Improving Compost Process Efficiency by Controlling Aeration, Temperature and pH

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Doctoral thesis
Swedish University of Agricultural Sciences
Uppsala 2005

Acta Universitatis Agriculturae Sueciae
2005:103

ISSN 1652-6880
ISBN 91-576-6902-3

Tryck: SLU Service/Repro, Uppsala 2005

Abstract

Sundberg, C. 2005. Improving compost process efficiency by controlling aeration, temperature and pH. Doctoral dissertation. ISSN 1652-6880, ISBN 91-576-6902-3

Biowaste composting is rapidly increasing, and many composting plants in Scandinavia have had problems with low pH during the process. The aim of this thesis was to develop methods to improve process efficiency in large-scale composting. The investigations focused on alleviation of acid-related process inhibition and the interrelationships between decomposition rate, temperature, aeration, evaporation and oxygen concentration. Composting experiments were performed at laboratory-, pilot- and full-scale, and the microbial and physical processes were modelled and simulated.

In composting of food waste a prolonged initial acidic phase can occur, resulting in low degradation rate. In successful composting, the initial phase is followed by high-rate composting at pH values above neutral. A combination of temperature above 40 °C and pH below 6 severely inhibits the composting process.

Experiments at large-scale composting plants showed that it is possible to increase the activity and shorten the acidic phase by increasing the aeration rate, even though the temperature remains above 40 °C.

When composting source-separated household waste in controlled 200-litre reactor experiments, the decomposition of organic matter during the high-rate phase was faster at 55 °C than at 40 or 67 °C. The ammonia emissions at 67 °C were more than double those at 55 or 40°C.

Experiments were also performed in rotating 3-litre reactors, which were fed daily with fresh waste and water. In reactors with a starting culture more than four times the size of the daily feed, a well-functioning process was established. In reactors with a starting culture less than twice the daily feed, the pH decreased and the composting process failed.

A dynamic model was developed for the initial self-heating phase of batch composting. It was possible to describe the process in simulations with a mesophilic microbial community tolerant to low pH and a thermophilic community inhibited by low pH.

Simulations showed that in large-scale composting, the water loss is mainly determined by the accumulated decomposition. The process temperature has very little effect on the water loss.

Keywords: carbon dioxide, composting, decomposition, energy, fatty acids, mesophilic, organic waste, temperature, thermophilic

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Sammanfattning

Sundberg, C. 2005. Förbättrad kompostprocesseffektivitet genom styrning av luftning, temperatur och pH. Doktorsavhandling.

Kompostering av källsorterat hushållsavfall byggs ut i snabb takt, men många anläggningar i Norden har haft problem med lågt pH i kompostprocessen. Syftet med avhandlingsarbetet var att utveckla metoder för processeffektivisering vid storskalig kompostering. Undersökningarna fokuserade dels på att motverka processhämning orsakad av organiska syror, dels på att förklara sambanden mellan nedbrytningshastighet, temperatur, luftning och syrekoncentration. Experiment utfördes i såväl laboratorie- som pilot- och fullskala. Dessutom modellerades och simulerades de mikrobiella och fysikaliska processambanden.

En utdragen sur fas kan uppstå vid kompostering av matavfall, vilket leder till långsam nedbrytning. Vid väl fungerande kompostering följs den sura fasen av en fas med snabb nedbrytning vid neutralt eller basiskt pH. En temperatur över 40 °C i kombination med ett pH-värde under 6 är starkt hämmande för processen.

Experiment vid storskaliga anläggningar visade att det är möjligt att öka nedbrytningen och korta den sura fasen genom ökad luftning, även om man inte lyckas hålla temperaturen under 40 °C.

Vid kompostering av källsorterat hushållsavfall i en experimentreaktor, var nedbrytningen efter den sura fasen snabbare vid 55 °C än vid 40 och 67 °C. Ammoniakemissionerna var mer än dubbelt så stora vid 67 °C som vid de lägre temperaturerna.

Experiment genomfördes också i reaktorer som dagligen tillfördes färskt avfall och vatten. Med en startkompost som var mer än fyra gånger än större än den dagliga tillsatsen etablerades en fungerande process. I reaktorer med en startkompost som var mindre än dubbelt så stor som den dagliga tillsatsen sjönk pH-värdet och nedbrytningen avstannade.

En dynamisk modell konstruerades för att simulera den initiala sura fasen. Processen kan beskrivas genom simulering av en mesofil mikroflora som tål lågt pH och en termofil mikroflora som hämmas av lågt pH.

Simuleringar visade att i storskalig kompostering bestäms uttorkningen främst av nedbrytningsgraden. Processtemperaturen har mycket liten effekt på vätskeavgången.

För även du och jag består av stoft från en stjärna

Falk, A.

Contents

Appendix

Papers I-V

This thesis is based on the following papers, which are referred to in the text by their Roman numerals. Published papers are reproduced with the permission of the publishers.

I. Sundberg, C., Smårs, S. and Jönsson, H. 2004. Low pH as an inhibiting factor in the transition from mesophilic to thermophilic phase in composting. *Bioresource Technology* 95 (2), 145-150.

II. Eklind, Y., Sundberg C., Smårs S., Steger, K., Sundh I., Kirchmann H. and Jönsson H. 2005. Carbon turnover and ammonia emissions during composting of biowaste at different temperatures. Manuscript.

III. Sundberg, C. and Jönsson, H. 2005. Process inhibition due to organic acids in fed-batch composting of food waste – influence of starting culture. *Biodegradation* 16 (3), 205-213.

IV. Sundberg C. and Jönsson H. 2005. Initial phase of biowaste composting – aeration and pH. Manuscript.

V. Sundberg C. and Carlsson B. 2005. Modelling the initial phase of batch composting. Manuscript.

Notes on the authorship of the papers:

Paper I was planned by Smårs, Sundberg and Jönsson. The experiments were performed by Sundberg and Smårs. Sundberg performed the statistical analysis and the writing, with input from Jönsson.

Paper II was planned by all co-authors in cooperation. The experiments were mainly performed by Smårs. Sundberg and Eklind performed the data analysis and wrote the paper, with input from all co-authors. Sundberg's main focus was on carbon dioxide and ammonia emissions.

In Paper III, Jönsson and Sundberg planned the investigation. Sundberg performed the experiments, the analysis of the results and the writing, with input from Jönsson.

In Paper IV, Jönsson and Sundberg planned the investigation. The experiments and statistical analysis were performed by Sundberg. Sundberg wrote the manuscript, with input from Jönsson.

Paper V was planned, performed and written by Sundberg, with input from Carlsson.

Background

Organic waste is not only the result of human activity. Since the beginning of life on earth, organisms have produced waste and other organisms have used the energy and nutrients contained in that waste. Recycling of organic waste is thus an integrated function of any ecosystem. Composting is a process in which the naturally occurring ability of organisms to recycle organic waste is used for the benefit of humans in an accelerated degradation of organic waste. With increased understanding of how this process works, we can control it more accurately and make it more efficient according to our requirements.

In this work, the effort to understand and control the composting process sets off from the present status of waste management in Sweden and from previous composting research at SLU. Swedish municipal solid waste management is in a process of transition where landfilling of organic waste is no longer allowed (prohibited since January 2005). Several other waste treatment processes are increasing, especially incineration, composting and anaerobic digestion. Source separation of biodegradable waste has been introduced in many municipalities in Sweden and in several countries in Europe. In Scandinavia the growing season is short, so during a large part of the year there is very little garden waste available and the collected biodegradable waste consists only of food waste.

The largest problem facing the composting industry in Sweden, as well as many other countries, is odour. However, problems with odour are often due to deficient process management. Improved process management and control requires more accurate knowledge of the process. Since knowledge about the composting process is limited, a major research effort has been devoted to composting in recent years. A composting reactor built at SLU has been used for research on gaseous emissions and process dynamics (Beck-Friis, 2001; Smårs, 2002). A major finding of that work was that the start-up in batch composting could be inhibited by low pH in combination with temperatures above 40 °C (Smårs, 2002). Results from research in a pilot-scale composting reactor are not directly transferable to large-scale processes, and this issue was the starting point for the work reported in this thesis.

Objectives

The overall objective was to increase current knowledge of methods to improve the turnover in practical biowaste composting and this was achieved through theoretical and experimental investigations. Specific objectives were to improve the understanding of acid-related process inhibition and methods for abatement of such inhibition in relation to temperature (Paper I); in fed-batch composting (Paper III); and in large-scale batch composting (Paper IV). A further objective was to investigate the effect of composting temperature on carbon and nitrogen turnover (Paper II), and the final objective was to construct and test a mathematical model of the microbial kinetics of composting during an initial low-pH phase (Paper V).

Structure of the work

The main concept in this thesis work was to apply fundamental understanding of the composting process to problematic issues identified in composting practice. The work was based on previous research by the composting group at SLU and is part of its further development of knowledge on composting of source-separated household waste, especially regarding pH and temperature. The ways in which the papers presented in this thesis relate to earlier work and to each other are illustrated in Figure 1.

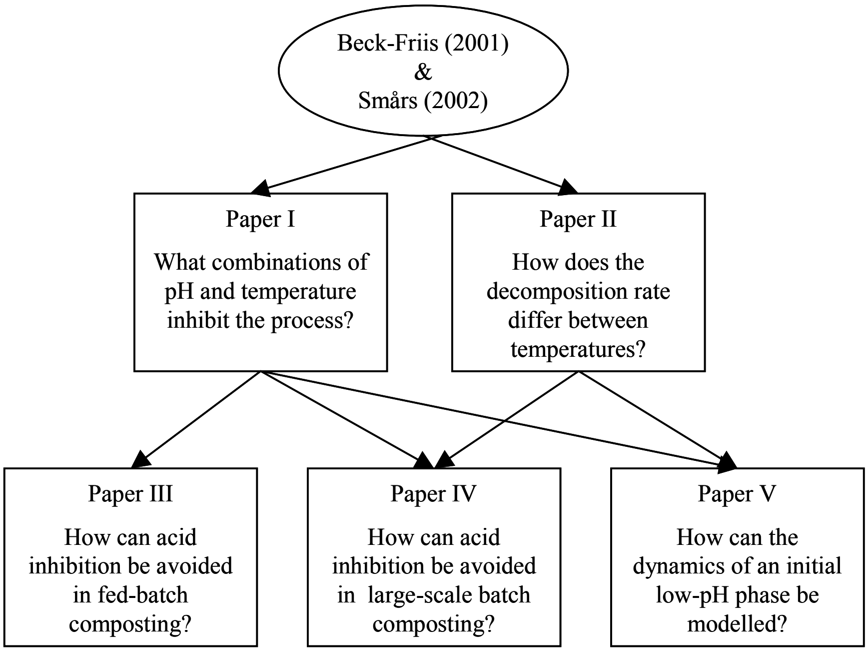

Figure 1. The main questions discussed in each paper and their relationships to earlier work and to each other.

Introduction

Why composting ?

Composting is a process that fulfils several waste management purposes - stabilisation, volume reduction and sanitation by thermal inactivation of pathogens. The aim of the stabilisation is to produce a material that does not putrefy, self-heat, deplete oxygen, produce odours or attract vermin. The compost product can be beneficial, supplying nutrients for plant growth, organic matter for soil improvement and agents for plant disease suppression.

In spite of the benefits of composting, there are several environmental issues to consider in composting. Ammonia emissions can be large, and ammonia contributes to eutrophication and acidification. The greenhouse gases methane and nitrous oxide are normally formed during composting, although in small amounts (Beck-Friis, 2001; Zeman, Depken & Rich, 2002). Moreover, malodorous substances, which can cause severe problems, are also formed during composting.

Composting is environmentally preferable to landfilling, but in Swedish environmental systems analyses of waste management based on substance flow analysis and life cycle assessment methods, composting has proved to be a less favourable option than incineration and anaerobic digestion for treatment of biodegradable household waste (Sundqvist *et al.*, 2002; Eriksson, 2003). The incineration systems in these studies produced both electricity and heat for district heating, and had advanced gas cleaning systems. The anaerobic digestion systems produced biogas that was used either for electricity and heat or as vehicle fuel, and the digestate was used as a fertiliser in agriculture. The composting systems were either open windrow systems or enclosed reactor systems. The compost was used in agriculture and was valued for replacing nitrogen and phosphorous fertiliser. Composting was judged to be less favourable because energy was not recovered, and because of larger emissions both during the process and from compost use. Contrary to this, a German environmental systems analysis concluded that for most environmental impact categories composting, especially home composting, was a better treatment method for household biowaste than incineration or anaerobic digestion (Vogt *et al.*, 2002). Important assumptions that differed from the Swedish studies were that only electricity, not heat, was utilised from incineration, that compost replaced peat as a soil amendment, and that peat was considered a fossil resource contributing to global warming. From an economic point of view, incineration is cost-effective if heat has an economic value, but not otherwise (Ayalon, Avnimelech & Shechter, 2000; Sonesson *et al.*, 2000). Anaerobic digestion is a more costly treatment method than composting, but its economic viability depends on the price of energy. Thus, it is not possible to draw any general conclusion on whether composting is environmentally or economically preferable to incineration or anaerobic digestion. The results depend not only on the waste management technology as such, but largely on local and regional external factors, especially the energy system.

There are also other reasons for composting to be an option for future waste management. For example, it is a technology that can be performed at any scale from household to regional. This makes local treatment possible, which reduces waste transport. Furthermore, the compost product is dry and can be sold on the soil product market, whereas the product of anaerobic digestion is normally a slurry which is only marketable to agriculture. Composting is also more flexible than anaerobic digestion and incineration in terms of size, time frame for planning and construction and pay-back time for investment.

In Sweden, waste incineration technology has been optimised for both process efficiency and pollution reduction as a result of consistent work within the industry, driven by demands from the public and government to reduce emissions. The Swedish composting industry has just started to work towards process efficiency and emission reduction, and there is a large potential for improvement. Consequently, a systems analysis that compares today’s best practice in composting and waste incineration does not compare the two treatment options at a similar potential, and thus gives little guidance regarding the best choices to make for the future.

Definitions

Temperature

Compost microorganisms are traditionally classified according to their temperature preferences. Microorganisms with optimum temperature for growth at 25-40 °C are called *mesophilic* and those with optimum temperature above 45 °C are called *thermophilic* (Madigan, Martinko & Parker, 2000). These terms can then be used to define temperature ranges, denoting temperatures below 40 °C mesophilic and above 45°C thermophilic. In this thesis the range 40-45 °C is called the *mesophilic-thermophilic transition* temperature range. The term *above mesophilic optimum* is used to denote temperatures of about 40°C and higher.

Batch and fed-batch

There are two principally different ways to perform composting; as a batch system or a fed system. In *batch* systems, a batch of waste is prepared and composted separately from other batches to a final product. In fed systems, fresh substrate is intermittently added and mixed with the active compost. Fed systems are often called fed-batch or continuous, but there is no consistent terminology within this field. In this thesis the term *fed-batch* is used to describe systems where substrate is added on a daily basis, while the word ‘continuous’ is reserved for systems with even shorter feeding intervals.

Process efficiency

In this thesis, compost *process efficiency* is defined as the rate of organic carbon turnover. The carbon dioxide emission or the mass loss of organic matter are two methods that are used to quantify the turnover or decomposition. *Decomposition* and *degradation* are two words that are used as synonyms to turnover.

Scales

Several terms are used to describe composting systems of different sizes. *Pilot-scale* refers to research reactors larger than 100 L and *laboratory-scale* to research reactors smaller than 100 L, as proposed by Mason & Milke (2005). *Full-scale* refers to systems used in composting practice, which are often more than 100 m^3, but may be as little as a few hundred litres, especially in fed-batch systems. *Large-scale* is used to describe systems of several m^3 with no upper size limit, and *small-scale* is used for systems of about 1 m^3 or less.

Introducing composting

Composting is commonly described as aerobic degradation of organic wastes where heat is released in the oxygen-consuming microbial metabolism, resulting in increased temperature. A composting system is dynamic, with very intense biological activity. This causes the system to change its own environmental conditions. Most notable is the increasing temperature. Equally important is the consumption of oxygen and production of carbon dioxide. In an active compost, the oxygen in the pore space is consumed within minutes, so a continuous supply of fresh air is crucial for the process to remain aerobic.

Composting is a microbial process, and the overall performance of the composting process is therefore the combined effect of the activity of individual microorganisms. It is thus important to understand and control the environmental factors that affect microbial life in composts. The most important parameters for the microorganisms are temperature, oxygen, moisture, pH and substrate composition (Miller, 1993).

All living organisms need water, so moisture is essential for the function of the composting process. For the microorganisms there is no upper limit for the water content as such, but excessive moisture reduces the air space in the compost matrix and thus causes oxygen limitation (Miller, 1993).

Based on their use of oxygen, microorganisms can be classified into three groups:

- Obligate aerobes, microorganisms that require oxygen for survival.
- Obligate anaerobes, microorganisms that cannot function in the presence of oxygen.
- Facultative anaerobes, microorganisms that have both aerobic and anaerobic metabolic pathways. Since aerobic metabolism renders more energy for the microorganisms, they grow faster when oxygen is present.

Heat is produced in aerobic decomposition of waste, which is a highly exothermic process. The heat produced can either remain in the compost mass, resulting in an increased temperature, or leave it either by conduction or radiation from the surface, or with the air passing through it.

The elevated temperature during composting is not only caused by the microorganisms, but also an important factor determining their activity. Each microbial species can only grow within a certain temperature range, and most microorganisms are killed by too high temperatures. Mesophilic microorganisms are active up to 40-45 °C, while thermophilic organisms have optimum temperatures above that. The temperature for maximum degradation rate in composting is normally near 55 °C, and the degradation rate is much lower at 70 °C (Miller, 1993).

The pH value changes during composting, due to changes in the chemical composition. In general, the pH falls below neutral in the beginning due to the formation of organic acids and later rises above neutral because the acids are consumed and because ammonium is produced (Beck-Friis *et al.*, 2003).

Substrate and *amendment* are two terms that are commonly used in composting. From a technical point of view, *substrate* normally refers to the wastes that are the primary objective of the operation, and an *amendment* is any compound that is added to improve the process, structurally, biologically or chemically. Most biological material can be composted. Depending on the composition of substrates and amendments, the composting process will be faster or slower, and composting will be easy or more problematic. Four characteristics of the substrate are of primary importance to the process: energy, nutrients, water and structure. Energy, nutrients and water are needed for microbial growth, and structure is essential for the aeration, which supplies oxygen and cools the compost.

Many composting systems have been used for treatment of different wastes under different conditions. Common systems at large scale include (i) naturally aerated windrow systems - long rows with a triangular cross section; (ii) forced aeration static pile systems; and (iii) tunnel systems – closed rotating cylinders, while smaller scale systems include static or rotating composts.

Most published research on composting has been on batch processes, probably because most large-scale operations are batch systems. However, some work on fed-batch composting was documented several decades ago (Schulze, 1962; Jeris & Regan, 1973) and in recent years, interest in continuous composting has increased, especially in Asia (Nakasaki, Akakura & Atsumi, 1998; Hwang, Shin & Tay, 2002; Haruta *et al.*, 2005). This is associated with an increased use of decentralised composting machines, mainly for food waste in households and restaurants, and the reported experiments have therefore been carried out on food waste, either real (Hwang, Shin & Tay, 2002) or artificial (dog food) (Nakasaki, Akakura & Atsumi, 1998).

In this thesis, the batch process is classified into three phases based on process dynamics and inspired by the works of Smårs (2002) and Hamelers (2001). These phases are (A) the initial phase; (B) the high-rate phase; and (C) the curing phase (Figure 2). In addition to particle size, temperature and moisture, which affect the degradation rate in all phases, there are certain parameters that are more important in each phase. Microbial biomass growth is limiting in the initial phase, often in combination with low pH. The initial phase is characterised by rising temperature and increasing CO_2 production. The duration of the initial phase can range from less than a day to several weeks. The next phase is the high-rate phase, when neither microbial biomass, substrate availability nor low pH is rate-limiting. This phase is normally thermophilic, and one rate-limiting factor is oxygen diffusion into the solid-liquid compost particles. The duration of the high-rate phase ranges from a few days to many weeks. In the curing phase, the readily available substrate is depleted while the microbial biomass is still large, so the substrate availability is rate-limiting. The CO_2 production declines and the temperature may also decline.

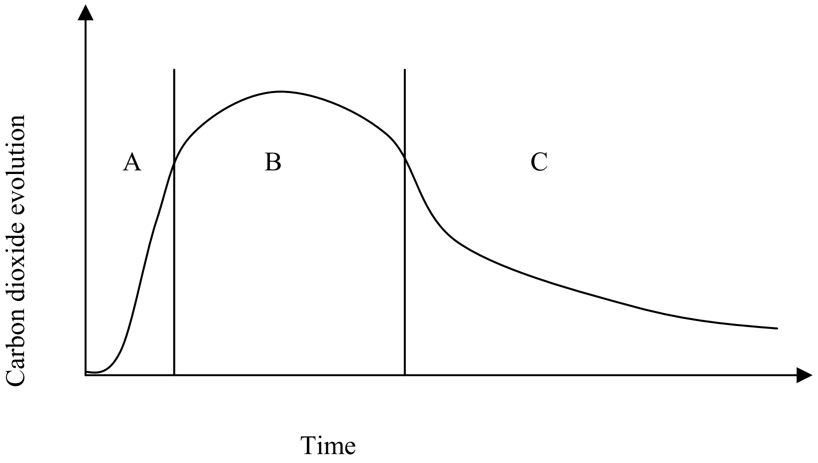

Figure 2. Schematic representation of the different phases of the composting process. A – initial phase, B – high-rate phase, C – curing phase. This is not a figure of a real process and the axes are not to scale.

Compost models

Increased computational power has made it feasible to use mathematical models of the composting process, which can improve understanding and reduce the need for costly experimentation. More than 30 papers have been published on compost models since the first publication on the subject by Finger, Hatch & Regan in 1976 (Mason). Coupled heat and mass balances solved over time, and sometimes space, form the basis for most models. Most models are deterministic, exceptions are those by Seki (2000), Liang, Das & McClendon (2003) and Scholwin & Bidlingmaier (2003). The composts are modelled either as lumped masses, in horizontal layers or as aggregations of three-dimensional elements. Most models have a macroscale perspective, but some have a microbial point of view (Hamelers, 1993; Kaiser, 1996; Stombaugh & Nokes, 1996; Hamelers, 2001; Seki, 2002; Liang *et al.*, 2004) and the model presented by Hamelers (1993; 2001) also considers the compost particle scale dynamics. Between one (Stombaugh & Nokes, 1996) and four (Kaiser, 1996) substrate components and groups of aerobic microorganisms have been included in the models. The Hamelers model (1993, 2001) includes aerobic biomass as well as anaerobic hydrolysis of macromolecules.

Several state variables are predicted in the different models. Most models predict temperature and substrate decomposition, either as substrate mass loss, oxygen uptake or carbon dioxide evolution (Mason). In microbial models, the microbial biomass is also predicted. There are also some models that predict moisture content (Nakasaki *et al.*, 1987; Seki, 2002) and oxygen concentration (Finger, Hatch & Regan, 1976; Mohee, White & Das, 1998; Seki, 2002). One published model describes the nitrogen dynamics in composting and predicts ammonia emissions (Liang *et al.*, 2004). Most models simulate forced aeration systems with fixed aeration rates (Mason). One model predicts the airflow in a passive aeration system (Veeken *et al.*, 2003).

In several models, some of the parameters have been fitted to experimental data, (e.g.VanderGheynst, Walker & Parlange, 1997; Seki, 2002). There are only a few examples where independent experimental data have been used for model validation (Liang, Das & McClendon, 2003). In a review, Mason (Mason) states that no models have shown acceptable predictive ability, either in predictions of temperature, or of degradation rate measured as O_2 consumption or CO_2 evolution. Despite this, several models were judged to have a good predictive ability of substrate degradation measured as loss of BVS or solids.

Facing low pH – the acidic phase

The pH in composts is influenced by three acid-base systems. One is the carbonic system, with carbon dioxide (CO_2), which is formed during decomposition and can escape as a gas or dissolve in the liquid, forming carbonic acid (H_2CO_3), bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}). This system has two dissociation constants (pK_a), 6.35 and 10.33 at 25 °C, and thus it tends to neutralise the pH of the compost, increasing low pH and reducing high pH. The second system is ammonium (NH_4^+) – ammonia (NH_3), which is formed when protein is decomposed. During the initial phase of composting, most of the metabolised nitrogen is retained by growing microorganisms, but during the high-rate phase ammonia is released. The ammonia system has a pK_a of 9.24 at 25 °C and thus increases the pH towards this value. The third system is composed of several organic acids, of which acetic and lactic acid dominate. This system can reduce pH down to 4.14, which is the pK_a of lactic acid at 25 °C (Weast *et al.*, 1989-1990), and it is important in the beginning of composting as is described below. These three systems combine to form the typical pH curve for batch composting, with initially falling pH and a sharp rise during high-rate degradation (Figure 3).

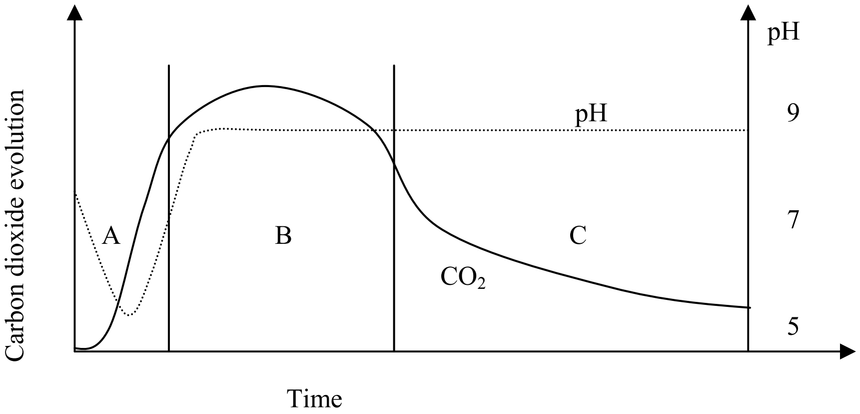

Figure 3. Representation of a typical pH curve in a batch composting process. The pH falls initially, and the start of the high-rate phase coincides with a rise in pH. A – initial phase, B – high-rate phase, C – curing phase.

Organic acids are formed during fermentation of organic matter. There are several metabolic pathways for acid production. One is fermentation by anaerobic microorganisms. Acetic acid is the main product of such processes, but longer-chained acids such as butyric or propionic are also formed. Another important organic acid found in composts is lactic acid, which is formed by lactic acid bacteria, a group of facultative anaerobes (Brock & Madigan, 1988). Acetic acid is mainly produced anaerobically, but it can also be produced when oxygen is present, *e.g.* when *E. coli* is subjected to high concentrations of glucose (Enfors & Häggström, 2000). The bacteria take up more glucose than they can oxidise aerobically and acetate is formed in what is called the overflow metabolic pathway.

Organic acids are not only formed in composts, they are also decomposed. Most microorganisms can utilise organic acids as a readily available substrate for aerobic oxidation. The acid concentration in composts is therefore influenced by both production and consumption of the acids.

Organic acids are suppressive to microbial activity and growth at low pH (Cherrington *et al.*, 1991). Different microorganisms have different sensitivity to organic acids. Generally, bacteria are more sensitive to acids than fungi (Atlas & Bartha, 1998). The acids interfere with cellular functions of the organisms. The acids can enter the cell when they are in their undissociated form, *i.e.* when the pH is low. The negative effect of organic acids on microbial activity is thus strongly dependent on the pH of the medium. The acids themselves reduce the pH, but in composts the pH is also influenced by the carbonic and ammonia systems. Acid concentration and pH are therefore connected and both influence the toxic effect of organic acids.

In municipal solid waste, acetic and lactic acids are produced during storage, and in source-separated organic waste these acids can reduce the pH to 4-5 (Eklind *et al.*, 1997). During the initial phase of batch composting, reduced pH and high concentrations of organic acids can occur (Beck-Friis *et al.*, 2003). During successful composting, the acids are decomposed and the pH increases. Compared to well-aerated composting, composting at low oxygen levels results in a larger acid production and a slower break-down of acids (Beck-Friis *et al.*, 2003). The initial period of low pH can be significantly reduced if the temperature in the compost stays below 40 °C until the pH rises (Smårs *et al.*, 2002).

The heat machine

During aerobic decomposition of organic substances, the chemical energy in the material is partly released as heat and partly used for the construction of new substances within the organisms consuming the organic material. A large proportion is released as heat, which can be quantified either calorimetrically or through indirect methods. Weppen (2001) calorimetrically measured a heat release of 15-19 kJ g^{-1} decomposed matter from composting organic waste amended with wood chips or straw. In experiments spiked with fat, this value increased to 22-25 kJ g^{-1}, and the average from all experiments was 20.7±2.9 kJ g^{-1} or 452±29 kJ mol^{-1} oxygen consumed. The energy release can be expressed in relation to the consumption of either substrate or oxygen and the latter shows less variation between different substrates (Haug, 1993). Indirect methods to estimate the heat released during aerobic decomposition use information on substrate composition or oxygen consumption. If the chemical composition of the substance is known, the heat release can be calculated from the proportions of carbohydrates, lipids and proteins. The heat release can also be calculated from the oxygen consumption. The heat of combustion per electron transferred to a methane-type bond is relatively constant, giving 110 kJ mol^{-1} e^{-} (Weppen, 2001).

The heat produced during decomposition can either remain in the compost mass or leave it. Both mass and energy are conserved, so balances can be set up. Several models of the physical processes in composts have been presented. The major difference between them is that some models are lumped (Bach *et al.*, 1987; Haug, 1993), whereas others include spatial variations (Stombaugh & Nokes, 1996; Bari & Koenig, 2000; Seki, 2002).

The energy for heating of the composting mass (Q_s) is calculated as:

$$Q_s = \Delta T\left(c_{pw} \cdot m_w + c_{ps} \cdot m_s\right) \qquad (1)$$

where ΔT is the temperature change, m_w and m_s are the masses of water and solids, and c_{pw} and c_{ps} their respective specific heat capacities (Bach *et al.*, 1987). The heat capacity of water (c_{pw} = 4.2 J g^{-1} °C^{-1}) is well known, whereas c_{ps} depends on the solids composition, but it is normally in the range 0.5-2.0 J g^{-1} °C^{-1}. In all calculations in this report, a literature value of 1.2 J g^{-1} °C^{-1} is used (Bach *et al.*, 1987).

If all the energy released during decomposition were to remain in the compost, the temperature would rapidly rise above the maximum temperature of microbial survival. The 15-20 kJ that are released from the decomposition of 1 g of organic matter are enough to heat 35-50 g of water by 100°C. It is thus obvious that most of the heat produced during degradation must leave the compost mass in some way, either by conduction or radiation from the surface or with the air passing through it. The conductive surface losses (Q_l) are calculated as:

$$Q_l = U \cdot A \cdot \left(T - T_o\right) \cdot \Delta t \qquad (2)$$

where U is the heat transfer coefficient, A is the surface area, T is the system temperature and T_o is the ambient temperature and t is time (Bach *et al.*, 1987). Surface cooling by conduction is important in small-scale composting (Weppen, 2001). In small composts (up to a few cubic metres), insulation is therefore often needed to obtain the elevated temperatures required for sanitation. Larger composts have a larger volume-to-surface ratio, so surface cooling is smaller. The effect of surface cooling is further limited by the low thermal conductivity of composts (Haug, 1993), which implies that large temperature differences can be maintained over short distances (Beck-Friis *et al.*, 2000). When surface cooling is limited, the major heat transfer process is by airflow. Both sensible heating of dry air and latent heat from water evaporation account for this heat transport. The water carrying capacity of air increases exponentially with temperature and therefore the evaporative cooling capacity per volume of air is larger at higher temperatures. The heat removal by heating of dry air (Q_a) is calculated as:

$$Q_a = c_{pa} \cdot m_a \cdot (T_{out} - T_{in}) \qquad (3)$$

Where c_{pa} is the specific heat of dry air, m_a is the mass of the air, and T_{in} and T_{out} are the input and output air temperatures. Changes in mass and specific heat capacity of the dry air occur due to the consumption of oxygen and production of carbon dioxide. These small changes are neglected in Equation 3.

Evaporation of water removes heat (Q_w) from the compost according to:

$$Q_w = c_{pe} \cdot m_{e,in} \cdot (T_{out} - T_{in}) + \Delta H_{vap} \cdot (m_{e,out} - m_{e,in}) \qquad (4)$$

where c_{pe} is the specific heat of steam, $m_{e,in}$ and $m_{e,out}$ represent the mass of steam in the incoming and outgoing air, and ΔH_{vap} is the enthalpy of evaporation at T_{out}.

The thermal characteristics of laboratory and pilot-scale compost reactors are normally quite different from those of large-scale facilities, which is explained by the difference in surface-to-volume ratio between laboratory and large-scale composts. Weppen (2001) and Bach *et al.* (1987) reported that 60% of the heat was conducted from their reactor surfaces even though they were well insulated, while it was found that in large-scale reactors, 4-11% of the heat dissipated through surface and walls, while the rest was removed with the air. In laboratory reactor studies, the surface losses can be minimised by keeping the reactors in a closed chamber where the conductive heat flow can be controlled (Hogan, Miller & Finstein, 1989).

The dilemma of air and water

Aerobic microorganisms require both oxygen and water. Since water is a liquid and oxygen is gaseous, there are obvious problems with achieving a balance between good moisture and oxygen conditions in compost. The relationship between solids, liquids and gas in the composting matrix is important for this balance. A useful conceptual model of the compost matrix involves aggregates of solid and liquid that are surrounded by a connected gas-filled pore space (Figure 4). Since oxygen is consumed and carbon dioxide is produced, the gas needs to be continuously replenished. In the gas-filled pores in composts, the oxygen transport is dominated by mass-flow, whereas in the liquid films on the compost aggregates the oxygen is transported by diffusion. Diffusional transport is slow and it is the limiting process in the oxygen supply of well-aerated composts (Hamelers, 2001). Since microorganisms consume much oxygen, only a thin outer layer of each compost particle is aerobic during the active phases. At low concentrations of oxygen, a few per cent or less in the composting gas, the process rate is highly influenced by changes in the oxygen concentrations, but this influence is less pronounced at higher oxygen concentrations. For example, during the first two weeks of composting in a reactor, a larger difference has been found between the process rate at 1% and 2.5% O_2 than between 2.5% and 16% O_2 (Beck-Friis *et al.*, 2003).

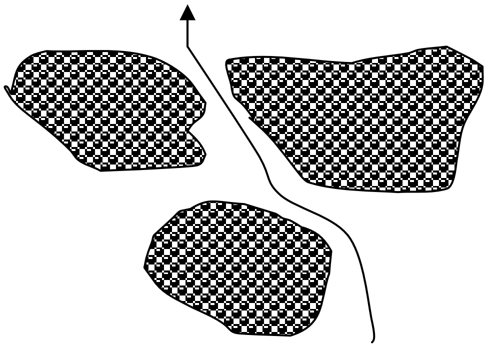

Figure 4. Schematic representation of compost aggregates. The gas is transported by mass flow (arrows). The solid components (substrate, microbial biomass and inert matter) form a matrix in which the pore-space is water-filled. The water in the outer part of the particles contains dissolved oxygen, whereas the inner part is anaerobic during active composting. Adapted from Hamelers (2001).

The effect of oxygen on the decomposition rate is normally modelled by a Monod function:

$$f(O_2) = \frac{O_2}{K_s + O_2} \quad (5)$$

where O_2 is the oxygen concentration in the gas phase and K_s is the half-saturation coefficient (Haug, 1993; Richard & Walker, 1999). However, microorganisms live mainly in the liquid phase on and in the compost particles, and the oxygen concentration there is not only determined by the gas phase concentration. It also depends on the oxygen consumption rate, the diffusion rate and particle size. The spatial gradient of oxygen within the compost particle has been taken into account in one model, which was shown to describe some oxygen-related phenomena better than simpler models (Hamelers, 2001).

The balance between water and oxygen is further complicated by the fact that the gas removes water from the compost, as water evaporates. Evaporation is the major means of cooling, and it is not possible to maintain the same moisture content over long periods of time. The relative humidity in exhaust gases from an active compost is always near 100% due to the high water content and large surface area in the compost matrix. The relative humidity of the gas passing through compost increases as the gas is heated and water evaporates from the compost. In large composts evaporation is a major mechanism in compost cooling. The 20 kJ produced by 1 g of decomposed matter is equivalent to the heat of vaporisation of 10 g of water. Therefore the evaporation due to heat has the capacity to dry the compost significantly, even though about 0.5 g of water is produced when 1 g of organic matter is decomposed. The aeration needed for cooling the compost is usually larger than the aeration needed for oxygen supply (Haug, 1993).

Drying of the substrate is often a key objective of composting (Nakasaki *et al.*, 1987; Choi, Richard & Ahn, 2001). In household waste composting, the waste normally has a solids content of 30-60% and the concentration of readily available substrate is high. This makes the compost susceptible to high temperatures and excessive drying. Therefore, water addition can often speed up the composting process (Walker *et al.*, 1999).

From organic to mineral

During high-rate decomposition, organic nitrogen in the substrate is microbially transformed into microbial biomass, and also mineralised to ammonium (NH_4^+). The mineralisation of nitrogen peaks shortly after the peak in carbon dioxide emissions (Nakasaki *et al.*, 1998). A major pathway for ammonium is release to the compost gas as ammonia (NH_3). This release is controlled by two chemical equilibria:

$$NH_4^+ \Leftrightarrow H^+ + NH_3(aq) \qquad (6)$$

$$NH_3(aq) \Leftrightarrow NH_3(g) \qquad (7)$$

Firstly, the pH-dependent equilibrium ammonium (NH_4^+) – ammonia (NH_3) ensures that practically all ammonia is present as ammonium ions at low pH. Therefore there are virtually no ammonia emissions when the pH in compost is below 7 (Beck-Friis *et al*., 2003). Secondly, the equilibrium between aqueous and gaseous ammonia describes the solubility of ammonia in water, which decreases with increasing temperature. When ammonia is removed from the gas phase by aeration of compost the equilibrium is driven to the right, and thus the ammonia emissions from compost increase at higher aeration rates.

The C:N ratio is relevant for the ammonia losses from composting processes, with higher losses at lower C:N ratios (Kirchmann & Witter, 1989). However, the availability of the substrate carbon is also important, as higher carbon availability gives lower nitrogen losses (Barrington *et al.*, 2002).

The nitrogen mass balance in composting is dominated by organic nitrogen and NH_3-NH_4^+, especially during the most intensive decomposition. However, there are also other important nitrogen transformations. Nitrification (oxidation of NH_4^+ to NO_3^-) and denitrification (reduction of NO_3^- to N_2) are processes carried out in several steps by various microbial groups in compost (Körner & Stegmann, 2002). Nitrous oxide (N_2O) is an intermediate in both nitrification and denitrification, which is of interest because it is a very potent greenhouse gas. Nitrous oxide has been identified in compost gases, but the mechanisms behind its formation are not fully known (Beck-Friis, 2001).

Summary of papers

Low pH as an inhibiting factor in the transition from mesophilic to thermophilic phase in composting (Paper I)

In this study, the effects of temperature (36-46 °C) and pH (4.6-9.2) on the respiration rate during the early phase of composting were investigated. Respiration experiments were carried out with source-separated organic household waste from two different sources. The wastes had been composted for 1-6 days, reached a temperature of 35-37 °C and a pH of 4.6-5.4. Before the respiration measurements, the samples were treated with sodium hydroxide (7 M) to adjust the pH to values ranging from 4.6 to 9.2. The compost samples (3.0 g) were placed in 118 mL airtight bottles. The bottles were placed in water baths that were held at 36±1 °C and 46±1 °C, respectively. The compost gas was analysed for CO_2 by absorption in 7 M sodium hydroxide.

At 36 °C, the CO_2 production of the compost samples was high irrespective of the pH (Figure 5). At 46 °C, however, the degradation rate at the naturally occurring pH of about 5 was very low. At this temperature the activity was substantially higher in the samples where the pH had been raised above 6.5 with sodium hydroxide.

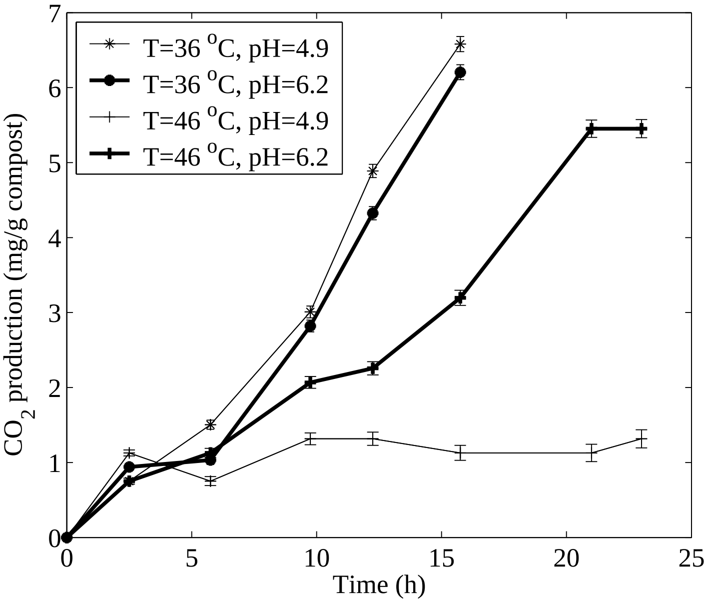

Figure 5. Time series showing the cumulative carbon dioxide production in compost samples kept at 36 °C (*) and 46 °C (+). The thick lines indicate samples that had been treated with NaOH to raise the pH value.

The pH was recorded before and after the respiration experiments. In compost samples with initial pH < 6.5 that were treated at 36 °C, the pH increased during the experiment, whereas the pH decreased in samples at 46 °C (Figure 6).

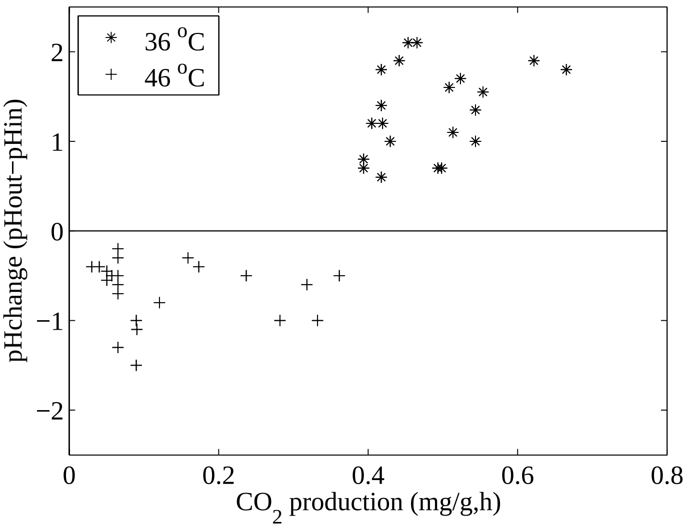

Figure 6. Change in pH versus total respiration during the experiment for all samples with an initial pH below 6.5.

Carbon turnover and ammonia emissions during composting of biowaste at different temperatures (Paper II)

Biowaste was composted at 40 °C, 55 °C and 67 °C in a 200-litre compost reactor. The substrate had an initial C:N ratio of 22 and was a mixture of wheat straw and source-separated organic household waste, mainly food waste. A start-up strategy was used that aimed at keeping the temperature below 37 °C during the first few days while the pH was still low. When the pH increased, the set-point temperature was changed for each experiment to either 40 °C (one run), 55 °C (three runs) or 67 °C (three runs). In two runs, the temperature was gradually lowered after 9 days. To keep the set process temperature, part of the compost gas was circulated through a cooler in which a condensate was formed. To keep the oxygen concentration at 16%, a regulator-controlled proportion of the cooled gas was replaced with fresh air.

The decomposition was fastest at 55 °C. At 67 °C the decomposition tended to be faster than at 40°C, but the difference was not significant (Figure 7). The ammonia emissions were considerably higher at 67 °C than at the lower temperatures (Figure 8). The difference in ammonia losses was also reflected in the nitrogen remaining in the compost, with less nitrogen and a higher C:N ratio after composting at 67 °C. When the temperature was reduced gradually from 67 °C to 40 °C (Run 67c), the ammonia emissions diminished considerably compared to continued treatment at 67 °C (Run 67a, Figure 8).

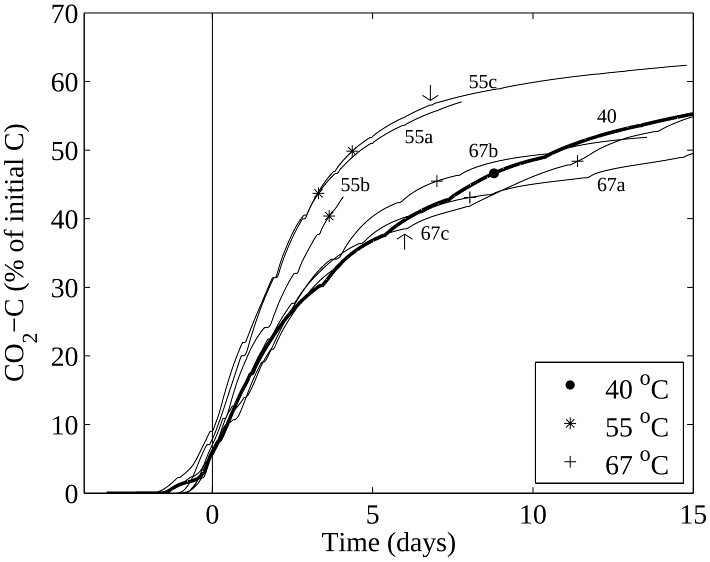

Figure 7. Cumulative carbon dioxide emissions in all runs. Arrows indicate start of successive decrease in temperature set-point in Run55c and 67c. Time=0 when the set-point temperature was changed from 37 °C to 40, 55 or 67 °C.

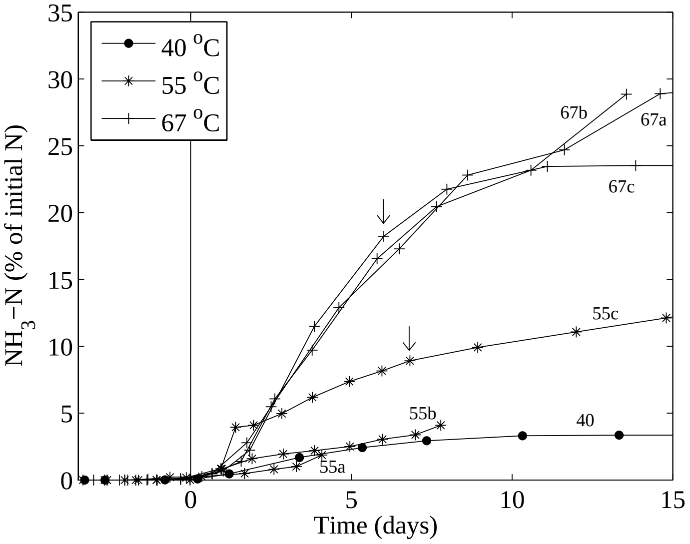

Figure 8. Cumulative ammonia emissions in all runs. Arrows indicate start of successive decrease in temperature set point in Run55c and 67c. Time=0 when the set-point temperature was changed from 37 °C to 40, 55 or 67 °C.

Process inhibition due to organic acids in fed-batch composting of food waste – influence of starting culture (Paper III)

The effects of starting culture on fed-batch composting of food waste was examined in laboratory reactors. An experiment with four treatments, two of which were duplicated, was carried out in laboratory reactors (Figure 9). Food waste (50 g), sawdust (5 g) and water (10-50 g) were added daily to reactors with different amounts of starting culture (20-400 g). Three identical cylindrical rotating reactors, made of 3-litre insulated Dewar vessels, were used for the experiments (Figure 9).

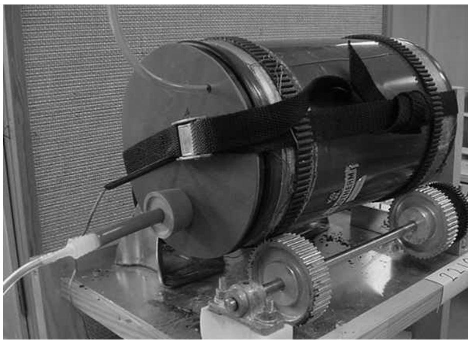

Figure 9. One of the 3-litre rotating reactors used for fed-batch composting.

The temperature in the reactors was monitored continuously with thermocouples. The reactors were aerated with 0.7 L min^{-1} with air pumps by negative aeration. The exhaust gas was bubbled through a sodium hydroxide solution to absorb the carbon dioxide produced. The absorbed carbon dioxide was later determined by titration with hydrochloric acid. Solid samples were taken daily and the solids content, ash, pH and concentration of organic acids were determined.

The process performance in the reactors with 200 and 400 g of starting culture was distinctly different from that in those with 20 and 100 g. In the reactors with more starting culture, pH declined during the first few days and then increased above 7 (Figure 10). The CO_2 production increased and after three days the temperature increased to steady thermophilic levels (Figure 10). In the reactors with less starting culture, the temperature did not exceed 42 °C, the CO_2 production remained low and the pH remained below 6 (Figure 11).

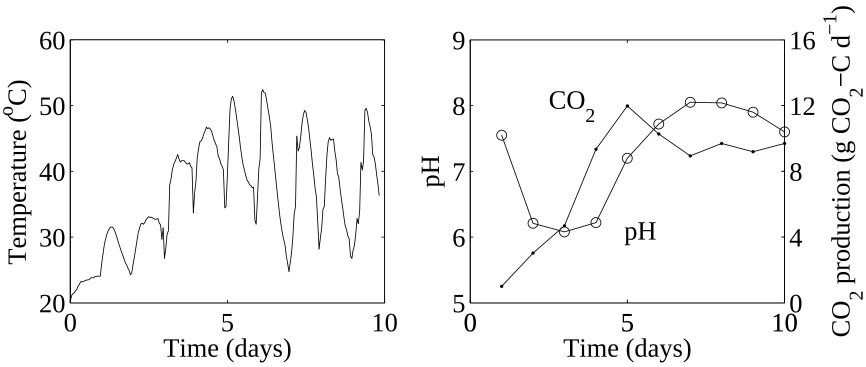

Figure 10. Temperature (left), CO_2 and pH (right) in a reactor with a successful process, with 200 g of starting culture.

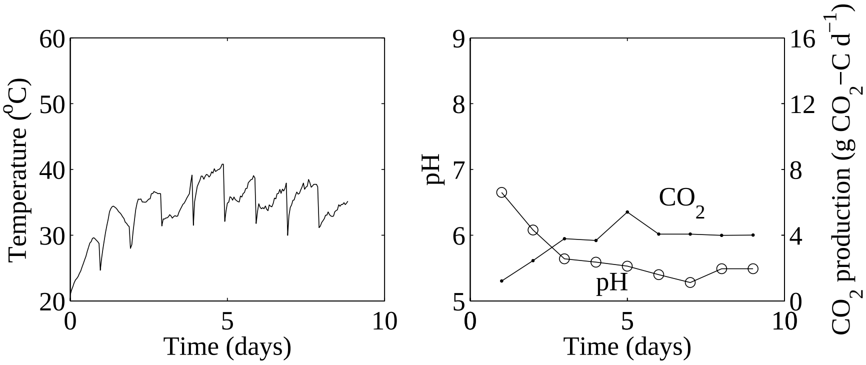

Figure 11. Temperature (left), CO_2 and pH (right) in a reactor with a failing process, with 100 g of starting culture.

Initial phase of biowaste composting – aeration and pH (Paper IV)

Biowaste composting at several plants in Scandinavia has been troubled by low pH during the process. The hypothesis that increased aeration during the beginning of the process would improve the process was tested in full-scale experiments at two large composting plants that have forced aeration indoor processes. Parallel runs with different aeration rates were compared. Temperature, CO_2 and O_2 concentration were measured during the processes, and material samples were analysed for pH, dry matter and volatile solids.

At both composting plants, increased aeration rates at the start of the process resulted in higher microbial activity, rising pH (Figure 12), and a more stable compost product. The O_2 concentrations were high (>15%) even at low aeration rates, so the consistently low pH was not due to an anaerobic process environment.

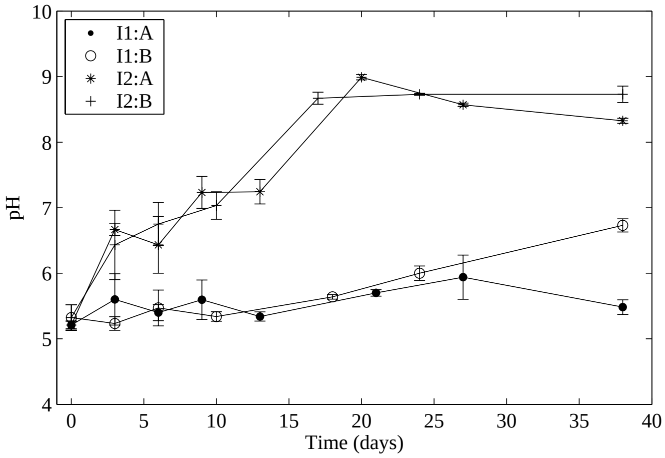

Figure 12. The pH profile in the different batches during the composting process at one of the composting plants. The standard error of the mean is shown in error bars.

At one plant the temperatures were very similar during the early process when pH was low (Table 1) even though the aeration rates differed. This is an indication of the same inhibition at low pH and temperature above the mesophilic optimum as was investigated in Paper I.

Table 1. *The number of readings in different temperature intervals at the RKR composting plant, measured in the centre of the bays 0.3 m from the upper surface. Data from when pH in the condensate was below and above 7, respectively*

Temperature	pH < 7	pH > 7
< 48 °C	4	4
48-56 °C	26	3
> 56 °C	2	5

As the O_2 and CO_2 concentrations were also similar during the early processes regardless of aeration rate, neither temperature, CO_2 nor O_2 concentration gave any indication of the decomposition rate. Increased aeration caused severe drying of the compost, but with a total water addition of 0.7 times the wet weight of the substrate (with 40% dry matter), drying was avoided at one plant.

Modelling the initial phase of batch composting (Paper V)

A model was developed that describes the dynamic development of temperature and microbial activity during the initial phase of composting under changing pH-conditions. The model was calibrated and validated with data from experiments in the same research reactor as used in Paper II. Three differential equations describe the kinetics of the modelled system, two for the microbial groups and one for the substrate:

$$\frac{dX_m}{dt} = \mu_m \cdot X_m - \lambda \cdot X_m \qquad (8)$$

$$\frac{dX_t}{dt} = \mu_t \cdot X_t \qquad (9)$$

$$\frac{dS}{dt} = -\frac{\mu_m}{Y} \cdot X_m - \frac{\mu_t}{Y} \cdot X_t - g \cdot (X_m + X_t) + \lambda \cdot X_m \qquad (10)$$

where X_m and X_t are the mesophilic and thermophilic biomass, μ is the growth rate, λ is the decay rate, S is the substrate, Y is the yield, g is the maintenance coefficient. The growth rate depends on temperature (T), oxygen availability and pH. The growth rate of the thermophiles is reduced at low pH.

$$\mu = \mu_{max} \cdot f_1(T) \cdot f_2(pH) \cdot f_3(O_2) \qquad (11)$$

The consumed substrate that is not used for biomass growth is released as CO_2:

$$CO_2 = -\frac{dS}{dt} - \frac{dX_m}{dt} - \frac{dX_t}{dt} \qquad (12)$$

This kinetic submodel is combined with a physical submodel that calculates the temperature as a function of CO_2 emission, and a submodel for pH and acid concentration (Figure 13). The combined model simulates time series of decomposition, temperature and pH, starting from a defined initial state of the system. The submodels can be used separately, but they then need time series of temperature, pH or carbon dioxide as input.

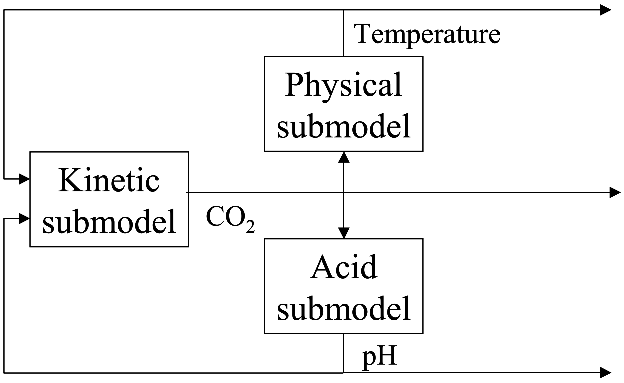

Figure 13. A conceptual representation of the model. The kinetic submodel calculates CO_2 emission, which is used to calculate pH in the pH submodel and temperature in the physical submodel. pH and temperature values are fed into the kinetic model for calculation of CO_2 emissions.

The calibrated model predicted the overall behaviour of the system, with oscillations in temperature and CO_2 emission during the acidic phase, followed by increased activity when pH rises. Data from a separate calibration of the kinetic submodel are shown in Figure 14.

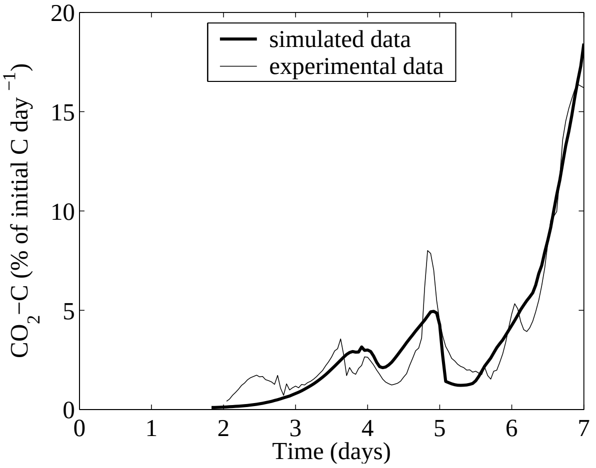

Figure 14. Kinetic submodel run with values from calibration.

Discussion

Acid inhibition

Organic acids that are formed microbially during decomposition of organic matter can severely inhibit the composting process. Investigations of low-pH conditions were performed with food waste composting, both in fed-batch processes (Paper III) and in the early phase of batch processes (Paper I). In batch processes, it was found that by keeping the temperature below the mesophilic optimum temperature, the acids disappeared and the pH increased faster than when the temperature was allowed to rise to thermophilic levels (Paper I). At thermophilic temperatures, the activity increased when the pH was raised artificially above 6.5. Although the acid concentration was the same, the acids were not inhibitory to the microorganisms. This is because it is the undissociated form of the organic acids, which are present at low pH, that can pass through the cell walls of microorganisms and cause damage (Cherrington *et al.*, 1991).

The inhibition of the process when low pH is combined with temperature above the mesophilic optimum is a likely explanation for the lag in the transition from mesophilic and thermophilic temperatures (Figure 15) that was observed by Sundberg & Jönsson (2003) and that has been noticed by many others when composting food waste or other acidic wastes (Haug, 1993; Day *et al.*, 1998; Schloss & Walker, 2000; Beck-Friis *et al.*, 2001; Weppen, 2001; Reinhardt, 2002). This transitional phase has normally been attributed to slow growth of thermophiles, but the synchronous rise in pH and temperature that was noted in Paper III and IV, as well as by Beck-Friis *et al.* (2001) and Reinhardt (2002), is not likely to be a coincidence. The transitional phase occurs because the mesophiles are inactivated by the high temperature, and the thermophiles are suppressed by the low pH and organic acids. When the pH rises, the thermophiles are no longer inhibited, their activity increases and the temperature rises further. This connection between the often observed lag and the low pH was noted by Reinhardt (2002) and is further supported by the experimental results in Papers I and IV.

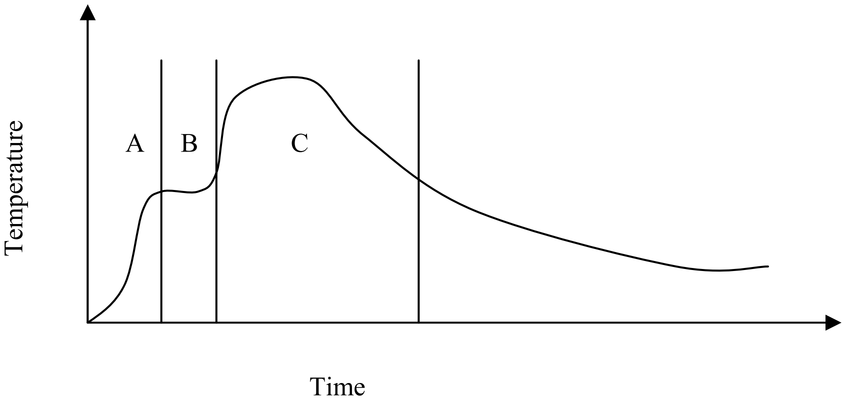

Figure 15. Schematic representation of the temperature development in a self-heating batch composting process. There is often a lag (B) in the transition from mesophilic (A) to thermophilic (C) temperature.

Adding lime or other alkaline substances to raise the pH is one method to alleviate the inhibition caused by acids in compost and it functions well (Nakasaki *et al.*, 1993, Paper I). However, it is not a perfect solution to acids in compost. First, liming involves extra costs, both for purchase and for the work of adding the lime to the substrate. Second, lime addition can be technically demanding, especially from a work environment perspective (Toren *et al.*, 1996). Third, liming increases the ammonia emissions, thus increasing the environmental effects of ammonia release and reducing the nutrient status of the product (Nakasaki *et al.*, 1993). As a consequence of the economic, technical and environmental drawbacks of liming, there is reason to search for other ways of overcoming low pH and acid inhibition. One such way is cooling to keep the temperature below 40 °C (Smårs *et al.*, 2002, Paper I). In large-scale composting it is difficult to keep the temperature as low as 40 °C, but by increasing the aeration rate it is possible to increase the decomposition rate and thereby shorten the period of low pH (Paper IV).

In the fed-batch process (Paper III), it was found that addition of a starting culture of active compost caused a quicker start of the aerobic degradation and prevented continued conditions of low pH and high organic acid concentrations. A starting culture has several stabilising effects on the composting process. First, it can provide a structure that supports the oxygen supply to the process. Second, it dilutes the fresh waste. This reduces the risk of local oxygen depletion caused by rapid degradation of easily degradable matter. Oxygen supply is important since the concentration of organic acids in composts becomes higher at low oxygen concentrations (Beck-Friis *et al.*, 2003), probably because of acid formation in anaerobic microenvironments. Third, a starting culture supplies microorganisms, which can decompose the acids that are formed. Fourth, the starting culture provides a chemical pH-buffer, which may reduce the inhibitory effect of the organic acids by reducing the depression of the pH. It is difficult to distinguish between these different effects, and no such attempt was made in our experiments. However, results from experiments with both microbially active and sterilised structural amendments show that the microorganisms in the starting culture can accelerate the start-up of the composting process when there is not an adequate microbial community present in the substrate (Nakasaki & Akiyama, 1988; Nakasaki *et al.*, 1992).

Modelling pH and acid concentration

From experiments in the same experimental reactor as used in Paper II, Smårs (2002) concluded that the low pH phase lasted until the first 10% of the organic carbon had decomposed. This took 2-13 days depending on temperature and O_2 concentration. In the experiments in Paper II the low pH phase consisted of the first 6.7 ± 1.3% of CO_2 emission. A similar pattern was also seen in the experiments at one large-scale plant (Paper IV), where the pH shift occurred at different times in different processes, but the cumulative decomposition when the pH increased was very similar (Figure 16). It was not possible to get an absolute value of the cumulative decomposition in that experiment due to a deficient mass balance.

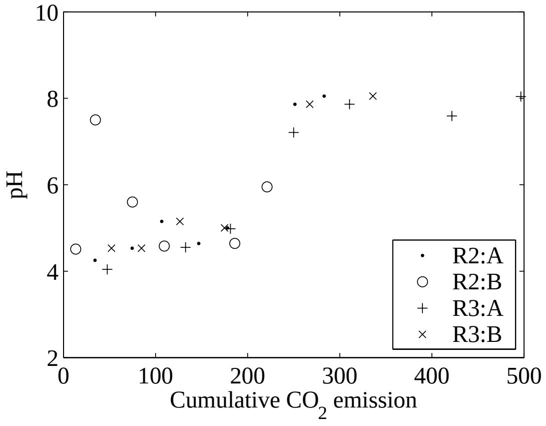

Figure 16. pH in condensate as a function of the cumulative CO_2 emissions in four batches at a large-scale composting plant (Paper IV).

The empirical observation that the pH shift occurs at the same cumulative decomposition for a certain substrate under various composting conditions (hereafter referred to as the 'cumulative model') does not provide an explanation of the process mechanisms. For that, we need to consider that the acid concentration is determined by both acid production and acid consumption. These two processes occur simultaneously and no attempt to isolate them was made in any of the experiments described in Papers I-V. Nevertheless, some assumptions were necessary when modelling the processes (Paper V). An acid production that was constant over time was assumed:

acid production rate = k_p k_p constant (13)

Based on the observed correlation between carbon decomposition and acid disappearance, acid decomposition was modelled as a rate proportional to CO_2 emission:

acid consumption rate = $k_d CO_2(t)$ k_d constant (14)

The pH value depends on the concentrations and equilibrium constants (pK_a) of the different organic acids involved, the carbonate and the ammonia systems, as well as the surface chemistry of the compost particles. Instead of making a complex model of the acid-base chemistry, a simple empirical model was used.

pH = k_c · acid conc. + pH (0) pH > 4.3 k_c constant (15)

Results from experiments with high acid concentrations show that a linear assumption is valid down to a pH of 4.3-4.6, and then the pH remains constant (Leonardsson & Östensson, 2005). This simple production-consumption model (Equations 13-15) managed to describe the development of pH and acid concentration well, with a mean deviation from the experimental results of less than 0.2 pH units (Paper V) and was therefore not elaborated further.

The cumulative model and the production-consumption model are thus both applicable to the experimental results (Papers II, IV and V). Therefore neither of them can be discarded as models of the low pH phase. However, it remains to be seen which one is more generally applicable.

There are indications that certain substrate components are critical for the acid dynamics in composting. During the initial low-pH phase, when organic acids are produced and decomposed at high rates, sugar and starch are the main substrate components that are degraded (Paper II). Furthermore, lactic acid bacteria are important acid-producing microorganisms, and they grow mainly on carbohydrates (Axelsson, 1998). It is thus likely that the acid production rate depends on the concentration of sugar and starch and that the rate is reduced when these components are degraded. If the production-consumption model was changed to account for decreased acid production when sugar and starch are depleted, the two models would give more similar results. In its current state, the production-consumption model does not predict similar cumulative CO_2 levels at the time of pH shift when there are large variations in the length of the acidic phase. For very slow decomposition, the production-decomposition model predicts that the compost will not be neutralised at all, since acids will be produced at a rate higher than that at which they are decomposed. With the calibrated values for k_p and k_d (Paper V), the CO_2 emissions need to be higher than 1.5% of initial C per day in order to get decreased acid concentration and increased pH.

Experimental scales

The composting experiments were performed at different scales. There are several advantages with this kind of approach, in terms of time, costs, applicability and flexibility. Large-scale experiments are costly and difficult as regards control, sampling and measurements. Pilot-scale reactors are easier to control and monitor, and the process conditions can be kept more uniform (Paper II), or they can be designed to have similar gradients to the large-scale process (Sundberg & Jönsson, 2003). Pilot-scale or large-scale experiments can be combined with detailed composting of parallel samples from the larger process at small scale. Such experiments can provide detailed insights into special issues, and allow certain parameters to be investigated in detail, as was the case with pH and temperature in Paper I. However, small detailed composting experiments cannot represent the process development over larger time scales.

The energy balance is largely affected by the scale of the operation. The surface-to volume ratio is much larger in a pilot-scale reactor than in a full-scale reactor, so the surface heat losses are larger at pilot-scale. This can be partly improved by insulating the reactor, but it is difficult to fully eliminate the conductive heat loss from small reactors. Therefore, the process conditions will be different in pilot-scale and large-scale reactors. To keep the same temperature at pilot-scale as at large-scale, the aeration must be reduced, which gives lower oxygen content and less moisture loss than in the full-scale process. In the pilot-scale reactor in Paper II, temperature and oxygen were independently controlled by cooling recycled air and adding a controlled amount of fresh air. This method decouples temperature and oxygen concentration, but not moisture loss. The chosen method of adding water to keep the moisture content constant allows the process conditions to be kept fairly uniform over time, and the effects of variable moisture contents are excluded from the investigation.

Moisture, temperature, oxygen consumption and aeration

The energy balance, *i.e.* energy flows and transformations, is crucial to the process development in composting since it (i) determines the temperature and (ii) affects the evaporation. The temperature is an important parameter that determines microbial activity and thus the degradation rate (Paper II). The evaporation reduces the moisture content, which is important for the degradation rate both directly, as it affects microbial activity, and indirectly, as it affects the structure and therefore the oxygen supply. The two main processes determining the energy balance are the heat generated by decomposition and the heat removed by aeration. In some systems, surface heat losses are also important in the energy balance. The aeration rate is a key parameter since the gas carries both heat and vapour, and it also supplies oxygen to the process. The multiple functions of aeration make temperature, oxygen consumption and water loss dependent variables, so the possible combinations are limited in a given physical system, *e.g.* a pilot-scale reactor or a large-scale plant. Some of these inter-dependencies were observed in the experiments (*e.g.* Paper II and Paper IV) and a theoretical investigation was set up in order to further study the relationships between temperature, moisture, aeration rate and oxygen concentration in different composting systems. More specifically, the aim was to compare the aeration rates needed to keep different constant temperatures at given decomposition rates in large and small composting systems, as well as the resulting moisture losses and exhaust oxygen concentrations.

Method

Calculations of the aeration needed for cooling were made with a static model based on energy and mass balances (eq. 1-4). The oxygen concentration of the exhaust gases and the water losses from the composting mass were also calculated. The assumed substrate was waste with a water content of 65% of dry matter and an ash content of 24%. The ambient and input air temperatures were set at either 0, 10 or 20 °C, and the relative humidity of the input air was set at 70%. The process temperature was set at 37, 55 or 70 °C. The exiting gas was assumed to be saturated with moisture.

Firstly, three systems were compared:

- Large-scale plant where 1% of the waste was decomposed in one day
- Large-scale plant where 5% of the waste was decomposed in one day
- Pilot-scale plant where 5% of the waste was decomposed in one day

The surface heat loss in the large-scale plant was assumed to be 1 W $°C^{-1}$ per ton of substrate, which is equivalent to 7% of the heat produced when 1% is decomposed in one day in a 55 °C compost, at an ambient temperature of 10 °C. Data for the pilot-scale reactor were taken from the reactor used in Paper II: surface area 1.84 m^2, thermal conductivity of insulation 0.04 W $°C^{-1}$ m^{-1}, 0.048 m of insulation, and 55 kg of substrate. This is equivalent to 28 W $°C^{-1}$ per ton of substrate.

Secondly, different scenarios for the decomposition rate and the composting time in large-scale composting were compared, with the following assumptions:

- Initial phase. The degradation of the organic matter was 10%. At 37 °C this was accomplished in 4 days. At 55 °C and 70 °C, twice that time was required. These assumptions are based on Smårs *et al.* (2002) and Paper II. The temperature of the material increased from ambient temperature to 37, 55 or 70 °C.
- High-rate phase. The degradation of 30% of the organic matter took 8 days at 55 °C, while twice that time was required at 37 and 70 °C (based on Paper II).

The model was static, which implies that variations over time were not considered and only total or average values were calculated.

Results and discussion

Evaporation was the main heat sink for all simulated systems (Figure 17). The surface heat losses under simulated large-scale conditions were below 10% at all temperatures (Figure 17). At laboratory scale, the surface losses were between 20 and 45%, which illustrates the large influence of scale on the thermodynamic conditions. The results are in agreement with previous thermodynamic studies of composting systems (Bach *et al.*, 1987; Weppen, 2001).

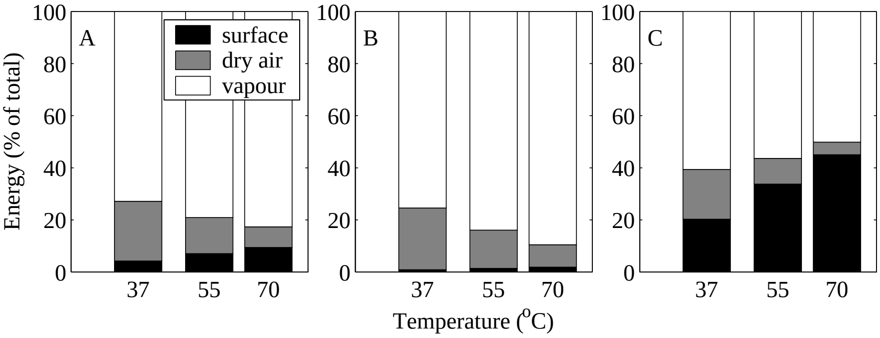

Figure 17. Heat sinks in large-scale composts with (A) 1% and (B) 5% decomposition per day, and (C) in a pilot-scale reactor with 5% decomposition per day.

There was a large difference in the aeration rates needed to maintain the different constant temperatures (Figure 18). Seven to ten times more air was needed to keep the compost at 37 °C compared to 70 °C. Large-scale composting required more air than pilot-scale, 14% more at 37 °C and 57% more at 70 °C.

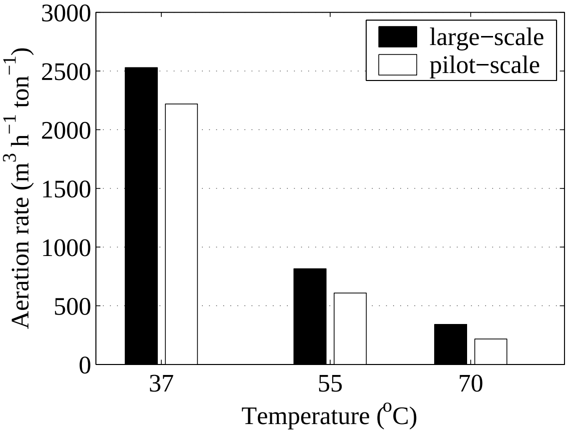

Figure 18. Aeration rate (m^3 h^{-1} ton^{-1} of fresh waste) required when composting at 37, 55, or 70 °C, in large scale or pilot scale, with ambient temperature 20 °C, when 5% is decomposed per day.

The ambient temperature had only a minor effect on the aeration rate required at thermophilic temperatures, but at 37 °C, 55% more air was needed at 20 °C outdoor temperature compared to 0 °C (Figure 19).

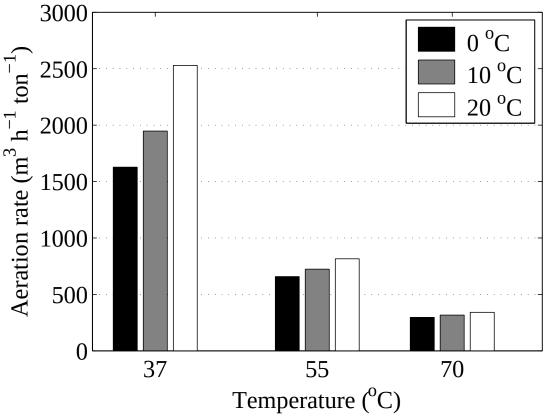

Figure 19. Aeration needed to keep the temperature of a large-scale compost constant at 37, 55 or 70 °C when the ambient temperature is 0, 10 or 20 °C. The decomposition rate is 5% per day and large-scale plant data were used.

The difference in water loss between different process temperatures was small, less than 22%, despite the large difference in aeration rate (Figure 20). At large-scale, when the surface heat losses were small, the water losses increased with process temperature, whereas the water loss decreased with temperature when the surface heat losses were large.

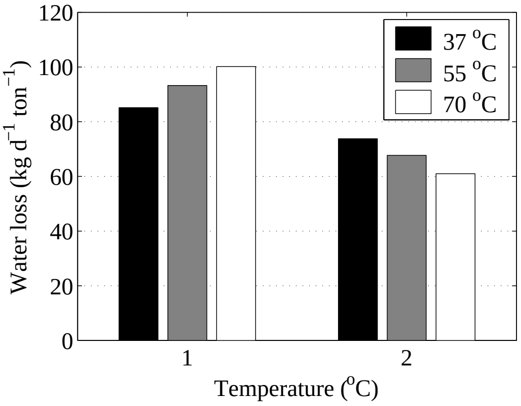

Figure 20. Water loss from 1 ton of fresh waste composted at 37, 55 or 70°C at large-scale (left) or pilot-scale (right). Input air temperature 20 °C.

The fraction of input oxygen that was consumed increased with temperature, but even at 70 °C more than 80% remained in the exhaust in the simulated large-scale systems (Figure 21). A larger proportion of the oxygen is consumed in the laboratory-scale reactor. This is a direct consequence of the lower aeration requirement for cooling due to the larger surface heat losses from the laboratory-scale reactor. The difference between decomposition rates at the same temperature is minimal, and thus the O_2 concentration cannot be used to determine the decomposition rate. This explains the finding in the large-scale experiment (Paper

IV) that the O_2 concentrations were very similar even when the aeration and decomposition differed widely.

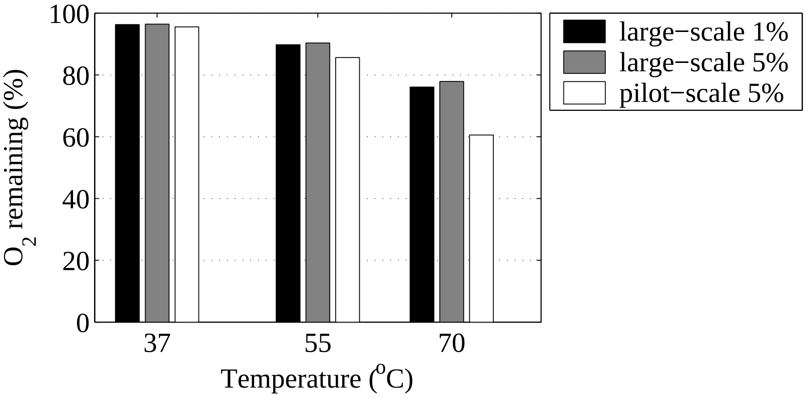

Figure 21. The remaining oxygen, as % of input oxygen, in the exhaust from composting at 37, 55 and 70 °C, with 1 or 5% of the organic matter decomposed in one day, at large-scale and laboratory-scale.

During the initial phase, heating the compost material accounted for 10-30% of the heat released (Figure 22). Evaporation was the major heat sink and accounted for about 60% of the heat produced in the initial phase and 74-86% during the high-rate phase. For the total evaporation it made little difference whether the heat was transported with much air carrying little steam (37 °C) or less air carrying more steam (70 °C) (Figure 20). During the initial phase the simulated water losses were highest at 37 °C (Table 2) whereas during the high-rate phase the water losses were highest at 70 °C (Table 3).

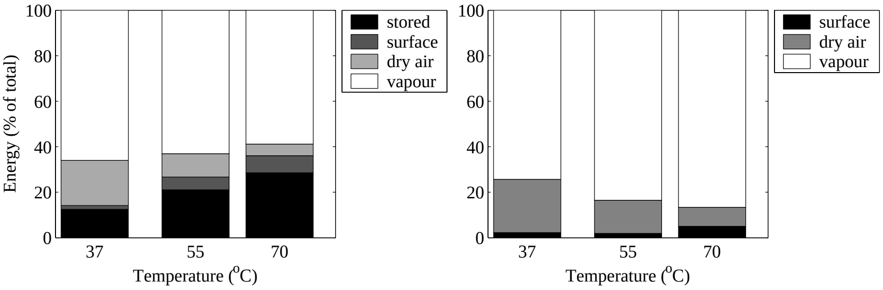

Figure 22. Heat sinks in simulated large-scale composts during the initial phase (left) and the high-rate phase (right). 'Stored' means the heat accumulated in the compost material.

Table 2. *Aeration requirement, water loss and oxygen remaining in the exhaust during the initial stage of composting of 1 ton of waste*

Temperature (°C)	Time (days)	Total aeration (m^3)	Aeration rate (m^3/h)	Water loss (kg)	O_2 remaining (%)
37	4	3250	33.9	129	96
55	8	1010	5.2	121	86
70	8	370	2.0	111	63

Table 3. *Aeration requirement, water loss and oxygen remaining in the exhaust during the high-rate phase of composting of 1 ton of waste*

Temperature (°C)	Time (days)	Total aeration (m^3)	Aeration rate (m^3/h)	Water loss (kg)	O_2 remaining (%)
37	16	11500	30.0	465	96
55	8	4300	22.5	538	90
70	16	1800	4.8	570	77

These simulations confirm that composting at low temperature requires much more air, but they also show that drying is fairly independent of temperature, and it is not proportional to aeration. If the increased aeration leads to lower composting temperature, aerating the compost more does not increase the evaporation taking place during the decomposition of a given amount of organic matter. This is because the vapour-carrying capacity of air increases exponentially with temperature, leading to increased evaporation at higher temperature. During the initial stage, a little more water is lost at 37 than at 55 or 70°C, but this may be acceptable considering that the reduction in start-up time may be larger than the 50% assumed here. These results indicate that moisture loss is a less important factor when deciding the aeration rate and process temperature.

The simulations are based on the assumption that the exhaust gas is saturated, which can be expected to be valid when the moisture level of the compost is higher than 50% (Bach *et al.*, 1987). During the high-rate phase (Table 3), the moisture content falls below this figure if no extra water is added. As a matter of fact, the simulated water losses are higher than the initial water content. This highlights the need for water addition, as discussed below.

Water ratio

Since evaporation is the major means of cooling a large-scale composting system (Figure O, E), this is a process with very high drying potential. The self-heating process can be used for intentional drying of wet substrates such as sewage sludge or manure (Nakasaki *et al.*, 1987; Choi, Richard & Ahn, 2001). However, when substrates are wet, there is a risk that there will be too little energy available to heat and dry the substrate during the process. As a rule of thumb the water ratio (W) defined as:

$$W = \frac{\text{weight of water}}{\text{weight of biodegradable matter}}$$

should not exceed 8-10 (Haug, 1993). The ratio of water to biodegradable matter (W) is also of importance for composting of dry substrates. Substrates with large amounts of easily degradable energy in relation to their water content can become too dry long before the compost is stabilised. With the same model as used in Paper I and the calculations above, the amount of water needed during decomposition was estimated. The calculations showed that to achieve 50% degradation of the organic matter and a product with 30% water content from a substrate with 76% volatile solids, the initial water content would need to be 69-73%, varying slightly with the ambient and process temperatures. This is equivalent to a water ratio of 3.2-4.0, if all the volatile solids are assumed to be biodegradable. It is therefore recommended that addition of water be considered for large-scale composting of waste with water ratios below 3-4. The wastes used in the experiments reported in this thesis had water ratios of 2.4 (Paper II), 0.9 (Paper III) and 1.7-2.4 (Paper IV), so they all needed water addition, otherwise they would have become too dry before 50% of the organic matter had decomposed. It is often impossible for the substrate to hold all the water that is needed during the process while retaining a structure that allows for aeration. Water must then be added during the process, as was the case in all experiments in Papers II-IV.

Process control

To increase the efficiency in composting, we want to maximise the decomposition rate. In a pilot-scale reactor, there are normally several options for control of the system, but in full-scale composting the options are fewer. In a large-scale system with active aeration, the main control variable is the aeration rate (Figure 23).

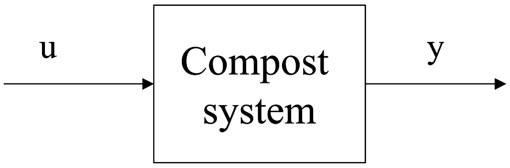

Figure 23. Aeration rate (*u*) is input to the composting system, and affects the goal variable, the decomposition rate (*y*).

A major problem when maximising the decomposition rate is that this rate is not a physical property that we can measure directly. However, variables such as temperature, O_2 and CO_2 concentration, pH and moisture can be measured more or less on-line during the process. Therefore the control problem can be divided into two steps (Figure 24). The aeration rate affects the measurable quantities (*x*) and they may influence, or tell something about the goal variable, the decomposition rate (*y*).

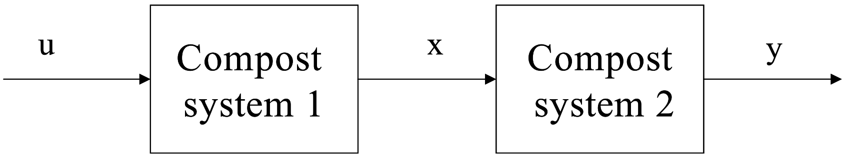

Figure 24. Aeration rate (*u*) is input to the system and affects the measurable variables (*x)*: the temperature, O_2 and CO_2 concentration, pH and moisture, which in turn affect the decomposition rate (*y*).

This leaves us with two principally different control options:

> A: If we know how *x* affects *y*, the aeration can be controlled by feedback from *x* (Figure 25).
> B: *y* can be estimated from *x*, and the estimated *y* can be used for feedback control (Figure 26).

These two options arise because of the double character of temperature and oxygen, which are both determinants of decomposition (used in A) and results of decomposition (used in B).

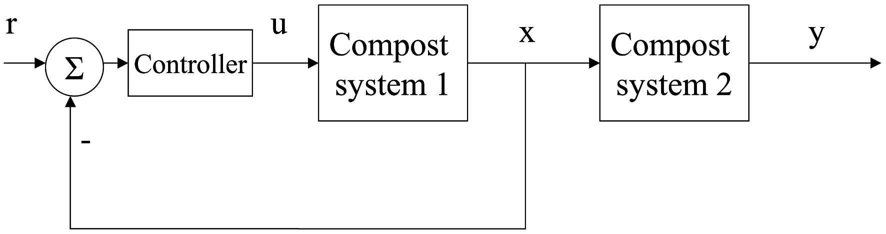

Figure 25. Control option A. The difference between the set-point value *r* for *x* and the actual value of *x* is input to the controller, which determines the aeration rate *u*, which is input to the system and affects the measurable variables *x*, which in turn influence the goal variables *y*.

For option A, there is a well-known strategy for the set-point for temperature control: keep the temperature at about 55°C. However, this strategy does not always work. Firstly, a control algorithm that says ‘reduce aeration when the temperature is below 55 °C’ is not functional before the temperature has reached 55 °C for the first time. Consequently, temperature control cannot be set in until the set-point temperature has been reached, and the aeration rate during the initiation of the process must be determined in some other way. Secondly, 55-60 °C is far above optimum when the pH is low (Paper I). Therefore, based on the results in Papers I, II and IV, an improved control strategy would be: When pH is below 6.5, aerate to keep the temperature below 40°C, or, if that is not possible, aerate at full capacity. When pH is above 6.5, the set-point temperature should be about 55°C. This control strategy needs only temperature and pH as measured *x*.

Another parameter that is often measured in composting is the O_2 concentration. This is relevant to ensure that the process is aerobic, but has limited value as a process indicator. As was shown in Paper IV, the difference in O_2 concentration is very small even at large variations in aeration rate and decomposition rate, so the O_2 concentration is not a sensitive parameter for use as a control variable x.

In control system A it is important that the x and set-points for x are chosen with care. For example, a common objection to increased aeration rates is that the compost will dry out. This is only true if moisture is related to time, but not if it is related to the decomposition rate (see section on Moisture, temperature, *etc.* above). Thus, using moisture as x for controlling aeration will not maximise the decomposition rate. Moisture is important for decomposition, but successful moisture control must include water addition.

The control strategy A with feedback based on pH and temperature (Figure 25) is a simple and straightforward method for process control. However, there are reasons to also take a closer look at control option B (Figure 26). It is possible to estimate the decomposition rate by combining data on aeration rate with either temperature, CO_2 or O_2 concentration, for example:

$$O_2 \text{ consumption} = \text{aeration rate} \cdot (O_{2,out} - O_{2,in}) \qquad (16)$$

where $O_{2,out}$ and $O_{2,in}$ are the O_2 concentrations of the output and input air, respectively. This gives us variables of heat evolution, CO_2 emission or O_2 consumption, which are all on-line measures of the decomposition rate. This requires that x includes a measure of the aeration rate, and either temperature, CO_2 or O_2 concentration. The measure of the aeration rate does not have to be an absolute value, it may be a relative value related to the maximum aeration capacity. To estimate heat evolution from temperature and aeration is slightly more complicated than Eq. 16, since heat evolution is not a linear function of temperature, but the principle is the same (see Paper V).

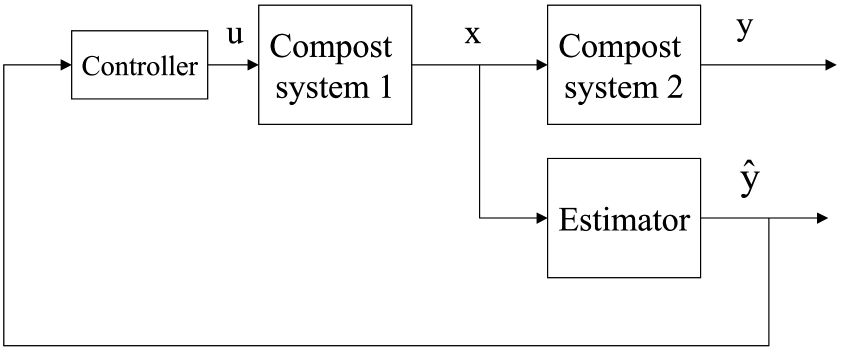

Figure 26. The measured values of x are used to estimate the decomposition rate y. This value ($\hat{y}$) is input to the controller that sets the aeration rate u.

Using feedback from the estimate of the variable of primary interest, the decomposition rate, has advantages. For optimising the process beyond the simple temperature control strategy, an estimate of the actual decomposition rate is valuable. In large-scale composts that are aerated in one direction only, large temperature gradients develop in the mass during decomposition, since cool air enters at one end and is heated while passing through the mass. It is therefore not possible in practice to keep a uniform temperature in large composts. With an estimate of the decomposition rate, it is possible to evaluate the effect of different temperature set-points. The optimum set-point depends on several factors, for example the location of the temperature sensors in the compost mass or in the exhaust (Ekinci, Keener & Akbolat, 2004).

Process efficiency

In this thesis, compost process efficiency is defined as rapid organic carbon turnover. To make composting efficient in an economic sense too, other things must be considered, such as the cost of energy, labour and equipment. Efficiency in terms of minimal negative environmental effects is also important. One important environmental aspect of composting is ammonia emissions, which are discussed in Paper II. Rapid decomposition is a way to achieve efficient use of space and machinery, as well as to minimise unwanted environmental effects such as methane and odour emissions. Furthermore, by knowing how the processes work and how they can be run efficiently, it is possible to make a rational trade-off between different management objectives.

Nitrogen

The nitrogen dynamics are important since emissions of NH_3 and N_2O affect the environment and should be minimised. It is also relevant to be able to optimise the nitrogen retention in the compost.

More than two-thirds of the ammonia released was collected in the condensate from the compost reactor in the experimental runs reported in Paper II, and in most runs more than 90% of the ammonia was in the condensate. This confirms that condensation of the off-gas is a good way to reduce the ammonia emissions, as was concluded by Beck-Friis *et al.* (2001) from experimental results in the same reactor.

The unaccounted-for nitrogen losses from the experiments in Paper II were large, 10-40% of the initial N. One possible explanations for this is denitrification, which has been shown to be significant in composting under a wide range of temperature and oxygen conditions (Körner & Stegmann, 2004).

NH_3 emissions depend strongly on temperature (Paper II). Therefore future laboratory experiments should preferably be carried out in reactors with controlled temperature. If auto-thermal reactors are used, the effects of different substrates, amendments, O_2 concentrations or moisture conditions tend to be confused with the effects of temperature. This has often been the case in the past, and therefore many experimental results are difficult to interpret.

Future research

It would be interesting to investigate the effects of sugar and starch on organic acid production in the initial phase of batch processes, or in fed-batch processes. So far, we have only investigated the same substrates under different conditions, but experiments in which a mixed substrate is spiked with pure starch or sugar could provide valuable information on organic acid production and accumulation.

The methods for investigating the microbial community in compost have improved in recent years, but there are few studies examining the link between process management and the microbial community. A better understanding of the interaction between process conditions and the microbial community would be valuable for the advancement of composting process knowledge.

There is still very much research needed for a full understanding of the nitrogen dynamics of composting of different substrates under various conditions. There is a need for laboratory and pilot-scale experiments to investigate details under controlled conditions, and for field-scale experiments to obtain information on the magnitude of the actual emissions from composting in practice and on methods to reduce the emissions.

The composting process can develop in very different ways depending on aeration, mixing, substrate preparation, *etc*. This will certainly influence the odour released during the process, as well as the remaining odour potential after a certain processing time. Further research is needed on the effects of different process conditions on the production and emission of malodorous substances.

There are many publications on models of batch compost processes but so far very little has been published on fed-batch modelling (Nakasaki & Ohtaki, 2002). It should be possible to apply principles from the model presented in this thesis or others to model fed-batch composting processes. To adapt the model in Paper V, substrate limitation must be included, since fed-batch models reach a stationary phase with a daily maximum in decomposition rate and no increase from day to day. The data from Paper III could be used for such a modelling investigation, although it may be easier to work with data from constant-temperature experiments.

The model presented in Paper V is based mainly on empirical data, while the static model used in the Discussion section is based on simple physical relationships. However, an improved model could be developed by utilising more knowledge on the actual physical processes taking place in the compost. The model is based on the composting reactor, which is a very homogeneous environment. In practice, however, most composts have large gradients in temperature, aeration rate and oxygen concentration. These gradients are important for the process, and could be modelled by taking into account the physics of thermal conduction and convection.

Conclusions

- In both fed-batch composting and batch composting of food waste, a prolonged acidic phase can occur, resulting in low degradation in composting processes.
- Inhibition of thermophiles at low pH is an important key to explaining the lag in the transition from mesophilic to thermophilic conditions in the initial phase of batch composting.
- The dynamics during the initial self-heating phase of batch composting can be modelled as a process governed by a mesophilic microbial community tolerant to low pH and a thermophilic community inhibited by low pH.
- The acidic phase can be shortened by:
    - Keeping the temperature low, below the mesophilic optimum
    - Increasing the aeration rate, even if the temperature remains above the mesophilic optimum
    - Mixing the fresh substrate with compost
    - Adding alkaline substances
- When composting waste with an initial low pH, the pH value is an important process parameter, firstly because it is correlated to the decomposition achieved and the stability of the compost, and secondly because the optimal process temperature depends on pH.
- When composting source-separated household waste at 40, 55 or 67 °C, the decomposition of organic matter during the high-rate phase is fastest at 55 °C. There is little difference in decomposition rate at 40 and 67 °C. Decomposition of fat is slower at 40 °C than at 55 or 67 °C.
- In controlled reactor experiments, the ammonia emissions at 67 °C were more than double those at 55°C, and they were higher at 55 °C than at 40 °C. It is thus important to keep the temperature down in order to minimise the ammonia emissions.
- Concentration of O_2 and CO_2 have a limited value as process control parameters, since they vary very little even when differences in decomposition rate are large. However, heat production, O_2 consumption and CO_2 emission are valuable process control parameters.
- Experimentation at different scales is an efficient experimental strategy, but requires that thermal issues be properly addressed, since the conductive-radiative heat losses vary between scales, and this affects the relationship between temperature, O_2 concentration and moisture.

- In large-scale composting, the water loss is mainly determined by the accumulated decomposition. The process temperature has very little effect on the water loss.
- If the ratio of water to degradable matter in the substrate is lower than 3-4, water addition is necessary in order to avoid premature drying in large-scale composting.

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Acknowledgements

The main financial support for this thesis project was provided by the Swedish University of Agricultural Sciences and the Swedish Research Council for Environment, Agricultural Sciences and Spatial Planning (FORMAS).

I am grateful to many people who contributed to my PhD adventure. It has been a very rewarding time and I want to thank you all!

First of all I want to thank Håkan Jönsson, my main supervisor, for support and inspiration. You have helped me turn heaps of waste into science with a never-ending enthusiasm that has inspired me through all these days!

I especially want to thank Bengt Carlsson for helping me to condense my volatile ideas on compost modelling.

Thanks to Sven Smårs and Barbro Beck-Friis. Your research and your warmth have been a great inspiration to me! I also want to thank Sven for help with practicalities such as experiments, computers and boat engines.

Thanks to Håkan for creating a lively team with your disciples, and to all team members, especially Björn Vinnerås and Pernilla Tidåker, for cooperation and friendship.

I want to thank everybody in the compost group at SLU; co-authors, Åsa Jarvis and others, for creating a multidisciplinary environment that was a great basis for learning composting research. I especially want to thank Ylva Eklind for good writing co-operation.

Thanks to Dick Gustafsson for practical help with the composting experiments.

Thanks to Britt, Elin, Kjell-Inge, Morten, Oddvar, Roy Harald and many others at the Norwegian composting plants for all the help with the experiments and the arrangements for my visits. And to Erik Norgaard who made the whole project possible!

To past and present doctoral students and other colleagues at the Department of Biometry and Engineering, thanks for caring and sharing.

Thanks to all the people I have met at the Department of Microbiology, for experimental assistance and for discussions on microbial and human life.

I want to thank Torbjörn Ånger, VAFAB, and Johannes Raivio, then at Joraform, for early eye-opening contacts with the real world of composting outside the university.

Thanks to the Department of Crop Science, Alnarp, and especially Martin Robertsson, for lending me the rotating reactors. They have done some good work here in Uppsala!

Many thanks to friends and family, for reminding me of life outside the compost world.

Thank you Petter, for every day.