularly addressing the lack of physical parameters. The bioassay with tomatoes only is insufficient test for phytotoxicity

7.2.7 Isolating biosurfactants from green waste compost for use in horticulture and following adequate research, potentially developing their wider use at the landscape scale, such as in golf greens. What is the effectiveness of surfactants, residence time and wider environmental implications compared with chemical/synthetic surfactants?

7.3 Wider Research

7.3.1 Considering "soils contain the greatest reservoir of biodiversity" (Feeney *et* al., 2006 p151) evaluating the relative merits of applying raw and composted green waste to soils warrants further study, particularly in relation to interactive properties such porosity, aggregate formation and stability, hydrophobicity and carbon storage and microbial diversity.

7.3.2 Diversifying the waste management options for green waste – such as energy from waste. More woody recalcitrant feedstocks – typical in winter months – could be used as fuel in biomass plants to working towards the UK renewable targets . A LCA of this option compared with composting would be valuable

APPENDIX 1: DISEASE SUPPRESSION (*IN VITRO*) OF THREE PLANT PATHOGENS USING IMMATURE GREEN WASTE COMPOST TEA

Introduction

Compost tea has been used as an effective fertiliser increasing plant yields (Haggag & Saber, 2007; Hargreaves *et al.*, 2008) and as a foliar spray to reduce plant pathogens (Yohalem *et al.*, 1994; Haggag & Saber, 2007). Most reports of effective disease suppression via compost tea have been using nutrient-rich manure-based compost although relatively few studies have looked at the efficacy of compost teas created from greenwaste (Noble *et al.*, 2006). Scheuerell & Mahafee (2002) did an extensive review on the use of compost teas for plant disease control. Compost tea differ from compost extracts by the fact that in teas there has been a fermentation process, compared with an extract which is filtered compost after mixing in a solvent. Compost tea production time can be anything from 1 day to several weeks and the 'steepage' process is either aerated or non-aerated (Ingram & Millner, 2006).

In this study three economically important plant pathogens were selected to test the disease suppression potential of compost tea produced from unstable greenwaste-derived compost. *Rhizoctonia solani*, (3 isolates), *Aphanomyces euteiches* (UK strain) and *Gaeumannomyces graminis* (take-all in wheat) were chosen on the basis that they infect a wide range of economically important plants globally. Studies have shown that sufficient maturation phase for solid state compost is required to produce composts that are disease suppressant (Veekan *et al.*, 2005; McKellar & Nelson, 2003; EL-Masry *et al.*, 2002; Sidhu, 2001); whereas others have reported a loss of disease suppression when highly stable composts are used (Danon *et al.*, 2007). Furthermore, extended compost storage times have been shown to decrease the suppressive potential of composts in soils (Sidhu *et al.*, 2001). To date, no studies have focused on the relevance of compost maturity in the preparation of compost teas for use as disease suppressants. This study aimed to demonstrate if compost tea (prepared from immature green waste compost) inhibited the growth of the selected pathogens, measured via hyphal growth (*in vitro*). The compost tea treatments were designed to show whether pathogen growth suppression was due to the antagonistic colonisation/resource competition from compost tea microbial consortia, or whether compost tea physico-chemical conditions were an inhibitive factor.

Methods & Materials

Compost tea was prepared using green waste that was approximately 1 week past the thermophilic phase and was obtained from a commercial composting facility operated by Flintshire County Council, North Wales (53°17′14′′N 3°12′11′′W). Compost had a starting moisture content of 40 ± 5 %. Compost tea was produced by mixing a 50:50 v/v ratio of compost to distilled water and shaking at 200 rev min^{-1} for 7 days at 20ºC. Although this process allowed minimal aeration the product would not qualify as an aerated compost tea (Hagaag & Saber, 2007). Compost teas were then filtered through sterile muslin cloth. The teas were centrifuged for 10 minutes at 14,000 *g* and the supernatant filtered through Whatman No. 540 filter paper. The supernatant was then centrifuged and filtered again due to the high volume of organic precipitates retained after the first filtering. At this stage the tea was referred to as filtered (f) compost tea and still retained its microbial consortia. Aliquots of the filtered tea were aseptically microfiltered (m) through a 0.25 µm filter (Millipore Corporation) to remove all of the biota, or autoclaved (a) at a temperature of 120 °C (393 K) for 20 minutes (pressure of 150 kPa), which in addition to sterilising the tea had the added effect of perturbing the chemistry due to the combined effects of heat and pressure. The filtered tea (CTf) retained the microbial biomass, microfiltering removed the biota (CTm) and autoclaving the tea (CTa) destroyed the biota and altered the chemical conditions (chapter 3 and 4).

Fungal isolates *Rhizoctonia solani*, (3 isolates – R42, R63, R106), *Aphanomyces euteiches* and *Gaeumannomyces graminis* were maintained on PDA (potato-dextrose agar) (Oxoid Ltd) in Petri dishes – standard 90 mm x 14 mm (VWR™ International).

PDA plates were prepared in sterile conditions and amended with compost tea (10%) at the preparation stage, i.e. 600 ml of media had 60 ml of either CTf, Ctm or CTa added ($n = 4$ per treatment for 5 pathogens, $n = 80$ in total including controls). Controls had sterile distilled water added at the same dilution rate. Disks (10 mm diameter) were taken from the leading edge of 2 day old fungal cultures and placed in the centre of each Petri dish. Plates were inverted and incubated at 25°C in the dark.

Hyphal growth from the centre of the disk was measured in four random directions across the plate every 24 hours and averaged. Measurements continued until the hyphae had reached the edge of the plate or growth was impeded for more than 48 hours by the tea consortia or conditions.

Results

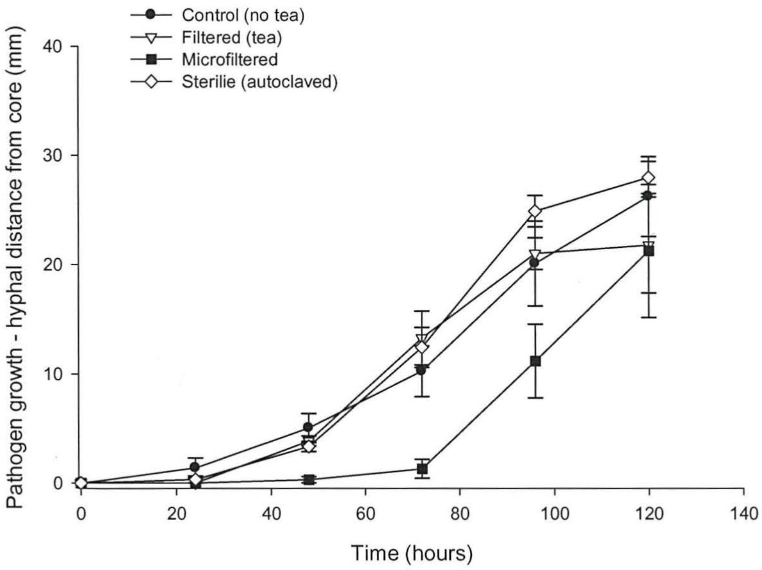


Figure 1: ***Aphanomyces eutheiches***

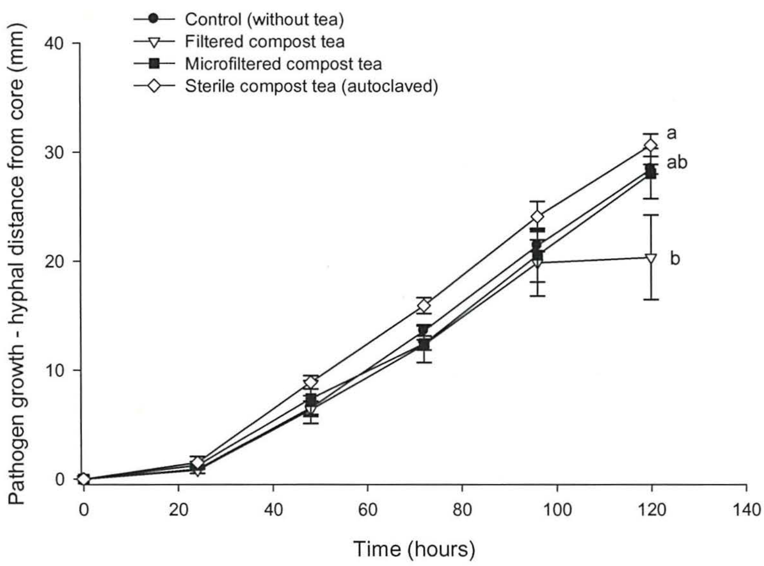


Figure 2: ***Gaeumannomyces graminis***

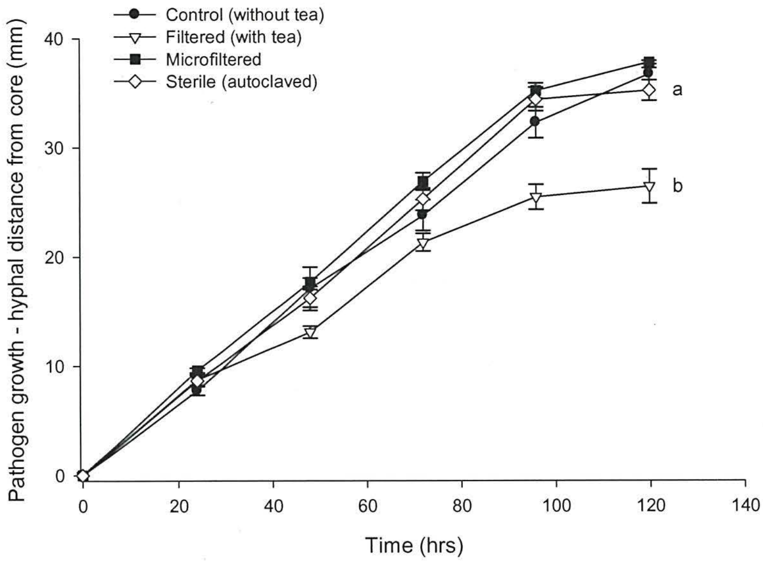


Figure 3 ***Rhizoctonia solani*** **(isolate R42)**

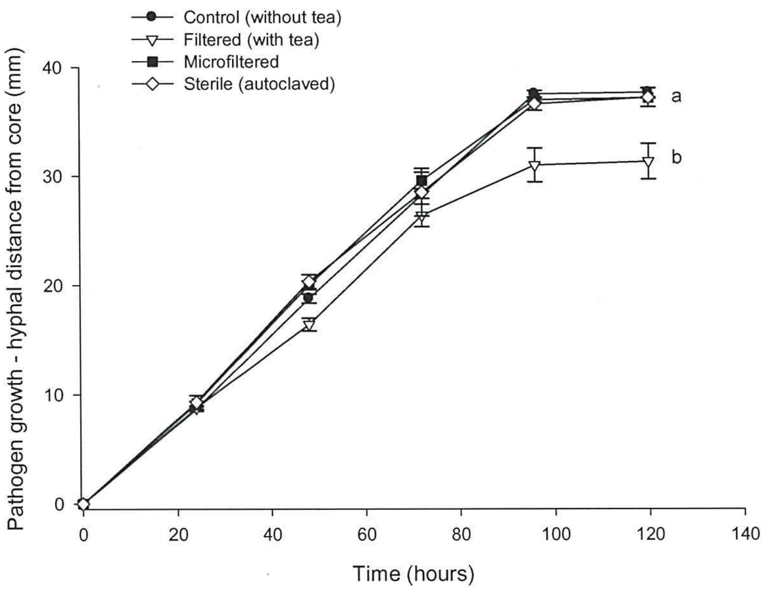


Figure 4 ***Rhizoctonia solani*** **(isolate R63)**

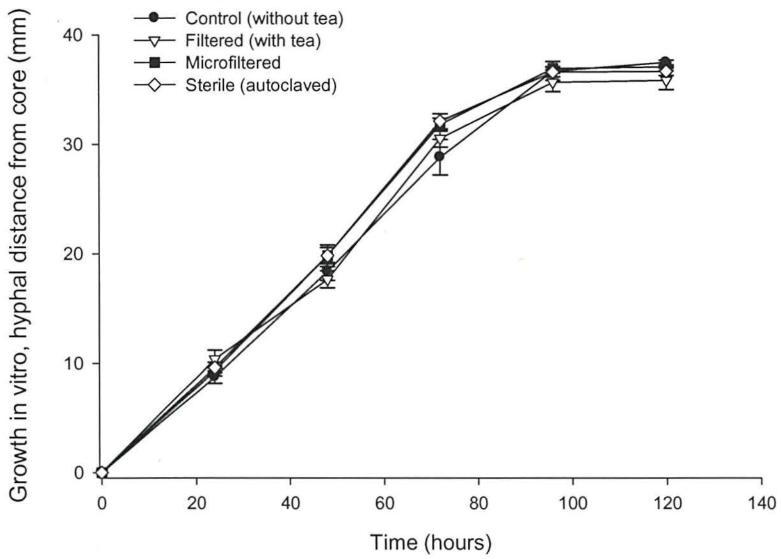


Figure 5 ***Rhizoctonia solani*** **(isolate R106)**

Figures 1 – 5 Hyphal growth of 3 plant pathogens (*in vitro*) when grown on compost tea amended PDA. Letter codes [a] and [b] denote a significant difference between treatments (one-way ANOVA and Tukey multiple comparison test, $P < 0.05$). Data points are the mean of 4 replicates + SEM.

Figure 6 ***Rhizoctonia solani*** **on PDA amended with compost tea (CTf), with bacteria impeding the growth of fungal hyphae from the core at 96 hours**

Figure 7 ***Rhizoctonia solani*** **on PDA amended with sterile compost tea (CTa) demonstrating complete coverage of the plate by pathogen hyphae at 96 hours**

Overview

Figures 2, 3 and 4 show that (*in vitro*) hyphal growth in *Gaeumannomyces graminis, Rhizoctonia solani* (R42) and *Rhizoctonia solani* (R63) was significantly reduced in the PDA amended with the compost tea treatment that retained the microbial consortia (CTf). This is most likely due to antagonistic colonisation via the compost tea microbes or competition for nutrients due to the compost microbial biomass limiting fungal growth. There were no differences between pathogen growth in CTa and CTm for *Rhizoctonia solani* and *Gaeumannomyces graminis* (see Figures 2-5) and hyphal growth was largely unaffected, indicating that compost tea physico-chemical conditions alone were not suppressive.

Figure 6 illustrates how the compost tea bacteria in the CTf amended PDA physically and/or biochemically inhibited hyphal growth of *Rhizoctonia solani* (R42), compared with Figure 7 - where CTa amended PDA resulted in the spread of *Rhizoctonia solani* across the whole plate. Results indicate that for two of the three plant pathogens, growth suppression (*in vitro*) was most likely due to antagonistic colonisation/resource competition (Berg *et al.*, 2005; Fravel, 1988; Agrios, 2005) via the compost tea microbial biomass.

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