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Biowaste and vegetable waste compost application to agriculture

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Biowaste and vegetable waste compost application to agriculture

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Abstract

The landfilling of biodegradable waste is proven to contribute to environmental degradation. Compost use in agriculture is increasing as both an alternative to landfilling for the management of biodegradable waste, as well as means of increasing or preserving soil organic matter. This research aimed to contribute to the identification of a system for managing the utilization of vegetable waste (agricultural plant-tissue waste) and biowaste (source-separated biodegradable municipal solid waste) composts for sustainable agriculture, with particular regards to nitrogen availability and leaching.

The two compost types were analyzed for physical and chemical properties potentially related to soil and crop parameters. Composts were applied to forage maize and annual ryegrass production on light textured soils, using different rates and methods, to determine the effects on crop production, soil properties and N leaching, in lysimeter, pot and field conditions. An incubation trial was carried out to study compost N mineralization potential under controlled conditions. The economic implications of using compost in agriculture were also considered.

Biowaste and vegetable waste compost properties varied significantly between compost types and dates of production, thus suggesting that to properly manage compost application, agricultural practise should adjust to the specific characteristics of each material. Both composts were low in nutrients. A quantity of 19-30 Mg biowaste or 50-63 Mg vegetable waste compost was necessary in order to apply 250 kg total N ha^{-1}.

There was higher nitrogen mineralization from vegetable waste compost than from biowaste compost. This was attributed to the lower C: N ratio and/or the lower content of recalcitrant to decomposition carbon sources. Compost with C: N lower than 15 is likely to result in net mineralization of organic N, thus increasing soil available N and excluding negative effects on crop production from N immobilization. N mineralization following the application of biowaste and vegetable waste composts with C: N ratios varying from 9 to 21, ranged from approximately -4% to 10% of the total N applied, depending mainly on the compost C quality, climatic and soil conditions. Compost N mineralization was not influenced by application rate and method.

Applying biowaste or vegetable waste compost based on their total N content, whilst taking into consideration the compost N mineralization potential, the crop N requirements and the soil N is not likely to increase N leaching on light textured soils, in comparison to non-amended soil. Properly managed compost application resulted in less N leaching than mineral fertilizer N, because of the slow release of the compost N mineralized, which allows for more efficient crop N uptake.

Forage maize and annual ryegrass dry matter yield significantly increased following biowaste and vegetable waste compost application. Crop N recovery varied from 0% to 12% of the compost total N applied. Higher DM yield and N uptake was achieved with higher rates of onion compost application.

The application of biowaste and vegetable waste composts increased soil total C, OM, organic N, and extractable K content and water retention of light textured soils (the higher the application rate, the higher the increase), thus improving soil fertility for better crop production, and water conservation. However, it also increased extractable Na levels hence potentially introducing salinity and/or sodicity problems in Na sensitive soils and crops. Surface applied biowaste or vegetable waste compost resulted in higher soil moisture, under enhanced evaporation conditions, and in lower Na content. Compost application increased P solubility and resulted in transport of soil available P below the root zone of coarse textured sandy soils thus reducing P availability for plant uptake and increasing the risk of water eutrophication. However, the increase of P solubility prevented the fixation of available P, thus suggesting that compost application in calcareous alkaline soils may increase P availability for plant uptake.

The cost of using biowaste or vegetable waste compost to agriculture is relatively high; considering a compost application rate of 250 kg total N ha^{-1} yr^{-1}, the cost ranged from £172 to £ 489 ha^{-1} yr^{-1}. Key factors influencing the cost were the compost N content in fresh weight basis and the haulage distance. Despite the high costs involved, compost application to agriculture is a sustainable practise because of the increasing mineral fertilizer prices, and the resulting soil quality improvement, organic waste diversion from landfills, decreasing of N leaching, and preserving of non-renewable resources.

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Notation

ABPR	Animal By-Products Regulation
AD	Anaerobic digestion
As	Arsenic
C	Carbon
C: N ratio	Carbon to nitrogen ratio
Ca	Calcium
Cd	Cadmium
CEC	Cation exchange capacity
CFU	Colony forming unit
Co	Cobalt
Cr	Chromium
Cu	Copper
DEHP	Di-2-ethylhexyl phthalate
DM	Dry matter
DON	Dissolved organic nitrogen
EC	Electrical conductivity
EC_e	Electrical conductivity (saturation paste)
EEA	European environment agency
ESP	Exchangeable sodium percentage
EU	European Union
FC	Field capacity
Fe	Iron
ha	Hectare
Hg	Mercury
Inc	Incorporation
K_2O	Potash
K	Potassium
LAS	Linear Alkylbenzene Sulphonate
LSD	Least significant difference
MBT	Mechanical biological treatment
MF	Mineral fertilizer
Mg	Magnesium
Mg	Megagram (metric tonne)
Mo	Molybdenum
MSW	Municipal solid waste
N	Nitrogen
Na	Sodium
NH_4-N	Ammonium nitrogen
NH_4NO_3	Ammonium nitrate

Ni	Nickel
NMR	Nitrogen mineralization rate
NO_3-N	Nitrate nitrogen
NO_2-N	Nitrite nitrogen
NVZ	Nitrogen vulnerable zones
OM	Organic matter
OSN	Organic soluble nitrogen
P	Phosphorus
PAHs	Polycyclic aromatic hydrocarbons
Pb	Lead
PCBs	Polychlorinated biphenyls
P_2O_5	Phosphate
PO_4-P	Phosphate phosphorus
PTE	Potential toxic elements
PWP	Permanent wilting point
SFA	Segmented flow analysis
Surf	Surface application
TDN	Total dissolved nitrogen
TON-N	Total oxides of nitrogen nitrogen
TSN	Total soluble nitrogen
WD	Working document
WRC	Water release curve
yr	Year
Zn	Zinc

1 Introduction

1.1 Research background

1.1.1 The importance of waste recycling to land

Waste volumes continue to rise, which leads to loss of resources and increased environmental risks (CEC, 2002a). Excluding the agricultural waste, the total amount of waste generated annually in the EU was estimated at about 1.3 billions tonnes in 1995 (EEA, 1999). Traditionally, the main option for the management of waste has been landfilling.

The landfilling of biodegradable waste is proven to contribute to environmental degradation, mainly through the production of highly polluting leachate and methane gas. Methane constitutes one of the six greenhouse gases responsible for the global warming, which needs to be reduced, in order to tackle climate change under the Kyoto Protocol (UN, 1998). The methane emissions from landfills constitute about 30% of the global anthropogenic emissions of methane to the atmosphere (COM, 1996). Reducing the amount of methane emitted from landfills is considered to have the greatest potential for reducing the overall climate change impacts of waste management (Smith et al., 2001).

At the EU level, the Landfill Directive (CEC, 1999) is the main driver for the management of biodegradable waste. It restricts the disposal of biodegradable waste in landfills and promotes its diversion towards material recycling and biological treatment. The biological treatment of waste includes composting, anaerobic digestion, or mechanical-biological treatment.

Composting is strongly supported in the UK as an important component of meeting the Waste Strategy (DETR, 2000) targets for recycling and composting of biodegradable municipal solid waste (MSW), which were set under the EU Landfill Directive. The amount of organic waste composted in the UK in 2004-05 was estimated at 2.7 million tonnes, increased substantially from about 2 million tonnes in 2003-04, and 1.1 million tonnes in 2000-01 (Boulos et al., 2006).

According to the European Environmental Agency (EEA), MSW and agricultural waste are two of the five major waste streams in the EU. MSW represents about 14% of the total waste generated in the EU, excluding agricultural waste (COM, 2003). The term biowaste refers to the biodegradable part of MSW, which accounts for about 65% of the total MSW generated (EC, 2003b). Biowaste contains mainly food, garden, paper and cardboard waste.

Agricultural waste includes organic waste, such as animal excreta, plant residues, soiled water and silage effluent, and waste such as plastic, scrap machinery, fencing, pesticides, waste oils and veterinary medicines (EEA, 2006). No official overall estimates are available relating to the amount of agricultural waste produced in the EU (EEA, 2006). There are, however, unofficial estimations suggesting that European countries produce approximately 700 million tonnes of agricultural waste per year (ARIC, 2004). The term ‘vegetable waste’ can be used to describe the agricultural plant-tissue waste.

1.1.2 Compost application to agricultural soil

Composting helps to optimise nutrient management and the land application of compost may contribute to combat soil organic matter decline and soil erosion (Van-Camp et al., 2004). Compost land application completes a circle whereby nutrients and organic matter which have been removed in the harvested produce are replaced (Diener et al., 1993).

The recycling of compost to land is considered as a way of maintaining or restoring the quality of soils, mainly because of the fertilising or improving properties of the organic matter contained in them. Furthermore, it may contribute to the carbon sequestration, and may partially replace peat and fertilizers (Smith et al., 2001). At the EU level, there is sufficient agricultural land to provide a market for the compost in all member states, except for Luxembourg (Smith et al., 2001).

On the other hand, the application of compost to soil could raise environmental risks mainly related to excessive or unbalanced supply of nutrients, introduction of heavy metals and organic pollutants, and the spreading of pathogens (EC, 2003b). At present in the EU, in order for a compost to be suitable for agricultural application it is considered necessary to fulfil the environmental quality classes established in the 2nd draft of the working document on biological treatment of biowaste and to contribute to the improvement of soil conditions for crop production (EC, 2001).

Compost application to agricultural land needs to be carried out in a manner that ensures sustainable development. Management systems have to be developed to enable to maximize agronomic benefit, whilst ensuring the protection of environmental quality. The main determinant for efficient agronomic use is nitrogen availability. High nitrogen utilization in agriculture from mineral fertilizers is well established and understood, whereas increasing the nitrogen use efficiency of organic fertilizers requires further investigation (Amlinger et al., 2003; Gutser et al., 2005).

The work which has already been carried out for using biowaste and vegetable waste compost in agriculture has shown the low nitrogen fertilizer value of composts (Amlinger et al., 2003; Hartl & Erhart, 2005; Gutser et al., 2005). The compost nitrogen added to the soil is present mostly in organic compounds, and it can be mineralized and thus be taken up by the plants, immobilized, denitrified, and/or leached. In different studies, crop nitrogen recovery was found to range between 2% and 15% of the total compost N applied, depending on various factors, including compost properties, climatic conditions, crop types, soil properties and management practises (Mamo et al., 1999; Nevens & Reheul, 2003; Wolkowski, 2003; Hartl & Erhart, 2005).

Compost nitrogen availability is still poorly understood. Better understanding of the fate of nitrogen from biowaste and vegetable compost application to soil is necessary in order to quantify nitrogen availability to plants and nitrogen losses to water-bodies. It is important to develop integrated approaches to compost use in agriculture, which take into account agronomic benefits, and environmental risks, while identifying the financial implications of compost application. Such a holistic approach is critical to promote acceptance of compost use within the public and agricultural sector. This thesis is attempting to contribute towards the better scientific and economic understanding of biowaste and vegetable waste compost application to agriculture and towards the development of management systems aiming to maximize agronomic benefit and reduce the environmental risk.

1.2 Research aim

The aim of the research is to contribute towards the identification of a system for managing the utilization of vegetable waste and biowaste compost for sustainable agriculture, with particular regards to nitrogen availability and leaching.

1.3 Objectives

In order to meet the project aim the following objectives were developed:

- To determine the properties of organic waste compost types in relation to crop and soil factors.
- To evaluate the effect of compost type, application rate, and method on crop production, soil properties, and environmental risk from nitrate leaching.
- To assess the economic benefits of compost application to agricultural land.

1.4 Outline methodology

Having considered previous studies in the research area, the methodology produced to meet the specific objectives of the research is outlined in Figure 1.1, along with the research deliverables.

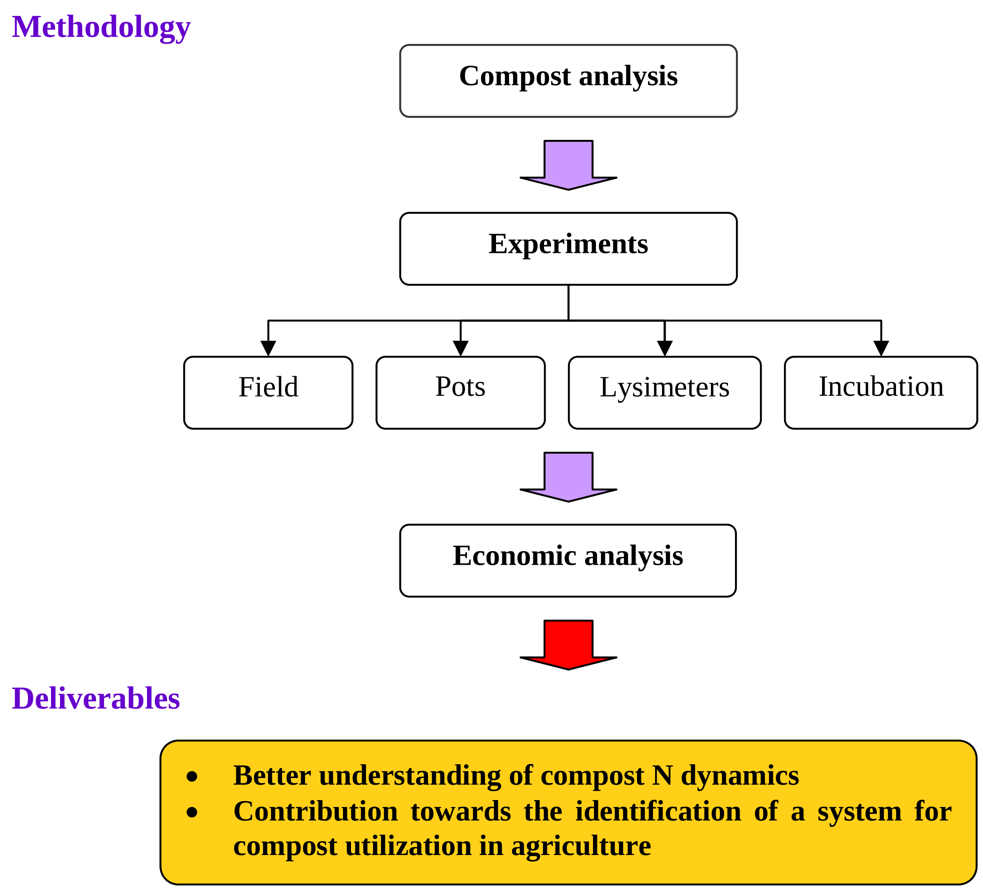

Figure 1.1 Outline of research methodology and deliverables.

Determination of compost analytical properties was required to assess the variation of compost properties in different compost types, and to produce understanding of the materials capability to supply crop essential nutrients and the potential interactions with soil quality parameters.

Experimental data were required for an accurate understanding of the relationship between applied composts and resultant crop yield and nitrate leaching. Three types of experiments were introduced: field, pot, and lysimeter. The field experiment was necessary to study the compost effects under actual farming conditions. This was essential in providing results which could be extended to practical recommendations for compost application to agricultural land, and also could allow for a more accurate assessment of the economic costs and benefits involved.

The pot experiment was supplementary to the field experiment, aiming primarily to allow for more accurate determination of the nitrate leaching risk. The lysimeter experiment was necessary to study the nitrogen leaching potential of the composts under high drainage rate conditions, and the agronomic benefits of compost application to poor quality soil.

An incubation trial was also included to determine the nitrogen mineralization potential of the composts under defined conditions of temperature and moisture, and to provide supportive evidence for the nitrogen availability data derived from the other three main trials.

Within each trial, the main factors under consideration were the compost type, the application rate and method. Table 1.1 summarises the four trials, including the variables evaluated and the factors considered.

Table 1.1 Outline of the experimental trials.

Trial type	Variables evaluated	Factors considered
Field	• Crop yield • Soil properties • Nitrogen leaching potential	• Compost type • Application rate
Pots	• Crop yield • Soil properties • Nitrogen leaching losses	• Compost type • Application rate • Application method
Lysimeters	• Crop yield • Soil properties • Nitrogen leaching losses under high drainage rate conditions	• Compost type • Application rate
Incubation	• Compost nitrogen mineralization	• Compost type • Application rate

An analysis of the economic implications involved in compost utilization in agriculture was required to study whether compost application could provide economic returns to the farmer, and thus whether farmers would be willing to use these materials on their land.

1.5 Thesis structure

Following this brief introduction to the research, Chapter 2 reviews the currently available knowledge on biowaste and vegetable waste composting and land application. Information about the two compost types considered in this research is presented in Chapter 3, along with the methodology followed for the compost analysis and also the results and discussion. Chapters 4, 5, 6, and 7 contain the experimental methodology, the results and analysis of the field, pot, lysimeter, and incubation trials, respectively. Chapter 8 provides an economic analysis and evaluation of compost application to agriculture. Chapter 9 contains an overall analysis and discussion of the main research findings from the four experiments. The final conclusions and recommendations are summarised in Chapter 10.

2 Literature review

2.1 Introduction

This chapter aims to engage in the main aspects related to biowaste and vegetable waste composts and their application to agricultural soil. The literature review summarizes the pertinent knowledge concerning biodegradable waste processing, biowaste and vegetable waste composts and their application to agricultural soil, focusing on the benefits and potential environmental impacts for the soil-plant-water system and the economic implications involved.

2.2 Biodegradable waste processing

2.2.1 General principles

The waste policy developed within the European Union is based on the waste hierarchy, which lists the different options for managing waste from the most favourable to the least favourable from an environmental perspective. After waste prevention and re-use, the waste management hierarchy accords the highest preference to recycling/composting, over energy recovery and disposal options (CEC, 2002a).

Waste prevention is the most favourable option. However, every material placed on the market is bound to become waste. Therefore, waste prevention must be complemented by a policy to promote environmental sound waste recovery (COM, 2003). The implementation of the Landfill Directive (CEC, 1999) promotes the diversion of waste towards material recycling and biological treatments. According to the landfill directive the amount of biodegradable waste landfilled needs to be reduced in order to reduce the production of methane gas from landfills, which adds to the global warming.

The general principles developed under the landfill directive for the management of biodegradable waste are in order of preference as follows:

i) prevent or reduce biodegradable waste production and its contamination by pollutants,
ii) re-use of biodegradable waste (e.g. cardboard),
iii) recycle separate collected biodegradable waste into the original material (e.g. paper and cardboard),
iv) composting or anaerobic digestion of separately collected biodegradable waste, which is not recycled into the original material, with the utilization of the end product for agricultural benefit or ecological improvement,
v) mechanical and biological treatment of residual waste, and
vi) energy recovery (EC, 2001; Smith et al., 2001).

Composting, anaerobic digestion, and biological mechanical treatment are types of biological waste treatment. The term biodegradable waste is used within the Landfill Directive to describe any waste which is capable of undergoing aerobic or anaerobic decomposition, such as food and garden waste, paper and cardboard.

According to the European Environmental Agency (EEA), between 30% and 40% of the municipal solid waste (MSW) consists of food and garden waste, and 20 % to 30% of paper and cardboard waste. In total between 60% and 70% of MSW can be considered as biodegradable waste (EEA, 1999). Biodegradable waste may also include biodegradable residues and wastewater treatment sludges produced by agricultural, food and drink, wood processing, paper, leather, and packaging industries, and sewage sludge (EC, 2001).

2.2.2 Biological treatment of biodegradable waste

The biological treatment of biodegradable waste involves either aerobic or anaerobic techniques. Biological processing includes composting, anaerobic digestion and mechanical biological treatment.

2.2.2.1 Composting

Composting is a natural process which involves the aerobic biological decomposition of organic materials under controlled conditions (‘Mac’ Safley et al., 1992; Pace et al., 1995; Misra et al., 2003). During composting organic matter from the biodegradable wastes is microbiologically degraded, resulting in a final product containing stabilised carbon, nitrogen and other nutrients in the organic fraction, the stability depending on the compost maturity (Zwart, 2003).

Biodegradable waste is composted with the objective of returning the waste to the plant production cycle as fertiliser and soil improver (Crowe et al., 2002). Composting helps to optimise nutrient management and the land application of compost may contribute to combat soil organic matter decline and soil erosion (Van-Camp et al., 2004). Furthermore, it may partially replace peat and fertilizers (Smith et al., 2001). Composting of vegetable waste produced in horticulture may reduce the environmental impact on climate change, at a rate of about 40% and 70% respectively, compared to landfilling and incineration (Antón et al., 2005).

Materials which are typically suitable for composting include green waste and biodegradable municipal waste, plus other enriched organic waste streams, such as sewage sludge, agricultural, and food processing wastes (Last, 2006). Biodegradable wastes must be separated prior to composting: only pure food waste, garden waste, wood chips and to some extent paper waste are suitable for producing good quality compost (Crowe et al., 2002).

Composting can be carried out in private gardens (home composting), on-farm (Figure 2.1), or in advanced highly technological centralised plants (Evans, 2001; Crowe et al., 2002; Misra et al., 2003).

The composting process may begin as soon as biodegradable waste is piled together. A structure allowing oxygen diffusion through the pile and a moisture content suitable to support the metabolic processes of the microorganisms are necessary (Pace et al., 1995; Last, 2006). The temperature of the biomass increases due to the microbial activity and the insulation properties of the piled material. The temperature often reaches 65–75°C within a few days (thermophilic phase) and then declines slowly. This high temperature furthers the elimination of pathogens and weed seeds (Crowe et al., 2002). However, for composting to occur in an optimum manner, five key factors need to be controlled: temperature, moisture content, oxygen content, material particle size and nature of the feedstock with particular importance to carbon over nitrogen (C: N) ratio (Pace et al., 1995; Evans, 2001; Last, 2006).

An inevitable by-product of all composting processes is heat. The heat recovery is poor compared to other processing alternatives, thus composting remains a net energy consumer (EA, 2007). The decomposition process also results in the production of carbon dioxide, water vapour, and humus, which is a stabilised organic residue. Depending on the composition of the waste material and the applied method of composting, the compost will be ready after 3 to 18 months.

Figure 2.1 On-farm composting.

During composting of biodegradable municipal waste (predominantly kitchen and garden waste) 20% to 40% of the nitrogen contained in the waste is lost as ammonia, and 40% to 60 % of the carbon as carbon dioxide (Crowe et al., 2002). Beck & Friis (2001) found that ammonia emissions from composting of organic household waste was between 24% and 32% of the initial total N. Komilis & Ham (2006) showed that food waste has a higher potential for ammonia volatilization during composting than yard waste, amounted to 65% and 25% of the initial total N, respectively. On average, about 40-50% of the waste input is converted to compost. Within the compost produced there is 2.5-10 kg N, 0.5-1 kg P and 1-2 kg K per 1 tonne of waste input (Crowe et al., 2002).

Composting of biodegradable waste reduces bulk, concentrates the nutrients, reduces odour, kills pathogens and produces a stabilized product for transport (Westerman & Bicudo, 2005). Organic matter generally breaks down more efficiently and completely in conditions of ready oxygen availability, largely as a result of the energy produced from the aerobic respiration (Evans, 2001). Composting is considered as a traditional way of returning organic matter to the soil. It is believed that adding considerable quantities of compost to agricultural land will reverse the trend of soil organic matter reduction, which is observed in much of the UK’s farmland, and thus benefit the soil structure, water retention, and fertility (EA 2007).

2.2.2.2 Anaerobic Digestion (AD)

Anaerobic digestion is a biological process which takes place in sealed vessels under anaerobic conditions. The process converts biodegradable waste to biogas (mixture of methane and carbon dioxide), solid and liquid residues. Waste treatment by AD is essentially a controlled and accelerated decomposition process using the same micro-organisms which produce methane in landfills (Smith et al., 2001).

AD is particularly suitable for the treatment of wet biodegradable wastes. Feedstock for AD include sewage sludge, animal manures, agricultural wastes, organic wastes from industry (mainly from food processing) and the biodegradable fraction of household waste (Smith et al., 2001; afbi, 2007; Last, 2006). Woody, lignin-rich waste components, e.g., tree and shrub clippings, bark, sawdust and shavings, and straw cannot be fed into the anaerobic process stages in large amounts (Fricke et al., 2005).

The main outputs of the AD process are biogas, liquid and solid residue (solid residues are separated by dewatering, in order to obtain solid material for composting or landfilling (Fricke et al., 2007)). Source separated waste is essential for higher quality solid and liquid residue. The liquid from the process is generally disposed of to the municipal wastewater treatment plants, even though it is rich in nutrients and could produce a liquid fertilizer (if it is of high enough quality). Liquid produced from residual waste requires further treatment (Fricke et al., 2007). The solid residue after a period of aerobically maturation by composting could be used for agricultural or horticultural applications. The biogas produced is used for heat or electricity production, potentially displacing fossil-fuels (Smith et al., 2001; Last, 2006; EA, 2007).

AD provides opportunities to produce renewable energy from organic materials in decentralized sites (AD-Nett, 2005). Within agriculture, AD is currently carried out on-farm, or in larger centralised AD plants. On-farm AD has potential to produce biogas to be used within the farm and to assist in managing the plant nutrients in slurry (afbi, 2007). Figure 2.2 shows an anaerobic digester on a dairy farm which processes slurry and supplies the farmhouse with gas for heating and cooking (from www.energyfarming.co.uk/technologies.asp).

Figure 2.2 On-farm anaerobic digestion plant.

At the moment there are uncertainties over the economics and practical applications of AD to treat MSW, as traditionally the method was used for the treatment of homogenous waste streams, such as sewage sludge. AD of biodegradable MSW would have to be depended on source separation of the biodegradable part, or pre-processing of the waste. However, contamination of the final product would be difficult to avoid and thus marketing problems with the end products are expected (Last, 2006). The liquid and solid outputs are regarded as waste in the UK, thus limiting the markets for these materials (EA, 2007).

AD involves higher investment costs in comparison to aerobic treatment plants, and it is considered to be less stable in operation. Moreover, operational problems are more difficult to remedy, once they have occurred. One important source of operational problems is the nitrogen compounds, which enter the process with the feed material. Even though nitrogen is necessary for the formation of new biomass and contributes to the stabilisation of the pH value in the reactor, different nitrogen compounds, such as ammonium, dinitrous oxide as well as nitrite and nitrate, can inhibit the biological process or make it difficult to comply with the corresponding emission limit values (Fricke et al., 2007).

2.2.2.3 Mechanical Biological Treatment (MBT)

MBT is used for the treatment of residual municipal waste, unsorted waste or any other biodegradable waste unfit for composting or anaerobic digestion in order to stabilise and reduce the volume of waste (EC, 2001).

The MBT process is normally considered as a treatment prior to landfilling. The process originally involved a simple combination of mechanical preparation, material separation and composting of the remaining waste to produce a more stabilised residue (Heerman, 2003). Nowadays, the MBT process maybe quite complex and the advanced MBT plant can be tailored to meet specific requirements and handle a varying waste stream (ASSURRE, 2006). The MBT process may involve recycling of the recyclable fraction of the residual waste, such as plastics, metals and glass, recovery of materials which are suitable for energy recovery from waste, and biological treatment (in-vessel composting and/or AD) of the biodegradable part (ASSURRE, 2006; Last, 2006). Figure 2.3 shows a modern MBT facility.

Figure 2.3 MBT plant.

The MBT of MSW before landfilling may result in minimisation of landfilled masses and volume, inactivation of biological and biochemical processes thus avoiding landfill-gas and odour emissions, and immobilisation of pollutants in order to reduce leachate contamination (Fricke et al., 2005). On the other hand, the landfill of residue is considered as biodegradable waste, and thus it should be subject to landfill tax. The recyclate derived from a mixed residual stream is considered of low value (Last, 2006).

MBT processes should be regarded as a part of a larger waste treatment strategy, and not as a complete solution, as waste treated by MBT will require additional treatment or further disposal (Heerman, 2003; EA, 2007). The use of the bio-material output can be very limited as it is regarded as waste in the UK. The contribution of MBT to the diversion of biodegradable waste from landfill is considered limited and difficult to demonstrate (EA, 2007).

2.3 Composting of bio- and vegetable waste

2.3.1 Definition of biowaste and vegetable waste

The term ‘biowaste’ was used in the 2nd draft of the Working Document on Biological Treatment of Biowaste (EC, 2001) to describe any biodegradable waste. Biodegradable waste was defined in the landfill directive (CEC, 1999) as the waste that is capable of undergoing anaerobic or aerobic decomposition, such as food and garden waste, paper and paperboard waste. Within the Working Document on Biowastes and Sludges (EC, 2003b) biowaste is meant to be the biodegradable fraction of MSW, containing food and garden waste, paper and paperboard waste. Hartl & Erhart (2005) used the term biowaste to describe source-separated biodegradable waste, which included household waste and yard trimmings. The compost quality protocol (Anon., 2007) defines biowaste as source-segregated biodegradable waste. Within this thesis biowaste means the source-separated biodegradable fraction of MSW, containing food and garden waste, paper and cardboard waste.

The term ‘vegetable waste’ can be used to describe the plant-tissue waste generated on agricultural land. It can include the non-edible parts of plant waste which are discarded during collection, handling, transportation and processing (Chang et al., 2006). Vegetable waste is differentiated from green waste, which includes the biodegradable waste from gardens and municipal parks (Hogg et al., 2002).

2.3.2 Composting methods

The variety of composting technologies is extensive as composting can be carried out in private gardens (home composting) to advanced, highly technological centralised plants (Crowe et al., 2002). The most appropriate composting method will primarily depend on land availability, the nature and quantity of biodegradable waste to be treated, financial considerations and available workforce, the desired quality of the end product, and the amount of the time available for processing (Evans, 2001).

A lot of confusion exists about naming the different compost treatment options (Crowe et al., 2002). In general the composting methods can be divided into two main categories: with and without forced aeration. In relation to the principles of feeding and turning schemes, composting without forced aeration involves static and batch-wise composting, whereas composting with forced aeration can be static and batch-wise composting or continuously fed and agitated. Agitated piles are continuously moved and thus continuous feeding is possible. If the materials are set up at the same time the feeding is batch-wise (Crowe et al., 2002).

Centralised composting processes for municipal waste management primarily fall into two categories: windrow composting and in-vessel composting (Last, 2006). On-farm composting usually involves types of static pile and windrow composting, simple in-vessel composting techniques, or vermicomposting (Misra et al., 2003). Below are summarized the main types of composting systems employed for the composting of biowaste and vegetable waste.

2.3.2.1 Static pile

Historically composting has been undertaken in large heaps. These heaps need to be designed and sized so as to allow adequate air flow otherwise anaerobic processes occur, and odour problems arise (EA, 2007). Static piles are not turned, or only infrequently, a few times per year (Evans, 2001). To aerate the pile, layers of loosely piled bulking agents (e.g. wood chips) are added so that air can pass from the bottom to the top of the pile. The piles can also be placed over a network of pipes (forced aeration system), which deliver air into or draw air out of the pile (Misra et al., 2003; USEPA, 2006a). Passive aeration systems can also be used. Perforated, ends-open pipes are embedded in each pile, and as a result, air flows into the pipes and through the pile because of the chimney effect created as the hot gases rise upward out of the pile (Misra et al., 2003). Figure 2.4 presents a passive aerated static pile.

The method works well for moderate quantity generators, such as local governments, landscapers, or farms (Evans, 2001; USEPA, 2006a). Static piles are suitable for a relatively homogenous mix of organic waste, such as vegetable residues, yard trimmings and biowaste. This method, however, does not work well for composting animal by-products or grease from food processing industries (USEPA, 2006a).

Figure 2.4 Static pile composting - passive aerated system.

2.3.2.2 Windrow

This method is suitable for large quantities of diverse biodegradable waste, such as waste generated by communities and collected by local governments, and high volume food-processing businesses (USEPA, 2006a). Organic waste is placed into long piles called windrows and aerated by turning the pile periodically by either manual or mechanical means. The equipment used for turning determines the size and shape of the windrow. A number of specialized machines have been developed for turning the windrows. Some of these machines are designed to attach to farm tractors or front-end loaders, others are self-propelled (Evans, 2001; Misra et al., 2003; USEPA, 2006a). Figure 2.5 shows the turning of the windrow by a self-propelled turning machine.

Suitable waste for windrow composting includes yard trimmings, grease, liquids, and animal by-products, but only with frequent turning and careful monitoring (USEPA, 2006a). However, in the EU the implementation of the animal by-products regulations (ABPR) (CEC, 2002b) mandates in-vessel composting for catering and animal by-products (Boulos et al., 2006).

Figure 2.5 Windrow composting: turning the pile by a self-propelled machine.

Windrow composting may be employed for small or moderate scale operations; however, in principle windrow composting is a large scale operation and might be subject to regulatory enforcement. Samples of the compost should be tested in a laboratory for bacterial and heavy metal content (USEPA, 2006a). The high land requirement of windrow composting is a significant drawback of the method. The leachate, which is released during the composting process, can contaminate local ground-water and surface-water supplies and thus, should be collected and treated. Odours, and bioaerosols released during the regular periods of turning, also need to be controlled (Evans, 2001; USEPA, 2006a).

2.3.2.3 In-vessel

In-vessel composting means the composting of biowaste in a closed reactor where the composting process is accelerated by an optimised air exchange, water content and temperature control (EC, 2001).

In-vessel methods rely on a variety of forced aeration and mechanical turning techniques to speed up the composting process. Many methods combine techniques from the windrow and aerated pile methods in an attempt to overcome the deficiencies and exploit the attributes of each method. There are a variety of in-vessel methods with different combinations of vessels, aeration devices, and turning mechanisms (Misra et al., 2003). In-vessel type of facilities includes static container, tunnel, drum and tower composting (Crowe et al., 2002).

Figure 2.6 In-vessel composter - static container.

In-vessel composting can process large amounts of waste without taking up as much space as the windrow method. In addition, it can accommodate virtually any type of organic waste (e.g. meat, animal manure, biosolids, food scraps). In-vessel composting (Figure 2.6) produces very little odour and minimal amount of leachate. Although this method is expensive and requires high levels of technical assistance, it uses less land and manual labour than windrow composting (USEPA, 2006a).

2.3.2.4 Vermicomposting

The term ‘vermicomposting’ means the processing and biodegradation of organic wastes primarily by appropriate worm species, in addition to micro-organisms (BSI, 2005). Vermicomposting can be used in home composting (USEPA, 2006a) or on-farm (Misra et al., 2003). Vermicomposts, unlike the traditional composts, are products of non-thermophilic biodegradation, but they are produced from the interactions between earthworms and microorganisms (Arancon et al., 2004). In vermicomposting, the earthworms take over both the roles of turning and maintaining the material in an aerobic condition, thereby reducing the need for mechanical operations. The end-product (vermicompost) (Figure 2.7) is derived from the worms' excreta (castings).

Figure 2.7 Vermicompost.

Earthworms can consume practically all kinds of organic matter typically placed in a compost pile, such as food scraps, paper, plant residues, and they can eat their own body weight per day; thus, for example, one kilogram of worms can consume one kilogram of residues every day. The castings are rich in nitrate, available forms of phosphorus, potassium, calcium and magnesium. The passage of soil through earthworms promotes bacterial and actinomycetes growth. The major drawback of the vermicomposting process is that the temperature is not high enough for an acceptable pathogen kill (unlike the thermophilic composting methods), as it must be maintained at less than 35 °C during the vermicomposting processes (Misra et al., 2003).

Earthworms are sensitive to variations in climate and care needs to be taken to keep the worms alive and healthy by providing the proper conditions and sufficient food. Maintenance procedures mainly include preparing bedding and separating worms from their castings (USEPA, 2006a).

2.3.3 Legislation and recommendations for composting

2.3.3.1 Current framework

The EU aiming to meet the challenges of today's environmental problems has developed the Environment Action Programmes. The 6th Environment Action Programme, entitled 'Environment 2010: Our future, our choice' covers the period from 1/1/2001 to 31/12/2010 and includes the following seven thematic strategies: air pollution, the marine environment, sustainable use of resources, prevention and recycling of waste, sustainable use of pesticides, soil protection, and urban environment (COM, 2001).

The strategy on the prevention and recycling of waste aims at reducing the pressure on the environment caused by waste production and management. The main advancement of the strategy is related to amending the legislation to improve implementation, and preventing waste and promoting effective recycling. As far as the waste recycling is concerned the strategy aims to promote the recycling sector in order to reintroduce waste into the economic cycle in the form of quality products, whilst minimizing the environmental impact of doing so (COM, 2005). The strategy places particular emphasis on biodegradable waste and the main driver is the Landfill Directive (COM, 2003).

The Landfill Directive (CEC, 1999) is the main driver for the management of biodegradable waste. The implementation of the Landfill Directive sets demanding targets to reduce the amount of biodegradable municipal waste landfilled. In particular, the biodegradable municipal waste landfilled should be reduced to 75% of that produced in 1995 by 2006, to 50% by 2009, and to 35% by 2016. For countries that landfilled in 1995 over 80% of their biodegradable municipal waste, such as the UK, a four-year extension to each of the above mentioned deadlines was provided.

Depending on local conditions, such as climatic conditions to the composition of the collected biodegradable waste, composting can be the environmentally best option for the management of biodegradable waste (EC, 2007). Composting is strongly supported in the UK as an important component of meeting the Waste Strategy (DETR, 2000) targets under the landfill directive, for recycling and composting of biodegradable waste (DEFRA, 2006b).

A separate Directive on biodegradable waste is currently not available. However, guidelines for the management of municipal biodegradable waste are under development (EC, 2007). Currently available guidelines were proposed in the 2nd draft of the working document (WD) on biological treatment of biowaste (EC, 2001), which includes standards designed to regulate potentially harmful aspects of compost production and use and complementary standards governing environmental and health aspects of application to land. For the utilization of compost as fertilizer or soil conditioner within organic farming (Eco-agric), specific compost quality standards are also provided (EnviroWindows, 2005). In the UK, compost quality standards are provided by the Publicly Available Specification for composted material BSI PAS 100, developed by the British Standards Institution and certified by the Composting Association (BSI, 2005).

Furthermore, standards (quality assurance systems) designed to give confidence to consumers, through quality assurance, as well as clear applications for specific market outlets, are developed. Such quality assurance systems are necessary so that markets for compost can develop. These systems include specific compost quality standards for specific end-users and quality symbols to facilitate marketing (Hogg et al., 2002; EC 2007). At the EU level, the Eco-label (CEC, 2006) constitutes a quality assurance system for using composts as soil improvers and growing media. In the UK, the compost quality protocol (Anon., 2007) is a quality assurance system providing standards for using compost, which meets the requirements of the BSI PAS 100: 2005, within land restoration and soft landscape operations, horticulture, or agriculture and soil-grown horticulture.

Specific regulations for the treatment of animal by-products in a composting plant were provided by the Animal By-Products Regulation (ABPR) (CEC, 2002b), which regulates the collection, transport, storage, handling, processing and use or disposal of animal by-products, including catering waste. These regulations are incorporated within the compost quality standards developed after 2002.

2.3.3.2 Compost quality standards for land application

Compost quality is highly influenced by the input materials. Specifications for the allowed input materials, which are provided by the 2nd draft of the WD on biological treatment of biowaste (EC, 2001), the Eco-label (CEC, 2006), and the compost quality protocol (Anon., 2007), include both vegetable waste and biowaste (as defined in this thesis). Within the Eco-agric (EC 2092/91 - EC 1488/97) only composted source-separated household waste containing only vegetable and animal waste is accepted.

Standards, which are related to the composting process aiming to ensure compost sanitisation, are provided by the 2nd draft of the WD on biological treatment of biowaste (EC, 2001) and the PAS 100 (BSI, 2005) and they are presented in Table 2.1.

Table 2.1 Composting process standards for compost sanitisation (EC, 2001; BSI, 2005).

	Composting process	Temperature (°C)	Moisture (w/w)	Time (weeks)	Mixing/turning
EC 2001	windrow	≥ 55	n.a.	2	5
	windrow	≥ 65	n.a.	1	2
	in-vessel	≥ 60	n.a.	n.a.	n.a.
BSI 2005	n.a.	≥ 65	≥ 50 %	1	≥ 2

In sanitised compost *Salmonella spp* should be absent in 50 g of compost, and *Clostridium perfringens* in 1 g of compost (EC, 2001). However, under the ABPR, it is regulated that *Salmonella spp* should be absent within 25 g of fresh compost, and this limit is adopted within the Eco-label (2006) and BSI PAS 100 (2005) standards. *Escherichia Coli* should not exceed the 1000 CFU in 1 g of fresh compost (CEC, 2006; BSI, 2005). For composts produced exclusively from green, garden or park waste also *Helminth ova* should be absent in 1.5 g of compost to be awarded the Eco-label (CEC, 2006).

Maximum allowable concentrations of heavy metals in composted material are always included within compost quality standards. Table 2.2 presents the maximum concentration of heavy metals allowed in composts as proposed in the 2nd draft of the WD on biological treatment of biowaste, the EU Eco-label, the EU Eco-organic, and the BSI PAS 100. Compost of Class 1 can be applied to land according to the best agronomic practice without any specific restriction, whereas compost of Class 2 can be used in a quantity not exceeding 30 Mg DM ha^{-1} on a 3-year average (EC, 2001).

Table 2.2 Maximum concentration of heavy metals (in mg per kg dry matter (DM) compost) allowed in composts as proposed in the 2nd draft of the WD on biological treatment of biowaste (Class 1 and 2), the EU Eco-label, the EU Eco-organic, and the BSI PAS 100.

Parameter			**Compost**		
(mg kg^{-1} DM)	**Class 1**	**Class 2**	**Eco-label**	**Eco-agric**	**BSI PAS 100**
Cd	0.7	1.5	1	0.7	1.5
Cr	100	150	100	70	100
Cu	100	150	100	70	200
Hg	0.5	1	1	0.4	1
Ni	50	75	50	25	50
Pb	100	150	100	45	200
Zn	200	400	300	200	400

In Germany, the heavy metal concentrations are standardised to a specific level of organic matter (30%) in order that the influence of compost maturation on the concentration of heavy metals is properly assessed, as organic matter is mineralized over time but heavy metals may accumulate in soils (Hogg et al., 2002).

Compost content of organic and physical contaminants, pathogens and weeds may also be included. Limits for organic contaminants were proposed only within the 2nd draft of the WD on biological treatment of biowaste, concerning the polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs) only, and their concentration was set to be in consistence with the Sewage Sludge Directive (86/278/EEC). In general, organic contaminants are expected to be at low levels in composts derived form source separated materials (Hogg et al., 2002; Amlinger et al., 2003), and therefore in most European countries there are no set limit values for organic contaminant in composts.

Physical contaminants with mesh size > 2 mm, such as glass, plastic, metal, should be typically lower than 5% of the dry matter mass of composts. No more than 5% of compost (DM basis) should include stones > 5mm (EC, 2001). Stones > 4 mm should not exceed the 8% or 16% of compost DM in grades other than mulch, and in mulch, respectively (BSI, 2005).

Weed seeds allowed per litre of compost range from 3 (EC, 2001) to 0 (BSI 2005). Further compost quality standards include compost maturity and plant response (emergence and growth) (BSI, 2005; CEC, 2006), and compost physical and/or nutrient related properties (CEC, 2006).

2.3.3.3 Rules for compost application to agricultural land

The land application of compost is supported within the soil thematic strategy (Van-Camp et al., 2004) as a measure to combat soil organic matter's decline. Within the 2003 CAP reform the reinforced cross compliance includes respect of standards of good agricultural and environmental conditions referring to protection of soil from erosion and maintenance of soil organic matter and soil structure (EC, 2003a). However, compost land application has to respect environmental parameters. In field applications, compost heavy metals and nutrient content need to be taken into account as covered by fertilizer, soil protection and water laws in each country (Hogg et al., 2002).

In the 2nd draft of the WD on biological treatment of biowaste it is proposed that compost of class 1 should be used according to the best agronomic practice without any specific restriction. Compost of class 2 should be applied without exceeding a quantity of 30 tonnes dry matter per hectare on a three years average (EC, 2001). However, every Member State is expected to decide on lower or higher maximum allowable quantities than those mentioned within the WD for land application (EC, 2001). The EU Nitrates Directive (CEC, 1991) limits the amount of total N which can be applied to agricultural land by manures (composts included) in order to reduce water pollution by nitrates from agricultural sources and prevent further pollution, and establishes a maximum limit of 170 kg total N per hectare per year in Nitrogen Vulnerable Zones (NVZ). Also, under the Nitrates Directive, codes of good agricultural practice are required to be developed by the Member States in order to cover activities as application periods, fertiliser use near watercourses and on slopes, manure storage and spreading methods, and other land management measure.

In the UK, compost land application needs to abide by the rules set within the codes for good agricultural practice to protect soil, water and air. The application of livestock manures or other organic wastes (including composts) is limited at a maximum of 250 kg ha^{-1} of total nitrogen in any one year. As an alternative in catchments less sensitive to nitrate leaching, some non-livestock wastes, such as sewage sludge cake or composted organic waste, which contain very little plant available nitrogen, may be applied up to 500 kg ha^{-1} of total nitrogen in one application every two years (MAFF, 1998a; MAFF, 1998b). However, in NVZs the compost application to non-grassland should not exceed the 170 kg of total N ha^{-1}, at farm size applications (DEFRA, 2002).

Compost with highly available N should not be applied to sandy or shallow soils between September to November, or August to November for non-autumn sown crops (MAFF, 1998b). Compost land application must be avoided: i) when the ground is waterlogged, flooded, frozen hard, or snow covered, ii) to sloping fields, and iii) within a distance of 10 m from watercourses (MAFF, 1998b). Compost must be spread uniformly on the soil. The fertilizer value of composts is particularly important, especially in NVZ, as crop N requirements should not be exceeded (MAFF, 1998b). Summarising the above it can be concluded that the fertilizer replacement value of composts and the total N supplied is essential information for compost use in agriculture under the codes of good agricultural practice.

2.3.4 Composting in the UK - quantities

The survey on the state of composting and biological waste treatment in the UK for the financial year 2004/05 (Boulos et al., 2006) which was carried out by the Composting Association with the support of WRAP and the Environment Agency, shows that composting in the UK is growing significantly. The amount of organic waste composted in the UK in 2004-05 was estimated at 2.67 million tonnes, increased at a rate of about

35% the amount composted in 2003-04, and 145% the amount composted in 2000-01. Table 2.3 presents the quantities and types of organic waste composted in the UK in 2004-05, as estimated in the survey.

Table 2.3 Estimated quantities and types of organic waste composted in the UK in 2004/05 based on producer returns (After Boulos et al., 2006).

Types	**Quantities**	
	Estimated total collected (million tonnes)	**Total percentage (%)**
Municipal household		
Garden waste	1.906	71.4
Garden and kitchen waste	0.128	4.8
Kitchen waste	0.003	0.1
Schools, HEI or colleges green waste	0.006	0.2
Other municipal household waste	0.135	5.0
Total	**2.178**	**81.5**
Municipal non-household		
Council park and gardens waste	0.051	1.9
Council - collected food processing by-products	<0.1	0.0
Council - collected food waste from shops/ catering outlets	<0.1	0.0
Other municipal non-household waste	0.012	0.4
Total	**0.063**	**2.4**
Non municipal waste (commercial)		
Landscape/ grounds maintenance	0.135	5.0
Forestry/ timber/ bark/ by-products	0.057	2.2
Food processing by-products	0.093	3.5
Food waste from shops/ catering outlets	0.002	0.1
Paper and cardboard	0.002	0.1
Sewage sludge	0.007	0.3
Paper sludge	0.001	0.1
Manures/ straw	0.003	0.1
Other	0.131	4.9
Total	**0.431**	**16.1**
Total composted	**2.672**	**100.0**

From the total amount of organic waste composted in 2004-05, 66% was processed by dedicated composting facilities, whereas on-farm processing represented the 13%. The compost produced was used in horticulture, landscaping, sport turf applications, landfill engineering and agriculture. Agriculture received 26% of the total amount of compost produced (Boulos et al., 2006).

2.4 Land application of biowaste and vegetable waste composts

2.4.1 Introduction

Compost application to agricultural soil needs to maximise agronomic benefits while protecting soil and water quality. The main determinant for efficient agronomic use is nitrogen availability. High nitrogen utilization in agriculture from mineral fertilizers is well established and understood, whereas increasing the nitrogen use efficiency of organic fertilizers requires further investigation (Gutser et al., 2005).

Most existing research has focused on the utilization of animal manure and sewage sludge in agriculture, whereas the investigation for biowaste and vegetable waste compost utilization in agriculture is limited. The paragraphs below give an overview of the pertinent literature concerning compost application to agricultural soil, with particular regards to crop production, soil properties, nitrogen dynamics, and a brief of the economic implications involved.

2.4.2 Effect on crop production

Iglesias-Jimenez & Alvarez (1993) showed that biowaste compost application at rates up to 50 Mg ha^{-1} increased ryegrass yield under greenhouse conditions, although not at the same levels as with mineral fertilization. Montemurro et al. (2005) demonstrated that the application of relatively lower levels of MSW compost to sunflower production in Italy resulted in similar oil and protein yield performance as mineral fertilization.

Clark et al. (2000) showed in a 3-year field experiment in Florida that the incorporation of MSW compost into sandy soils under drip-irrigation provided improved vegetable growth and yield. Work undertaken by Svensson et al. (2004) during a 4-year field experiment in Sweden indicated that biowaste compost should not be used as a sole fertilizer, but it should be complemented with mineral N for optimum crop yield, unless very high rates of application are used. However, high rates of compost application may increase N loss potential by leaching (Mamo et al., 1999).

The low fertilizer value of composts is reported for both biowaste and vegetable waste composts (Båth & Rämert, 2000; Sikora & Enkiri, 2001; Nevens & Reheul, 2003). The main reason for this is found to be the low release of compost N following compost application and hence the low plant N availability (Svensson et al., 2004). However, the longer-term effects of repeated compost application on crop yield are better. Leroy et al. (2007) showed increased maize yields in the last years of a 7-year fruit, vegetable and garden waste compost in Flanders. The increase was attributed to stimulation of the soil food web, through cumulative applications, and thus enhance N turnover. Mamo et al. (1999) found that adequate maize grain yield was achieved in the 3rd year of consecutive biowaste compost application, without any mineral fertilizer addition to be necessary, in a field experiment in Minnesota. Compost application also influences soil properties (see paragraph 2.4.3), thus improving the growing conditions for the crop and consequently the crop yield.

2.4.3 Effect on soil properties

Compost application to agricultural soil has been reported to increase soil organic matter, improve soil fertility, conserve soil moisture, control weeds and suppress disease.

2.4.3.1 Soil chemical properties

The application of biowaste and vegetable waste composts increases soil organic matter and total N content (He et al., 1992; Crecchio et al., 2001; Nevens & Reheul, 2003; Hartl & Erhart, 2005). Hadas et al. (2004) showed that the application of MSW compost as mulch at a rate of 43 Mg ha^{-1} resulted in an increase of soil organic matter at a rate of 21% of the organic matter applied by the compost in 3 years. Increase in the quantity and quality (humic acids) of soil organic matter were shown in both sandy (Weber et al., 2007) and clay soils (Melero et al., 2007).

Biowaste and vegetable compost application can increase plant available P, K (Hartl et al., 2003; Martínez et al., 2003) and Mg (Parkinson et al.,1999; Weber et al., 2007) levels of soils, and soil CEC (Bartl et al., 2002; Weber et al., 2007).

Increased levels of soil EC (Stamatiadis et al., 1999; Madejón et al., 2001) due to the application of mixed green and animal waste compost, and also vinasse compost have also been reported. Although the increase of EC was not found capable of causing a sodium hazard to the soil, it indicates potential problems following the repeated application of compost to agricultural soil.

Compost application can ameliorate soil acidity (Wong et al., 1998; Van De Berghe & Hue, 1999), contribute towards the reclamation of sodic soils (Kochba et al., 2004), and prevent the acidification effect of mineral fertilization (Stamatiadis et al., 1999).

2.4.3.2 Physical properties

Biowaste and vegetable compost application to agricultural soil is shown to improve soil physical properties. This improvement is mainly a result of the organic matter addition to the soil by the compost application.

Jakobsen (1995) found that after 8 years of different types of compost application to sandy clay soil the soil structure was improved, especially when the compost was applied on the soil surface. The surface was thereby protected from the compressing effects of rain drops and the rapid drying. Soil infiltration rate was also found to be higher. Contradictory results were found by Stamatiadis et al. (1999), who demonstrated that after 1 year of compost application to silt clay loam soil, the infiltration rate decreased. Pandey & Shukla (2006) showed that compost addition to sandy soil resulted in higher retention of rainfall, if application levels are sufficiently high.

Urban waste compost application to the soil was shown to increase soil total porosity (Guisquiani et al. 1995) and aggregate stability (Aggelides & Londra, 2000; Annabi et al., 2007). Increased soil porosity, field water capacity and the amount of plant available water, but only in the short time after MSW compost application were demonstrated by

Weber et al. (2007). Johnson et al. (2006) showed that surface application of dairy manure compost increases water retention capacity and decreases soil bulk density. These results are in accordance with findings by Aggelides & Londra (2000) who used a composted mixture of MSW, sewage sludge and sawdust. Aggelides & Londra (2000) also showed that compost application can reduce the penetration resistance of the soil.

2.4.3.3 Biological properties

The application of compost to agricultural soil is shown to reduce the number of parasitic nematodes, and increase both the numbers of micro-arthropods and earthworms (Leroy et al., 2007). Stimulation of soil biological activity and increase of micro-arthropods were also demonstrated by Petersen et al. (2003). Melero et al. (2007) showed a clear increase of microbial biomass and enzymatic activities at the fourth year of compost application to clay soils.

Composts have the potential to provide biological control of many soil-borne plant diseases. Foliar, vascular, and root pathogens may be affected by compost application (Hoitink et al., 1997; Noble & Coventry, 2005; Yogev et al., 2006). Reported levels of disease suppression vary, even if similar composted material is used at the same rates. Sterilisation of composts generally results in a loss in the disease suppressing capability of composts, indicating that the mechanism is predominantly biological, although chemical and physical factors have also been implicated. The mechanisms and antagonistic micro-organisms involved in disease suppression need further investigation (Bailey & Lazarovits, 2003; Noble & Coventry, 2005).

The use of compost as a mulch in an orchard ecosystem was shown to be beneficial to management of weed, fungal, and insect pests (Brown & Tworkoski, 2004). Other experiments have shown that compost application can benefit weeds. Blackshaw (2005) demonstrated that the gradual N release from manure and compost over years appeared to have more positive effect on weeds rather on the growing crop. Boyhan et al. (2006) found that compost was not very affective for weed control in onions. A study by Law et al. (2006) indicated that shallow cultivation following transplanting, combined with midseason compost mulch application, can result in high yields in an organically managed bell pepper system, which was comparable to yields of most varieties grown under conventional practises.

2.4.3.4 Heavy metals and other potential toxic elements (PTE)

Soil contamination by heavy metals or other potential toxic elements is often of concern, especially with high loads of MSW composts application to the soils (He et al., 1992).

Madrid et al. (2007) applied MSW compost at moderate rates on a sandy loam soil for almost three years. They found that soil heavy metal content increased compared to the control treatment and their results indicated that metals applied by compost were more available than native metals in soil. Research undertaken by Wang et al. (2003) showed that after a year of yard waste compost application at a high rate, the leaching potential of a gravely loam soil in respect of microelements (Zn, Cu, Cd, Cr, Pb, As, Ni, Fe, Mo and Co) was found at similar levels as the control.

The application of 32 Mg ha^{-1} of biowaste compost to a sandy to loamy silty soil for 5 years was found not to influence the total soil contents of Cd, Mn, Mo and Ni, whereas the Zn and Pb contents were higher than the control. The crop contents of Zn and Cu were found higher for both the biowaste compost and the mineral fertilizer treatments, in comparison to the control (Bartl et al., 2002). Martínez et al. (2003) applied biowaste compost (40, 80 and 120 Mg ha^{-1}) to a degraded semi-arid ecosystem, in the first year of a 3 years field experiment. They found that the biowaste application did not increase the levels of Cr, Cd, Pb and Ni in the soil. Even though the levels of Zn and Cu were found increased, especially for the higher rates, the highest concentrations of both metals were far below the levels considered phytotoxic. Petersen et al. (2003) demonstrated that a 3-year household compost application to loamy sand and sandy loam soils resulted neither in accumulation of organic contaminants, such as PAH, LAS, and DEHP, in soils, nor in plants.

2.4.4 Effect on N dynamics

Knowledge of the short and long term availability of N following biowaste and vegetable waste compost application is essential in order to meet crop requirements, whilst ensuring environmental protection from excessive nitrate leaching. Increasing the N use efficiency of organic amendments and understanding the N dynamics in compost amended soils remain important issues for research (Amlinger et al., 2003; Gutser et al., 2005).

The availability and binding dynamics of total compost N in soils during the long and short term is strongly linked to the soil organic matter and the relative magnitude of the active pool and the more passive pool of the soil organic matter (Nortcliff, 1999; Amlinger et al., 2003). Compost addition to soils was shown not to increase mineralization of soil OM (Sikora & Yakovchenko, 1996). Figure 2.8 illustrates the potential fate of compost N in the soil.

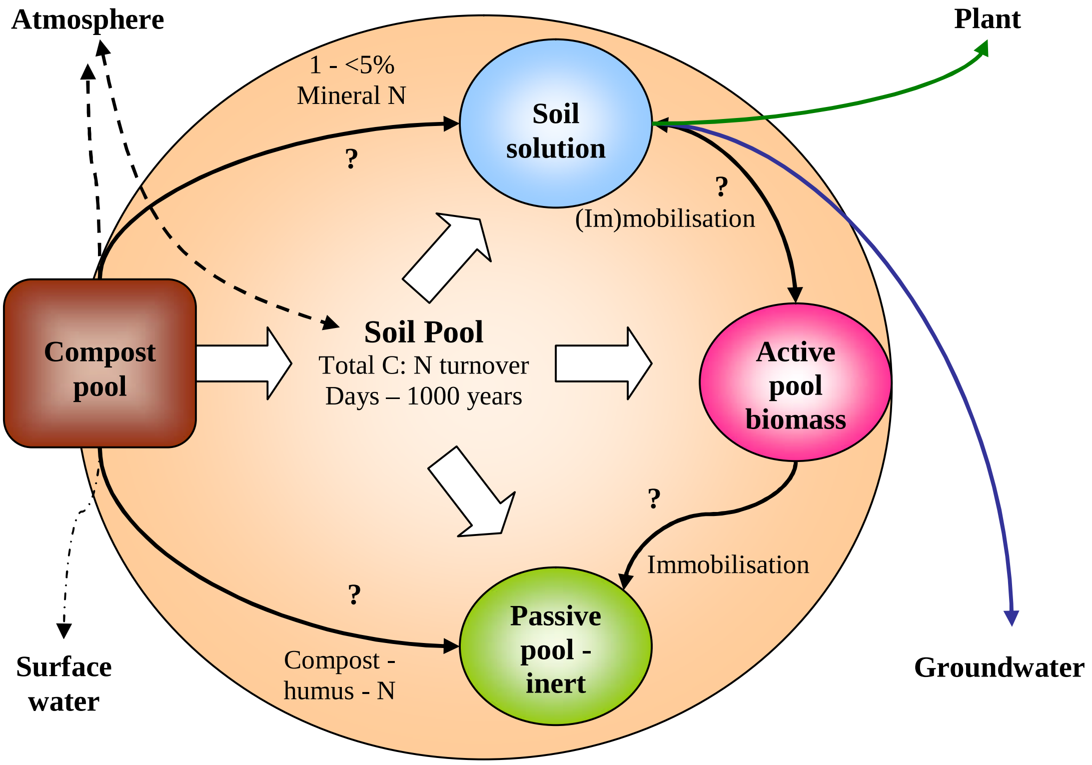

Figure 2.8 The still sparely known fate of compost N in the soil organic matter pools (Adopted from Amlinger et al., 2003).

In general, the N availability to plants is low since the majority of the total compost N is bound to the organic N pool (Amlinger et al., 2003; Gutser et al., 2005; Hartl & Erhart, 2005). The mineral nitrogen content of composts is generally low, since N is partly lost during composting due to volatilisation (Zwart, 2003).

The organic portion of compost total N which is not readily available to plants, can be mineralized, and then potentially taken up by the plants, immobilized, denitrified, volatilized, fixed within the clay minerals and/or leached. Figure 2.9 illustrates the main transformations of compost N applied to the soil.

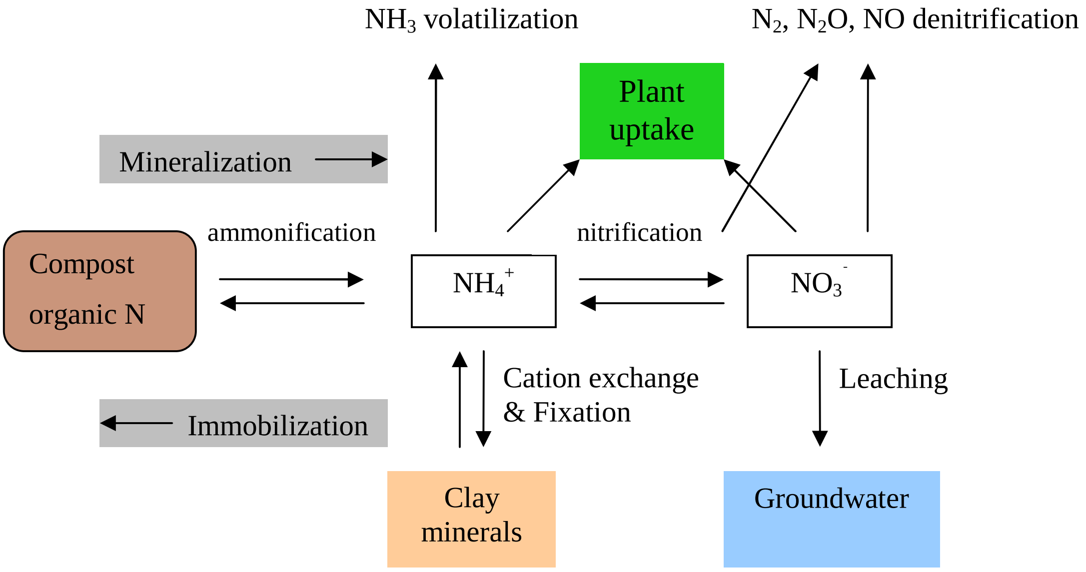

Figure 2.9 Schematic diagram of the main organic compost N transformations in the soil.

N dynamics in compost amended soils may be influenced by various factors related to compost parameters, climatic conditions, crop types, soil properties and soil management practises (Amlinger et al., 2003). The main N transformations following compost application to soil are summarized below.

2.4.4.1 Mineralization and immobilisation

Mineralization of compost N involves the conversion of organic forms of N (e.g. amino-sugars and amino-acids) to ammonia (NH_3) or ammonium (NH_4^+) and nitrate (NO_3^-).

The mineralization process involves the ammonification and nitrification. During ammonification the organic N is hydrolysed to NH_4^+ as shown in Equation 2.1.

$$\text{Organic–N} \longrightarrow NH_3 \xleftrightarrow{\pm H^+} NH_4^+ \qquad (2.1)$$

The subsequent conversion of NH_4^+ to NO_3^- is called nitrification and it is carried out in two steps mainly through two groups of bacteria (*Nitrosomonas* and *Nitrobacter*). The NH_4^+ is first converted to nitrite (NO_2^-) as shown in Equation 2.2 and then (NO_2^- is further oxidised to NO_3^- (Equation 2.3). The process of nitrification is generally fast and any NH_3 produced due to ammonification is rapidly nitrified to NO_3^- (Benbi & Richter, 2003).

$$2\ NH_4^+ + 3\ O_2 \xrightarrow{\textit{Nitrosomonas}} 2\ NO_2^- + 2\ H_2O + 4\ H^+ \qquad (2.2)$$

$$2\ NO_2^- + O_2 \xrightarrow{\textit{Nitrobacter}} 2\ NO_3^- \qquad (2.3)$$

N mineralization always occurs simultaneously with N immobilisation in the soil, which operates in the reverse direction (Figure 2.9), with the soil microbial biomass assimilating inorganic N forms. The relative magnitudes of N mineralization and immobilization determine whether the overall effect is net mineralization or net immobilisation (Bear, 1964; Cabrera et al., 2005).

Whether N is mineralized or immobilized it depends primarily on the C: N ratio of the material undergoing decomposition by the soil microorganisms (Wild, 1988; Tisdale et al., 1999; Benbi & Richter, 2003). The soil application of organic fertilizers which consist of low N organic substances (C: N ratio >15), can result in limited N availability due to immobilization (Gutser et al., 2005). Organic materials with C: N ratio > 25 are expected to result in net immobilization of native or added inorganic N (Sims, 1990). Iglesias-Jimenez & Alvarez (1993) showed that composts with C: N ratio < 12 can result in net N mineralization.

The contribution of compost organic N towards the N mineralized following the application of different types of composts and different C: N ratios to poor acidic, clay, and sandy loam soils, was found to range from negative to lower than 12% (Beloso, 1993; Gagnon et al., 1998; Gagnon & Simard, 1999; Mamo et al., 1999).

Compared to sewage sludge, biosolids, manures, or other non-composted organic amendments, compost application results in lower N mineralization rates (Petersen, 2003; Gutser et al., 2005).

Compost N transformations also depend on the contents of indigenous mineral N and organic C in soils. Han et al. (2004) demonstrated that a combined application of compost with chemical fertilizer could improve the compost use efficiency by increasing mineralization of compost N, especially in soils with low mineral N content. Mineralization of compost N also depends on temperature. Chodak et al. (2001) showed that the increase of temperature increased net release of nitrate nitrogen, whereas it did not influence the ammonium nitrogen. Cumulative net release of mineral N after 112 days of incubation ranged from 2% of the total N content at 5 °C to 16% at 25 °C, for a MSW compost with a C: N ratio of 15.4. Soil moisture content is shown not to influence compost N mineralization (Benitez et al., 1998), with significant declines only under extreme wet or dry conditions (Amlinger et al., 2003). Kruse et al. (2004) showed that N mineralization was higher under constant water content, than under repeated wet and dry conditions.

2.4.4.2 Plant N uptake

Nitrogen is essential for plant growth as it is a constituent of all proteins and nucleic acids. The usual sources of N to plants are NH_4^+ and NO_3^- (Wild, 1988). The process of uptake involves the transport of water soluble species to the roots followed by their absorption across the root surface (Benbi & Richter, 2003).

The N availability is the main factor determining crop production. The increase of plant N uptake increases production; however, with abundant N, dry matter and grain production does not increase further, as growth is N saturated (Lawlor et al., 2001).

The mineralization of compost N and the N availability to plants over a certain period can be evaluated via (i) laboratory incubation tests, (ii) field or pot experiments in the basis of crop N uptake, or (iii) N leaching tests with the use of lysimeters, suction cups, or measurements of soil mineral N content in the leaching horizon.

In different studies, the biowaste or vegetable waste compost N recovery by crops was found to range between 2% and 15% of the total compost N applied (Mamo et al., 1999; Amlinger et al., 2003; Nevens & Reheul, 2003; Wolkowski, 2003; Hartl & Erhart, 2005).

The response in cumulative yield and N uptake was proportional to the rate of compost (C: N ratio = 10) application in the first growing season after application (Iglesias-Jimenez & Alvarez, 1993). Eriksen et al. (1999) found no crop response to compost (C: N ratio = 40) application in the first year, but yield and N uptake were observed to increase in the second year after the application with the increase of compost application rate. In both studies it was shown that the mineralization of organic N was inadequate to meet crop growth requirements for maximum yield.

Sullivan et al. (2002) showed that a single application of food waste compost (C: N ratio ~ 20) did not affect tall fescue yield or N uptake in the first season, but increased grass yield and N uptake during the second and third seasons after application. The reduction in N fertilizer requirement during midseason growth was equivalent to 3.6-5.2% of compost N in the second year, and 2.0-3.5% of compost N in the third year after application.

2.4.4.3 Leaching

The process of N losses by the downward movement of water through the soil profile is called leaching (Benbi & Richter, 2003). Between the various forms of N, only nitrate is washed out of the soil in considerable amounts by percolating water (Wild, 1988). Nitrate leaching is generally a major N loss mechanism from field soils in humid climates and under irrigated cropping systems (Tidale et al., 1999).

N leaching can be of serious environmental concern. The increase of the amount of nitrates in surface water may contribute to the acceleration of the processes of acidification and eutrophication of surface water (Hornung, 1999). Elevated levels of nitrite may have a negative impact to river biota (microorganisms, invertebrates, fish) due to toxicity (Kelso et al., 1999).

High levels of nitrates in drinking water may result in infantile methaemoglobinaemia (blue baby syndrome), which can be fatal for young infants (Mensinga et al., 2003). Although nitrates and nitrites are not carcinogenetic, they or a chemical substance derived from them, may be involved in the generation of nitrosamines and nitrosamides, which are of toxicological importance (Walker, 1999; Mensinga et al., 2003). Another impact of high nitrate levels in drinking water could be associated with insulin depended diabetes mellitus (IDDM) (McKinney, 1999). The amount of nitrates in drinking water is limited to a maximum of 50 mg L^{-1} (91/676/EEC) in the EU, and about 44 mg L^{-1} (10 mg L^{-1} nitrate-nitrogen) in the USA (USEPA, 2006b).

Mamo et al. (1999) in a 3-year field experiment demonstrated that compost application to maize production at high rates (90 Mg ha^{-1} annually, or 270 Mg ha^{-1} once) may increase nitrate leaching from the first year of application. Li et al. (1997) applied five different compost types to sandy soil at 100 Mg ha^{-1} in leaching columns. Nitrate leaching was observed to be inversely related to the C: N ratio of the composts. The amount of N leached was found to be between 3.3% and 15.8% of total compost N applied. Eriksen et al. (1999) demonstrated that at the second year of compost application, compost N mineralization resulted in increased soil NO_3-N at 30-90 cm depths with increased compost application rate. Despite that, only the highest rate (189 Mg ha^{-1}) resulted in higher residual soil NO_3-N compared to the control.

Wolkowski (2003) showed that the application of MSW compost at rates up to 90 Mg ha^{-1} resulted in lower soil NO_3-N levels, measured in the top 90 cm, than in treatments receiving the recommended N fertilizer. Hart & Erhart (2005) demonstrated that the NO_3-N content of biowaste compost amended soil, within a 10-year field application of up to 23 Mg ha^{-1}, was at the same levels with soil receiving mineral N fertilizer, although total N loads were higher in the compost treatments and the N recovery by the crop was lower for the compost treatments. Nevens & Reheul (2003) showed that the application of 22.5 Mg ha^{-1} of fruit, vegetable and garden compost did not result in excessive amounts of residual NO_3-N, in a 4-year field experiment.

Díez et al. (1997) showed in a 3-year experiment that the application of municipal organic waste compost at a rate of 22.5 Mg ha^{-1} combined with irrigation adjusted to crop needs has the potential to control nitrate leaching losses in comparison to other fertilizers. Nitrate concentration in groundwater was measured beneath sandy terrace soil amended with spent mushrooms compost and chicken manure compost at rates of 56 and 112 Mg ha^{-1} for three years. The nitrate concentration remained below 10 ppm during the experiment for both compost treatments, but was up to 14.7 ppm for the mineral fertilizer treatment during a wet spring (Maynard, 1993).

Maynard (1994) also studied the effect of the application of MSW compost at rates of 56 and 112 Mg ha^{-1} (incorporated) and 224 Mg ha^{-1} (mulch) on nitrate leaching. Again nitrate concentration in groundwater remained below 10 ppm for all treatments. Jaber et al. (2005) showed that the NO_3-N concentration of groundwater below sandy and calcareous soils amended with biowaste compost, biosolids compost and co-compost of biosolids with MSW did not exceed 10 ppm within the 2 years of the study.

2.4.4.4 Gaseous losses

The main losses of N from the soil are due to crop removal and leaching. However, N losses may also occur by the conversion of N to gases and the subsequent loss to the atmosphere. The primary gaseous N losses involve denitrification and ammonia volatilization (Tisdale et al., 1999).

Denitrification is an anaerobic bacterial process by which nitrate is reduced to nitrite and further reduced to nitrous oxide (N_2O) or dinitrogen (N_2) gas. The reduction process is thought to proceed as shown in Equation 2.4, although it is uncertain whether nitric oxide (NO) is an intermediate.

$$NO_3^- \longrightarrow NO_2^- \longrightarrow NO \longrightarrow N_2O \longrightarrow N_2 \qquad (2.4)$$

Denitrification occurs only when the supply of O_2 required by the soil microorganisms is restricted, generally by extensive moisture (Bear, 1964). The potential for denitrification is high in most field soils, as large populations of denitrifying organisms exist in arable soil (Tisdale et al., 1999). The magnitude and rate of denitrification are strongly influenced by soil, plant and environmental factors, the most important of which are the oxygen availability, nitrate and ammonium concentration, temperature and readily decomposable compounds. N losses under water logged conditions are directly related to the amount and availability of the organic material present. In poor soils or in the absence of an available energy source denitrification is significantly reduced (Bear, 1964; Tisdale et al., 1999).

Ebertseder et al. (2003) in a 4 year field experiment, found that compared to slurry, the compost application resulted in a low increase of N_2O, despite the significantly higher accumulation of humus by compost. Yang et al. (2003) found that the application of urban yard waste compost, liquid pig manure and wheat straw compost, and liquid pig manure to a clay loam soil increased N_2O emissions compared to the control, under an aerobic incubation trial. The addition of nitrate to the pig slurry amended soil increased the N_2O emissions over seven times, but decreased them by 93% and 50% for the pig slurry and wheat straw compost and the urban yard waste compost, respectively.

Chaves et al. (2005b) applied green waste compost to celery residues amended sandy loam soil, in a laboratory experiment, and they demonstrated that the N_2O emissions were reduced at a rate of about 50% compared to the celery-only treatment. Meijide et al. (2007) in a field experiment on irrigated sandy loam soil under Mediterranean conditions found that the application of composted MSW mixed with urea reduced the N_2O emissions by 55% in relation to only urea amended soil, during maize growing season. Ginting et al. (2003) investigated the residual effects of beef cattle manure and composted feedlot manure on N_2O emissions, four years after manure and compost applications had stopped. Both manure and compost were applied for 4 years in a every-year or every-2-years basis. They found that fluxes of N_2O were nearly zero.

The process of gaseous loss of ammonia from the soil to the atmosphere is called volatilization and it may be represented by the Equations 2.5 and 2.6.

$$NH_4^+ \longleftrightarrow NH_{3\,(s)} + H^+ \qquad (2.5)$$

$$NH_{3\,(s)} \longleftrightarrow NH_{3\,(g)} \qquad (2.6)$$

The main factors which influence the equilibrium between NH_4^+ and NH_3 in the soil solution is the pH and the temperature. As the pH increases, the relative concentration of NH_4^+ decreases and that of NH_3 increases. In a similar way, the increase of temperature results in increasing the proportion of NH_3 (Benbi & Richter, 2003). At pH =7 about 1% of the NH_4^+/ NH_3 in solution is present as NH_3 (for a temperature range from 10 to 40 °C), but the percentage increases rapidly with the increase of the pH (the higher the temperature the higher the increase) (Wild, 1988; Benbi & Richter, 2003).

The loss of NH_3 from the soil to the atmosphere is influenced by a variety of soil, environmental and management factors. Maximum NH_3 loss occurs when the soil surface is at or near field capacity and when slow drying conditions exist for several days. Water evaporation from the soil surface encourages NH_3 volatilization (Tisdale et al., 1999). The application of the fertilizer below the soil surface can minimize NH_3 volatilization as any NH_3 that is released will be absorbed by the soil before it can reach the atmosphere (Wild, 1988). In comparison to NH_3 volatilization from N fertilizers, the NH_3 losses from N mineralized from organic N is relatively small (Tisdale et al. 1999).

Jakobsen (1995) found that the risk of ammonia volatilization in the field followed compost application is very little as NH_3 is converted into biomass during decomposition. He et al. (2003) studied the ammonia volatilization of biosolids and West Palm co-compost in an incubation experiment. They found that biosolids resulted in higher amount of NH_3 being volatilized compared to the compost. Also, the incorporation of the organic materials into the soil decreased NH_3 emissions by 5- and 150- fold, respectively for the biosolids and the compost, in comparison to the surface application. Research undertaken by Zubillaga et al. (2005) under field conditions showed that the application of sewage sludge increased NH_3 emissions compared to sewage sludge compost and cattle manure.

2.4.4.5 Ammonium fixation

Certain clay minerals, in particular vermiculite and illite, are capable of holding NH_4^+ as non-exchangeable ion (fixing NH_4^+). The non-exchangeable NH_4^+ can be either native-fixed, during the genesis of silicate minerals, or recently fixed as a result of organic or mineral fertilizer application and organic N mineralization (Benbi & Richter, 2003). Fixation of freshly applied NH_4^+ can occur in clay-, silt-, and sand- size particles if they contain substantial amounts of vermiculite. Coarse clay (0.2 to 2 μm) and fine silt (2 to 5 μm) are the most important fractions in fixing added NH_4^+.

Clay fraction of NH_4^+ provides some degree of protection against NH_3 volatilization, or rapid nitrification and subsequent leaching or denitrification (Tisdale et al., 1999). The recently fixed NH_4^+ may be released during the crop growing season and thus be available for plant uptake (Benbi & Richter, 2003).

Sowden (1976) applied ammonium sulphate and dairy cattle liquid manure to a sandy clay loam soil and found that about 40% of the ammonium nitrogen of the manure and the fertilizer were fixed by the clay. The fixed N was reduced after about one year to a value slightly higher of the control soil at the beginning of the experiment, in the case of the mineral fertilizer. With the manure, a smaller proportion of the fixed ammonium was reduced within the same period. Research on the fixation and release of NH_4^+ under high- and low-fixing soils (Drury & Beauchamp, 1991) has shown that the fixed NH_4^+ pool is a slow release reservoir, with fixed NH_4^+ release being slower than the rate of fixation. Scherer & Weimar (1994) demonstrated that most of the NH_4^+ fixed in clay minerals after slurry application was released during the crop growing season.

2.4.5 Economic implications

The current UK market for compost generally assigns a very low value to the products. Compost benefits to the soil and plant growth are not generally given a monetary value by the farming community who, on the contrary, are often paid to take it away. It is for these reasons that in most case studies a zero cash value is assigned to composts (EA, 2007).

Marketing towards the agricultural and horticultural sectors (including private gardens) requires knowledge of plant requirements as well as an understanding of the needs of the different sectors (Crowe et al., 2002). Figure 2.10 illustrates the potential markets for high quality composts and the relative market size and the attainable price within each market. The market size ranges from small (S) to extra-extra large (XXL) and the prices are in Euros (€) per m^3 within the market segment. The prices are the average of the prices in four EU countries where the composting industry is well established (Austria, Germany, Denmark, and Belgium (Flanders)). It can be seen that there is a variety of uses for compost with different potential market sizes.

The agricultural market is very important in regions where there is a large and rapidly expanding expansion of compost (Crowe et al., 2002). The agricultural sector is a very large market paying low prices. The sector may be willing to pay for the nutrients available in compost if there is no surplus available manure in the neighbouring area. Organic agriculture often pays more for compost products compared to high-input agriculture. The nutrient content of composted biodegradable municipal waste can be high thus paying for transport and application on farmland up to 20-40 km away. Farmers only apply the compost for a short period during both spring and autumn, which is a key consideration when developing a production and marketing plan for a plant. The agricultural market is sensitive towards negative consumer reactions on using composted biodegradable municipal waste (Crowe et al., 2002).

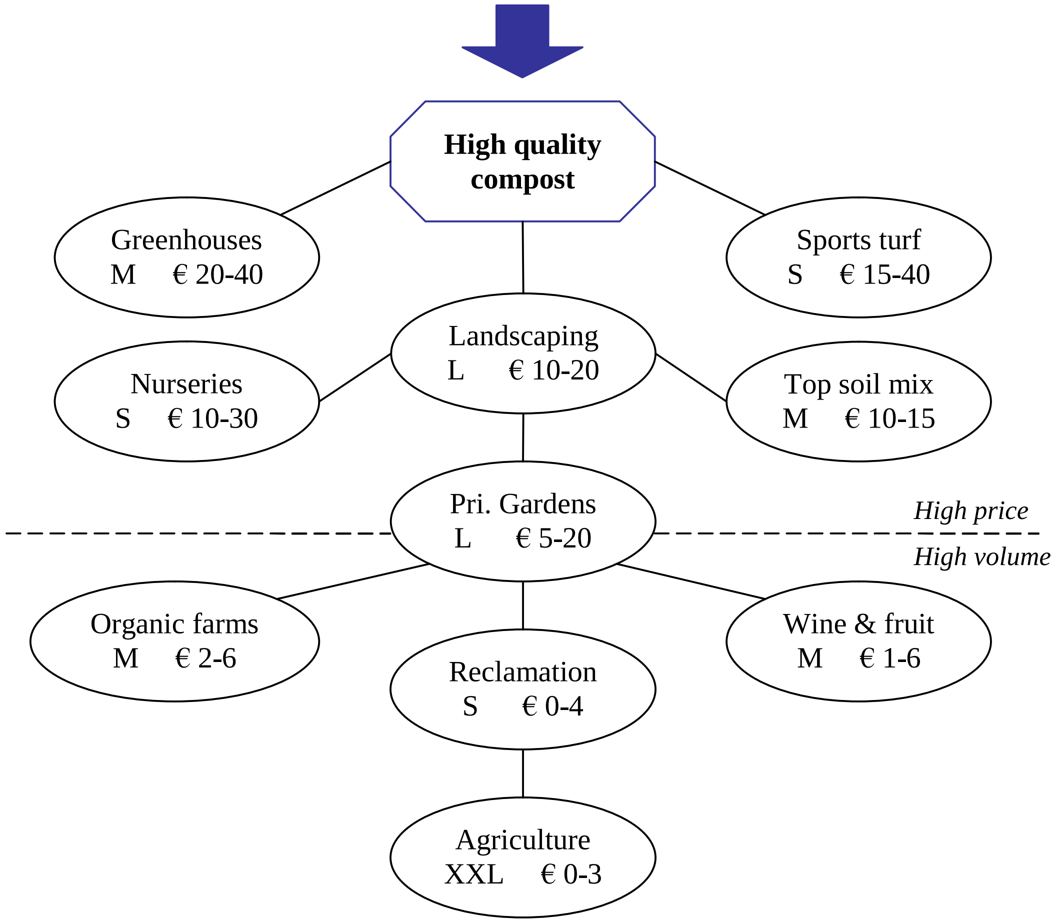

Figure 2.10 Potential compost markets within Europe. For each market the attainable price (bottom right) and market size (bottom left) are given (Adopted from Crowe et al., 2002).

According to Wallace (2005), high quality compost applied at 30 Mg ha^{-1} can have a net value £ 2.55 Mg^{-1} (delivered and spread using farm equipment), depending on the crops being grown, soil type and the circumstances around a given farm. Analytically, compost may have a nutrient component, a liming effect and a beneficial effect on soil physical and biological properties. The compost total available nutrients can have a replacement value of £ 2.50 Mg^{-1} (£ 75 ha^{-1}), the lime equivalent value was estimated for that of 2 Mg of chalk delivered and spread (£33 ha^{-1}), and savings in irrigation were estimated at £ 144 ha^{-1}. The associated costs of compost spreading and haulage, which can be carried out either with farm units or by contractor, are affected by the distance from the composting plant. When farm equipment is used, typical cost was £ 120 ha^{-1} for haulage and £ 55.50 ha^{-1} for spreading (Wallace, 2005).

Another fact that should be taken into consideration is that within the EU Common Agricultural Policy (CAP) reform 2003, the reinforced cross compliance includes respect of standards of good agricultural and environmental conditions, referring to soil protection from erosion, maintenance of soil organic matter and soil structure, and supporting the application of good quality compost (EC, 2003a). In the UK in particular, the application of composts is encouraged under the Single Payment Scheme (SPS) of cross compliance for the soil protection (DEFRA, 2006a).

2.5 Overview

Composting and the land application of the end product is increasing as both an alternative to landfilling for the management of biodegradable waste, as well as means of preserving or increasing the soil organic matter.

Composting is a natural, aerobic process. Different composting systems have been introduced to meet the requirements of the different types and amounts of biodegradable waste, varying from simple static pile to highly controlled in-vessel systems. As the main requisite of composting is the land application of the end product, control of the quality of the compost produced is necessary in order to ensure that compost land application will not endanger the environment. Compost quality standards, legislation and recommendations establishing maximum limits of contaminants or nutrients already exist, but systems managing the utilization of biowaste and vegetable waste compost in agriculture are not yet in place.

The investigation of the effect of compost type, application rate and method on agronomic and environmental parameters is essential for the determination of a proper management practise that ensures sustainable agricultural development. The work which has already been carried out on using biowaste or vegetable waste compost in agriculture is limited, as the majority of the research focuses on other organic materials, such as animal manure or sewage sludge. Moreover, most of the existing work involves the application of a biowaste or a vegetable waste compost under specific experimental conditions (e.g. field trial), whereas there is a lack of integrated studies investigating the same types of compost under varying experimental conditions. There is a necessity, therefore, for comprehensive studies to cover the lack of holistic science and allow for better evaluation of biowaste and vegetable waste compost characteristics in relation to agronomic and environmental parameters.

Better understanding of the fate of compost organic matter and nutrients, especially N, is critical to properly manage compost application. The fate of N following compost application to soil is still not fully understood. The quantification of the availability of biowaste and vegetable waste compost N is essential in order to meet crop requirements, whilst ensuring minimum environmental impacts from nitrate leaching.

It is important to develop integrated approaches to biowaste and vegetable waste compost use in agriculture, which take into account agronomic benefits, and environmental risks, while identifying the financial implications of compost application. Such a holistic approach is critical to promote acceptance of compost use within the public and agricultural sector.

3 Biowaste and vegetable waste composts

3.1 Introduction

Chapter 3 focuses on composts, with particular regards to the biowaste and onion composts used in the experiments. Details about the composting process, the input materials and the properties of the end products are provided. Compost properties were evaluated through laboratory analysis.

The composition of composts varies due to seasonal and geographical variations in the quality of the waste stream, differences in the climatic conditions, composting methods and degree of compost maturity (Eriksen et al., 1999). The laboratory analysis of composts constitutes a guideline for assessing the quality and consequently the utilization perspectives of composts. Knowledge of the nutrient availability in composts is important in order to ensure compliance with land application restrictions and avoid phytotoxic effects.

The specific objectives of the compost analysis were (i) to assess the variation of compost properties in different compost types, and (ii) to produce an understanding of the materials capability to supply crop essential nutrients and the potential interactions with soil quality parameters.

3.2 Composts

Four compost types were considered for analysis: a biowaste, a garden waste, a vegetable waste, and an MBT waste compost.

3.2.1 Biowaste compost

The term ‘biowaste’ is used to describe the source-separated biodegradable part of municipal solid waste (MSW), which contains predominantly food, garden, paper, cardboard waste and maybe some textiles. The biowaste compost used in the experiments was the end product of the 'High Heavens Composting Facility' in Buckinghamshire.

The facility receives two separate incoming waste streams:

1. Kerbside kitchen and catering waste from domestic collections. This stream contains meat inclusive kitchen waste (raw and cooked food scraps), vegetable matter, paper, cardboard packaging and garden waste.
2. Garden waste collected from Buckinghamshire County Council. It comprises mostly tree limbs and hedge trimmings.

The exact composition of both streams shows great seasonal variation. The facility treats all incoming waste in accordance with the Animal By - Products Regulations (ABPR) to produce pathogen-free composted product. The production of the biowaste compost is summarized below (CRS, 2005a):

1. Initial monitoring: inappropriate material are identified and rejected.
2. Shredding: necessary for the increase of the surface area of the compostable materials for enhancing biological activity during composting.
3. Wetting: breaks down the paper waste and promotes biological activity.
4. Screening: removal of any solid item with dimension greater than 40 cm.
5. Composting: the wastes are placed into the composting bay which comprises a sealed vessel (Figure 3.1a). Air is circulated through the material and the natural process of composting proceeds. The temperature is monitored with temperature probes (Figure 3.1b). Once the temperature is maintained over 60 °C for a minimum period of 48 hours (usually it takes 7 to 10 days), the wastes are moved to another composting bay and the composting procedure proceeds. Once the temperature is maintained again over 60 °C for a minimum period of 48 hours (usually 5 to 10 days), a *Salmonella* analysis is carried out. If it is an acceptable S*almonella* test, then the composted material is kept on site for 8 to 10 weeks (curing period).
6. Screening of the compost: the end product results after the removal of bulk materials by screening.

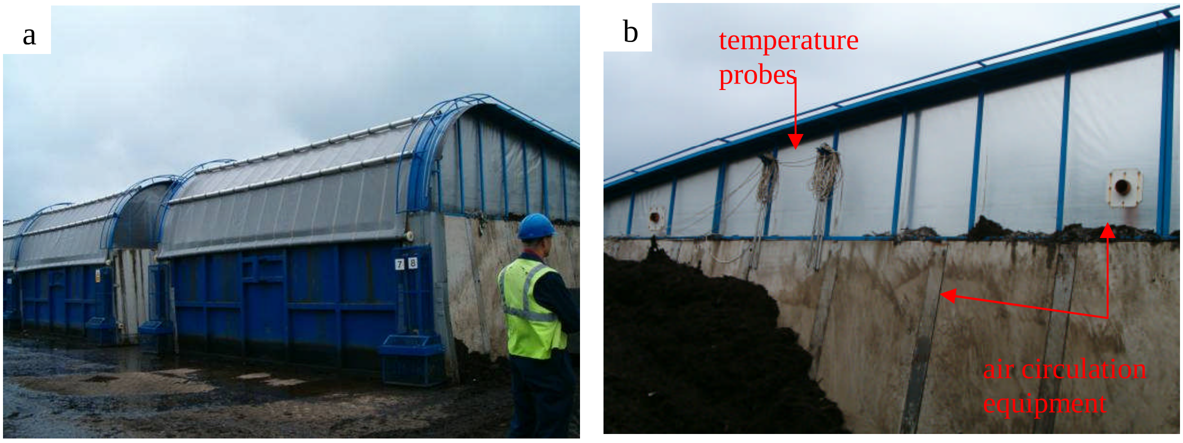

Figure 3.1 (a) The composting vessel sealed, and (b) the inner part of the composting vessel: air circulation equipment and temperature probes.

Samples of the biowaste compost obtained in spring 2005 and 2006 are presented in Figures 3.3a and 3.4a, respectively.

3.2.2 Garden waste compost

The garden waste compost (Figure 3.3b) was also produced at the 'High Heavens Composting Facility' in Buckinghamshire. The composting procedure followed for the production of the garden waste compost was the same as the one described in paragraph 3.2.1. The only difference is that the process was applied only to the second stream of the garden waste.

3.2.3 Vegetable waste compost – onion compost

The vegetable compost used in the study was produced mainly from onion waste, containing outer dry parts of onions, onion tops, and whole, poor quality onions unsuitable for promotion to the market. This onion waste was mixed with straw and composted in static, uncovered piles on the onion production farm (Parrish farm in Bedfordshire) (Figure 3.2).

Figure 3.2 Static pile on-farm composting, employed for the production of the onion compost.

The static pile process employed for the onion compost production involves infrequent turning of the pile (1-2 times a year). The waste materials are not well mixed and the composting process is not consistent within the pile, often resulting in poor quality end-product, especially towards the outer parts of the pile.

The duration of the composting process is about 1 year, allowing for a long curing period. During the curing period the compost temperature is ambient, allowing for the presence of earthworms in the pile. The compost is not supplied deliberately with earthworms, but the worms occur naturally in the pile from the soil, since it is produced on the farm site. Different samples of onion compost are illustrated in Figure 3.3c and 3.4b, collected in spring 2005 and 2006, respectively.

3.2.4 MBT compost

The MBT waste compost used in the laboratory analysis was the end product of the ‘Dunoon IVC’ MBT composting facility in Scotland. The facility receives non-separated MSW generated from households, which constitutes a mixture of organic biodegradable waste, glass, plastic, metals and textiles. The Mechanical Biological Treatment (MBT) procedure followed is summarized below (CRS, 2005b):

The waste is delivered to the facility and inspected for waste-type compliance. The waste is loaded into a drum for mechanical processing. There, the waste is pulverised and oversized particles and ferrous metals are removed. The waste is transported in the composting vessel and the composting procedure proceeds as described for the biowaste compost (paragraph 3.2.1). If the *Salmonella* test is acceptable, the waste is moved to a drying bay in order to be reduced to dust dryness. Finally, the product is milled to shatter glass and screened to remove metals and plastics larger than 4mm. An MBT compost sample collected in spring 2005 is shown in Figure 3.3d.

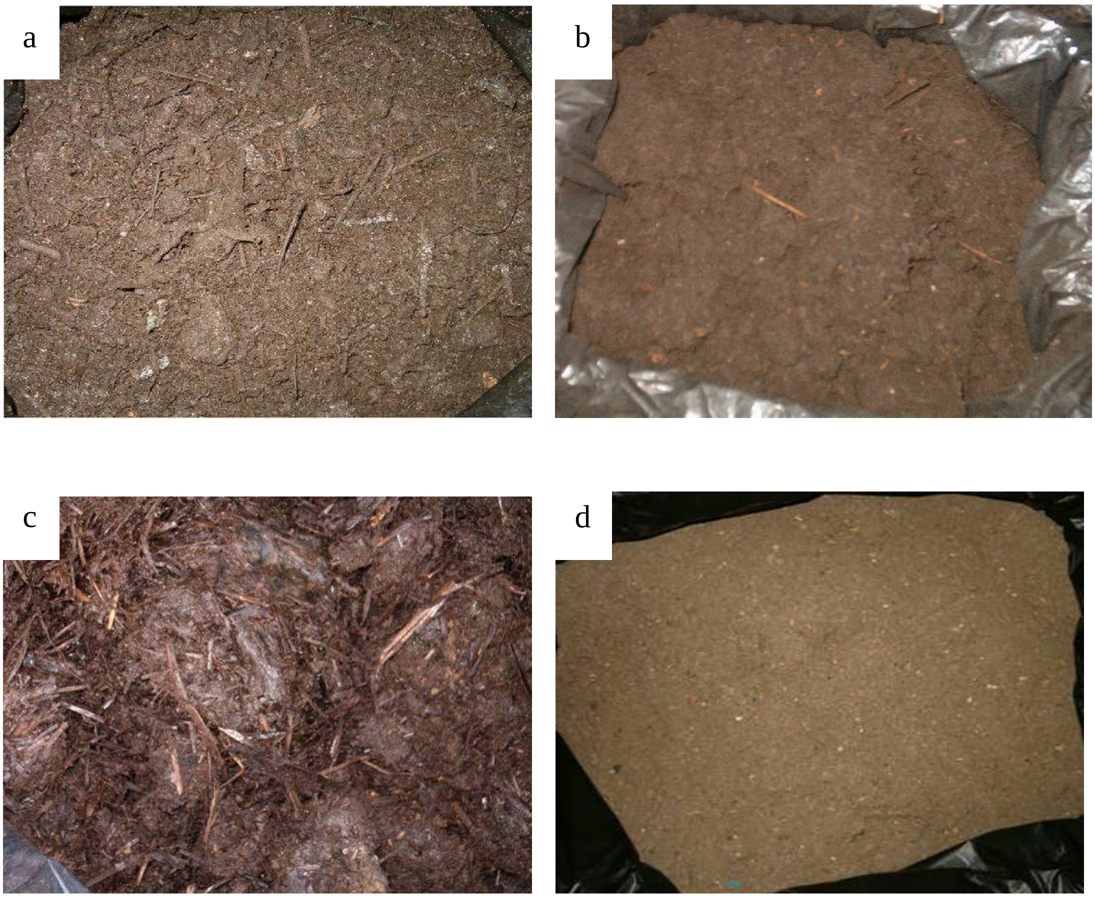

Figure 3.3 The four compost types: (a) biowaste compost, (b) garden compost, (c) onion compost, and (d) MBT compost (samples collected in spring 2005).

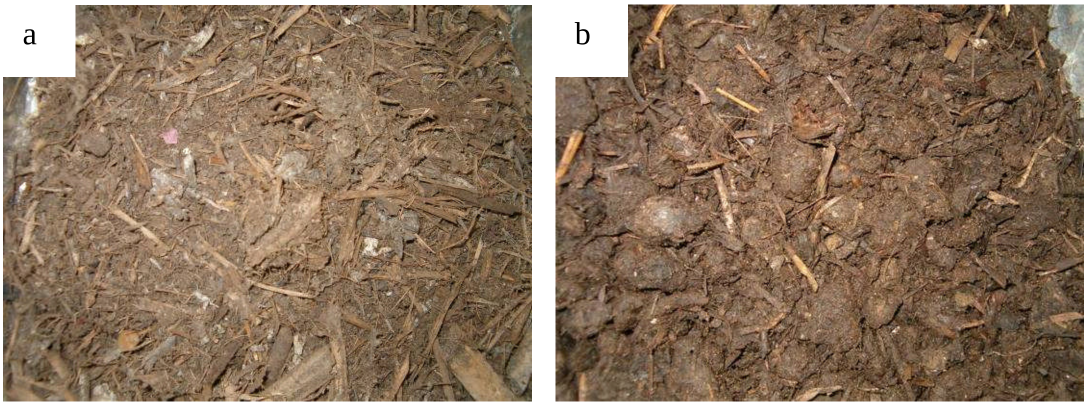

Figure 3.4 Samples of (a) biowaste compost, and (b) onion compost in spring 2006.

3.2.5. Overview

Biowaste compost is of particular importance nowadays as municipalities try to find alternatives to the landfilling of biodegradable MSW. Even though biowaste contains garden waste, a separate garden waste compost was also included for analysis to investigate the differences in the properties of the two composts, as mainly affected by the difference in the input materials. Therefore both the biowaste and the garden waste composts analyzed were produced from the same composting facility, by the same process and the samples analyzed were from the same season. The investigation of the potential utilization of biowaste compost in agriculture was considered more important than the garden waste compost, as biowaste compost may divert larger amounts of biodegradable waste material from landfills than garden waste alone (as shown in Table 2.3, paragraph 2.3.4).

A vegetable waste compost produced on farm, following a simple and low cost composting technique was also included for analysis. It is important to investigate the effects of using vegetable waste compost in agriculture, as its beneficial use may result in good economic returns to the farmer. Vegetable waste is considered a good input material as it is a pure organic material and it is relatively easy to decompose (especially if compared to garden waste, which usually is high in lignin, due to the wood waste included). The quality of both biowaste and garden waste is depended on the efficiency of the source-separation system.

Finally, an MBT compost was also studied. The MBT compost is usually considered suitable for landfilling, especially when it is a product of the residual MSW, or of non-separated MSW. In municipalities where source-separated collection schemes are not in place, MBT can be of great importance for the management of MSW. Therefore, MBT compost produced of non-separated MSW was also considered in the analysis to evaluate potential perspectives for its utilization in agriculture.

3.3 Methods of sampling and analysis

Biowaste and onion compost analysis in spring 2005 and 2006 was required to estimate the nutrient content and other compost properties related to plant and soil factors prior to the experimentation. The two compost types were analyzed for physical and chemical properties, including: bulk density, dry matter content (DM), organic matter content (OM), pH, electrical conductivity (EC), total carbon (C) and nitrogen (N) content, C: N ratio, available/soluble nutrients (mineral N, P, K and Na), and heavy metals content. In 2005, a garden waste and an MBT compost were also analyzed for the above mentioned properties.

In 2005, an initial analysis of the four compost types was undertaken, aiming to investigate into the potential variation of the characteristics of the different compost types. For the initial analysis one sample per compost type was supplied by the compost providers. Following this initial analysis biowaste and onion composts only were sampled from the composting sites in adequate quantities for use in the experiments. These main samples were taken from the same batch as the initial samples. The collected composts were stored in the university farm (under a shed) until they were used in the experiments. The composts were sub-sampled and a bulk sample was produced per compost type. A minimum of three replicates was used per sample. The same methodology was employed in 2006 for sampling the biowaste and onion composts (without initial analysis). An overview of the analytical methods used is summarized below.

Compost bulk density evaluation was based on the BS EN 13040:2000, with the adaptation that the compost was not compressed, but instead it was compacted, since certain compost types were sticky and coarse materials. The compost DM was estimated according to BS EN 13040:2000, by oven drying at a temperature of 103 °C until it reached constant weight. The OM of the composts was calculated as the loss of mass on ignition (450 °C) of the dried compost sample (BS EN 13039:2000). The dried compost sample was defined in the BS EN 13040:2000 (dried at 75 °C and ground to pass through a 2 mm sieve). For the pH measurement fresh samples were extracted with water in an extraction ratio 1+5 (v/v) (BS EN 13037:2000). The pH of the suspension was measured using a pH-meter.

Measurements of total N and total C in composts were carried out on the dried sample by catalytic tube combustion using the Vario EL III, CHNOS elemental analyzer. The C: N ratio was calculated as the quotient of the total C over total N. For the soluble nutrients determination, fresh samples were extracted with calcium chloride DTPA in an extraction volume ratio of 1 + 5 (BS EN 13651:2001). Ammonium nitrogen (NH_4-N) was estimated by an automated phenate method, whereas total oxides of nitrogen (TON) were determined as the sum of nitrites and nitrates, by an automated hydrazine reduction method. Soluble P was determined by an automated ascorbic acid reduction method and K using Flame Photometry.

For the determination of the electrical conductivity two methods were used. According to the first method, the electrical conductivity was measured in the saturated paste extract of the dried compost sample (EC_e) (VanderGheynst et al., 2004). This method was employed for the determination of the EC_e between the four compost types in 2005. According to the second one (BS EN 13038:2000), the electrical conductivity was measured in the water 1+5 (v/v) extract of the fresh compost sample (EC). The second method was used for the onion and biowaste composts only, in both years of the analysis. The electrical conductivity of both extracts was measured at a temperature of 25 °C using a conductivity meter. The measurement of sodium (Na) in the compost samples was carried out in the extracts prepared for the measurement of electrical conductivity, using Flame Photometry.

The composts were analyzed for their content of total Cr, Cd, Cu, Hg, Ni, Pb, Zn, and As according to BS EN 13650:2001. All metals were measured by the Inductively Coupled Plasma (ICP) technique, except for Hg, which was measured according to the atomic fluorescence technique. The analysis for heavy metals was carried out by a commercial laboratory.

The variation of compost properties between the different compost types and the two years of analysis was assessed by analysis of variance (at 5% probability level), using the statistical software GenStat (GenStat, 2006). Basic statistical analysis of compost properties in each year was carried out by Statistica (Statistica, 2005).

3.4 Results and discussion

3.4.1 Composts collected in 2005

Detailed results of the biowaste, onion, garden and MBT composts analyzed in 2005 are presented in Tables A.1.1 and A.1.2 in the Appendix.

The four compost types were found to differ significantly for the majority of the examined properties. Compost bulk density was not found significantly different ($P>0.05$) between the analyzed compost types. The mean bulk density value was found equal to 0.45 g ml^{-1}. Compost dry matter content differed significantly ($P<0.001$) between the four types, as shown in Figure 3.5. This should be mainly attributed to the different input materials and composting process used for the production of each compost type (Tang et al., 2003). The driest compost was the MBT (DM = 89%), which was dried during its production process, whereas the most moist one was the onion compost (DM = 33%), which was produced in an uncovered static pile on farm. The DM of the garden waste compost was 15.6% higher than the biowaste. Since the composting process followed was the same, this difference should be attributed to the input material. Optimum values of compost DM may range from 40% to 60% (Bary et al., 2002).

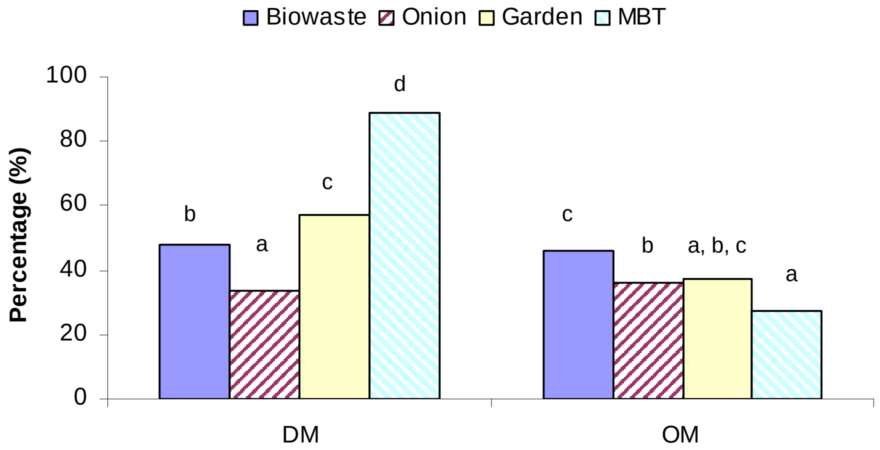

Figure 3.5 DM and OM of the biowaste, onion, garden, and MBT composts analyzed in 2005. Within each property, columns labelled with the same letter are not significantly different (P>0.05).

Differences in DM imply management issues for compost application in agricultural practice. Low DM suggests potential odour problems, and also difficulties with compost application to land due to the bulk of the material. On the other hand, high DM indicates a dusty material which is irritating to work with (MUE, 2005).

Significant differences (P<0.001) were also found in the OM of the four compost types (Figure 3.5). The highest OM was found for the biowaste compost (OM = 46%), whereas the lowest (OM = 27%) for the MBT. It should be noted that the statistical analysis shows that the garden waste compost is not significantly higher in OM than the MBT, even though the onion compost with lower OM is found significantly different. This result is due to the different number of samples of each compost type used for the analysis (see Table A.1.1). The MBT compost was expected to have low OM content as it contains non-biodegradable materials as well.

Zmora-Nahum et al. (2007) showed that the OM of composts containing green waste, vegetable waste, and/or manure may vary from 24% to 77%. Canet et al. (2000) found that the OM of source separated MSW may vary between 22% and 71%. In general, the OM of all composts is high compared to the soil OM, indicating that composts could be applied to land as soil conditioner to increase soil organic matter content, as also stated by He et al. (1992). However, there are some reservations about MBT compost utilization as soil amendment, because its land application could contribute to soil contamination with non-biodegradable materials, such as plastic, glass and metals.

The pH of the biowaste and garden composts was significantly higher (P<0.001) than of the onion compost. The mean pH of the biowaste, garden and MBT composts was higher than 8, whereas the mean pH of the onion compost was 7.1. Compost pH may vary between 4.4 and 9.4 (Tang et al., 2003). According to Barker (1997) pH may indicate the level of compost maturity (see Figure 3.6), as pH varies in composts with time and subsequently with maturation level. Findings of Said-Pullicino et al. (2007) showed the same trend for pH change with composting time as in Figure 3.6. The decrease of pH during the early stages of composting is attributed to the formation of organic acids (typically due to the anaerobic conditions that are established in the waste materials before the beginning of the composting process). As these acids are degraded the pH begins to rise. During the mineralization process the release of ammonium or ammonia increases the pH (Said-Pullicino et al., 2007).

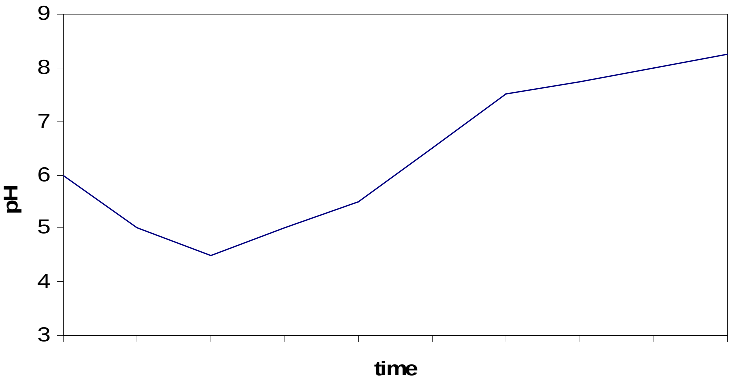

Figure 3.6 Model of trends of pH in composts during the process of composting from addition of raw materials to maturity (after Barker, 1997). Time is in arbitrary units.

Although pH is often not considered as a representative indicator of compost maturity (Sánchez-Monedero et al., 2001; Francou et al., 2005), in the case of the onion compost the lower pH should be attributed to lower levels of compost maturity. The low onion compost maturity was also indicated by the fact that the input materials were recognisable and large particle sizes were present (MUE, 2005), as it is shown in Figure 3.3c.

Furthermore, the pH values of the different onion compost samples, ranged widely in comparison to the other three compost types (see the minimum and maximum values, and the confidence interval of the mean shown in Table A.1.1), indicating the heterogeneity of the onion compost. Onion compost samples were found variable also for DM, OM and nutrients results (Tables A.1.1 and A.1.2).

Compost total N content ranged between 0.8% and 1.8%, as measured for the MBT and the biowaste compost, respectively (Figure 3.7). Typical values of compost total N in the literature may vary from 0.8% to 3% (Iglesias-Jimenez & Alvarez, 1993; Wolkowski, 2003; Zmora-Nahum et al., 2007). MBT compost was found to have significantly (P<0.001) the lowest total N content compared with the other three types.

Total C content followed the same pattern (Figure 3.8) and thus consequently the C: N ratio of the four composts was not found to be significantly different. The C: N ratio constitutes an important indicator of compost decomposition and nitrogen availability to plants. A C: N ratio between 9 and 15 is expected to exclude negative yield effects because of high N immobilization (Nevels & Reheul, 2003). The mean value of all four compost types was close to 16 (Figure 3.8).

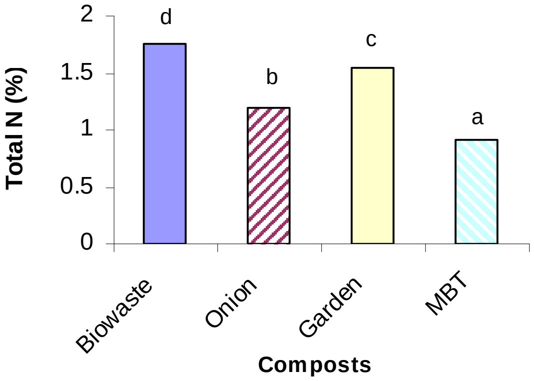

Figure 3.7 Total N content of the biowaste, onion, garden, and MBT composts analyzed in 2005. Columns labelled with the same letter are not significantly different (P>0.05).

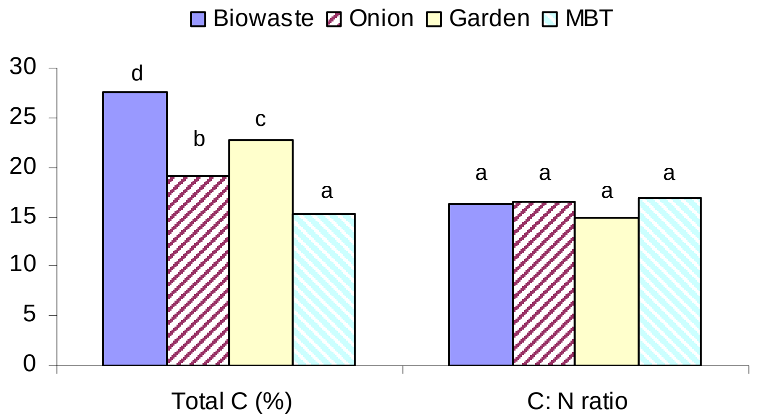

Figure 3.8 Total C and C: N ratio of the biowaste, onion, garden, and MBT composts analyzed in 2005. Within each property, columns labelled with the same letter are not significantly different (P>0.05).

Table 3.1 presents the soluble P, K and mineral N (ammonium (NH_4), and total oxides of nitrogen (TON) -N) content of the four composts. Although there is a trend for the biowaste and onion composts to contain higher levels of NH_4-N and TON-N, than the MBT and garden composts, the differences were not found significant (P>0.05) due to the variability of the mineral N concentrations. The garden waste appeared to have no mineral (readily available) N, and the MBT limited mineral N in the form of nitrogen oxides. In general, the mineral N content of all composts is low and their soil application would result in the addition of limited amounts of mineral N. For example, the application of onion compost at a rate of 250 kg total N ha^{-1} would result in the addition of 6.5 kg mineral N ha^{-1} (according to the values of Table 3.1 and Figure 3.7). The same is true for soluble P.

Table 3.1 Soluble mineral N, P and K content of the biowaste, onion, garden and MBT composts. Columns labelled with the same letter are not significantly different (P>0.05).

	NH_4-N (mg kg^{-1})		TON-N (mg kg^{-1})		P (mg kg^{-1})		K (g kg^{-1})	
Biowaste	221.0	a	62.4	a	268.3	a	7.3	b
Onion	227.7	a	86.2	a	202.5	a	10.5	c
Garden	0	a	0	a	211.8	a	5.1	ab
MBT	40.9	a	0	a	5.2	a	2.4	a

The four compost types did not differ significantly (P>0.05) as for their soluble P content at the 5% level. However, the MBT compost contained significantly less available P than the other 3 compost types at the 10% level of significance. The four composts were significantly different between each other as for their K content. The highest average K content was measured for the onion compost and the lowest for the MBT. Generally, nutrients concentration in composts varies with the original materials and the decomposition degree (Zaccheo et al., 2002; MUE, 2005).

The results of the analysis for EC_e and Na content of the four compost types measured in the saturated paste extract are presented in Figure 3.9. High values of EC_e indicate a large amount of available minerals. The mean EC_e values of the onion and biowaste composts were higher than 4 dS m^{-1}, which could indicate harmful effects on the germinating seeds and plants, even though diluting effects of mixing the compost with soil are expected (MUE, 2005). García et al. (1992) demonstrated that compost phytotoxicity can be related to EC_e > 4 dS m^{-1} when composts are used as growing substrate. According to Bary et al. (2002) EC_e is considered critical for greenhouse potting mixes, but less critical for farmland application, especially in humid regions. Composts often have high values of EC_e due to extensive mineralization of organic substrates (Barker, 1997).

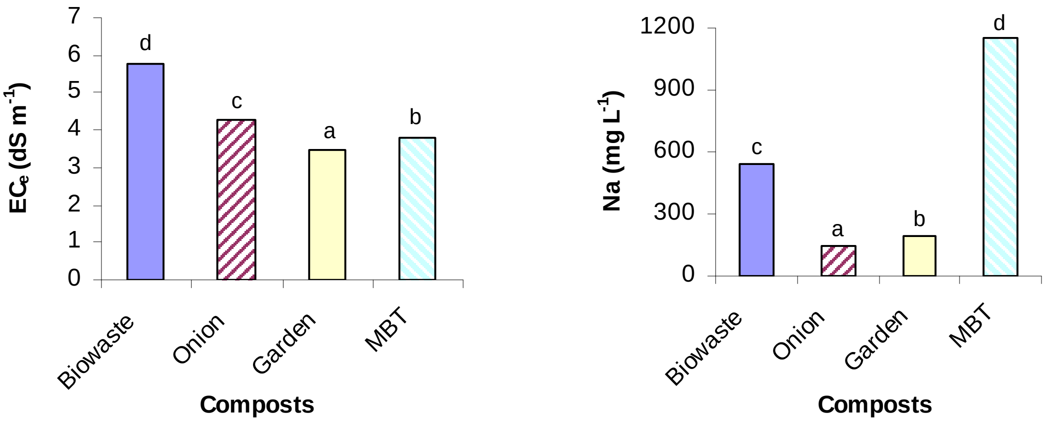

Figure 3.9 EC_e (left), and Na concentration (right), measured in the saturation paste extract of the biowaste, onion, garden, and MBT composts analyzed in 2005. Columns labelled with the same letter are not significantly different (P>0.05).

The important feature to note in Figure 3.9 is that the Na content of the composts is not following the same trend to the EC_e. The onion compost even though it has an EC_e value higher than 4 dS m^{-1}, it is characterized by the lowest Na content. This finding indicates there is a large amount of other available minerals, such as K, mineral N, which can be beneficial for plant growth. Sánchez-Monedero et al. (2001) demonstrated that the increase in compost EC_e values was significantly correlated with the increase of NO_3-N concentration.

On the other hand, the high EC_e value of the MBT compost should be mainly attributed to the high soluble Na, indicating potential sodicity problems following the application of the MBT compost to the soil. The high Na content of the MBT compost may be explained by high levels of salt (e.g. sodium chloride) in the food waste and/or the presense of glass (which contains sodium) in the input waste materials. Potential salinity and sodicity problems could be expected from the land application of the biowaste compost, because of the high values of both EC_e and Na. The garden waste compost with the lowest EC_e and a low Na content seems to be the best of the four types considering the particular properties.

The analysis of biowaste and onion composts according to the BS EN method showed that the difference between the two composts was significant for their Na content, but not for their EC. The mean EC was measured equal to 0.8 dS m^{-1} for both composts. As it is obvious, this method gives lower values for the electrical conductivity compared with the saturated paste extract. This difference should be attributed to the different amount of compost and also the different compost/water ratio used within the two methods. However, the results for Na are consistent with the saturation paste extract results, in the sense that the onion compost was found to have significantly lower amount of soluble Na. The Na content of the onion compost was 37 mg kg^{-1} dry matter (see Table A.1.1), which was only 7% of the Na amount of the biowaste compost.

3.4.2 Composts collected in 2006

Detailed results of the biowaste and onion compost physical and chemical properties determined in 2006 are presented in Table A.1.3 in the Appendix. The two compost types were found to differ significantly for all the examined properties, apart from the soluble P content (Table A.1.3).

The onion and biowaste composts collected in 2006 were quite different materials compared to the respective ones collected in 2005. Some physiological differences of the composts can be observed by comparing the biowaste and onion compost pictures in 2005 and 2006, presented in Figure 3.3 and 3.4, respectively. Biowaste compost in 2006 appears to contain more wood waste, as is suggested by the higher amount of tree limbs seen in Figure 3.4a. The onion compost in 2006 looks completely different compared to 2005. First of all the colour is more brown in 2005. Also, smaller particle sizes can be seen in 2006 and the straw is much less visible, indicating higher maturation level.

Figures 3.10 and 3.11 illustrate the differences of the physicochemical properties of the two composts between 2005 and 2006. Generally the properties of the two composts were found to vary significantly within the two years. This indicates the variability of composts and implies the necessity for compost analysis prior to potential utilization.

As shown in Figure 3.10, in both years the DM, OM, total N and C content of the biowaste compost were significantly higher ($P<0.001$) than of the onion. In 2006, both biowaste and onion composts were drier compared to 2005. The important feature to note is that the onion compost in 2006 with a DM of 50% was less odorous and not as bulky as in 2005, and therefore could be spread on land more easily and evenly.

The biowaste compost OM increased in 2006 by 59%, whereas the onion compost OM was decreased by 46%. The onion compost OM in 2006 was particularly low (20%), which is mainly attributed to the amount of soil included (Bary et al, 2002). In Spain, the minimum acceptable compost OM for spreading to farmland is 25% (Canet et al., 2000). The increase of biowaste compost OM should be due to different input materials.

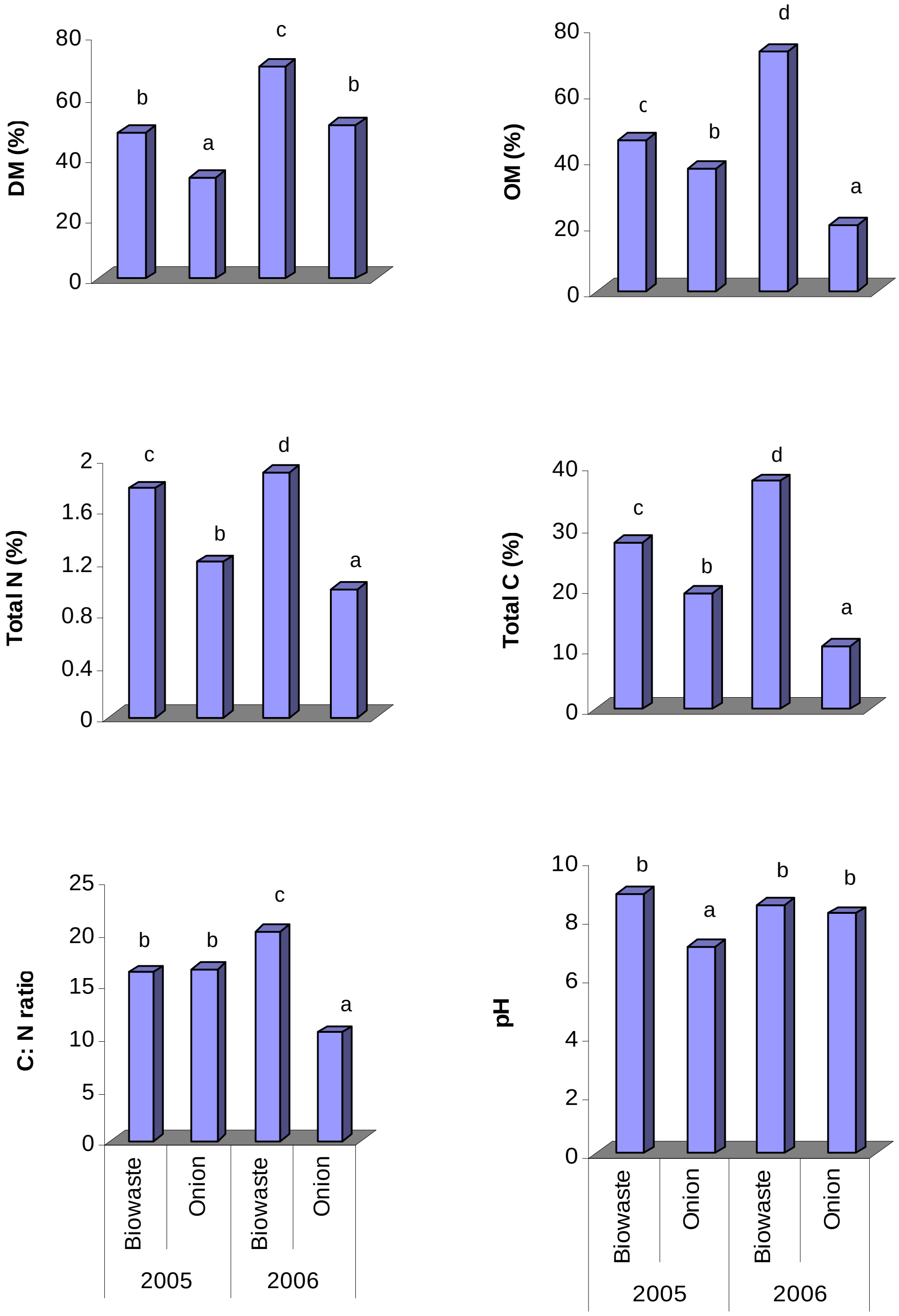

Figure 3.10 Biowaste and onion compost properties in 2005 and 2006: DM, OM, total N, total C, C: N ratio and pH. Columns labelled with the same letter are not significantly different ($P>0.05$).

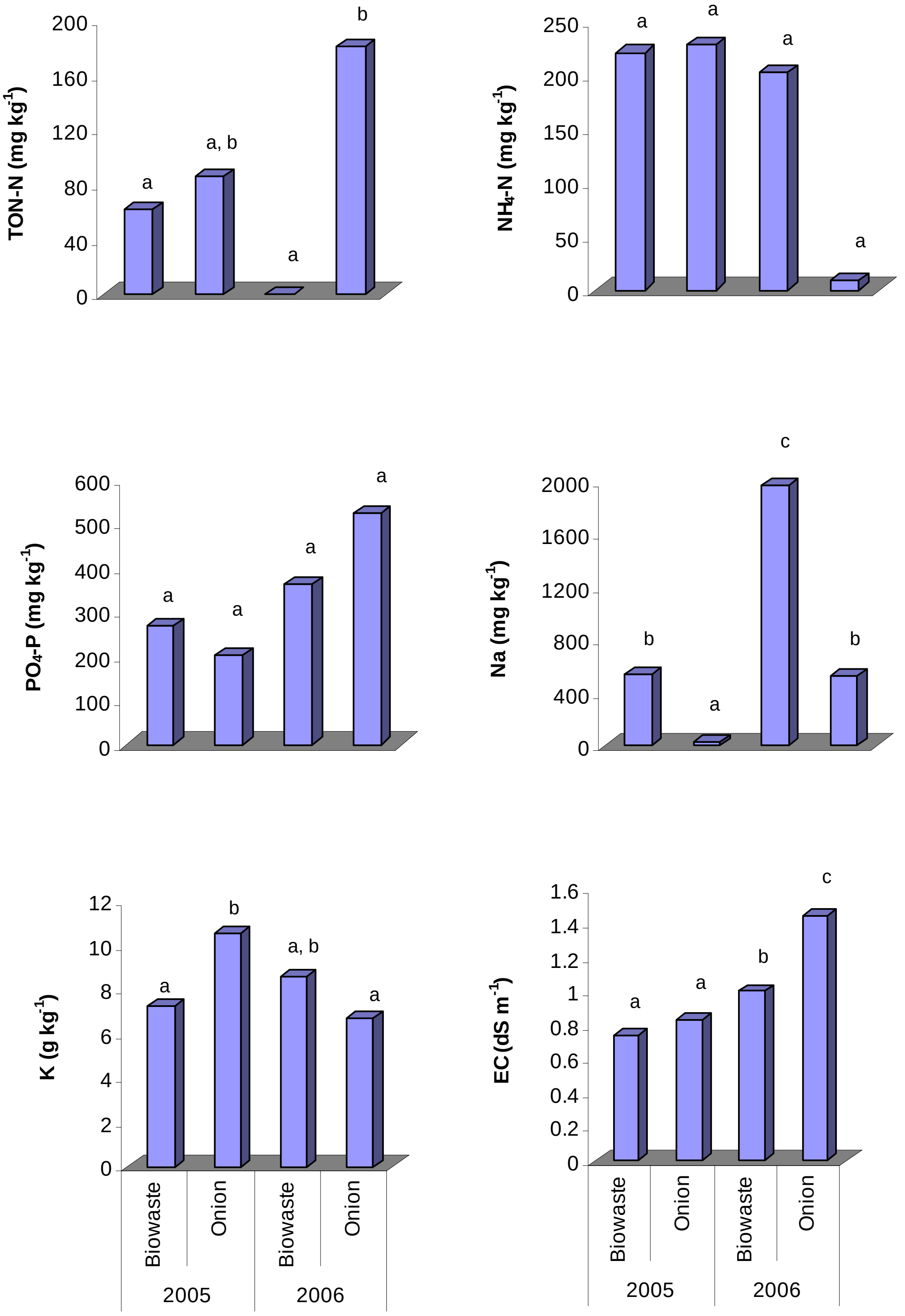

Figure 3.11 Biowaste and onion compost properties in 2005 and 2006: EC, and soluble K, P, Na and mineral N content. Columns labelled with the same letter are not significantly different (P>0.05).

Total C followed the same trend with the OM for both composts. The high total C content of the biowaste compost in 2006 is justified by the presence of wood waste, as according to findings of Tang et al. (2003) wood waste composts are characterized by high total C, and subsequently high C: N ratios. Total N was significantly higher for biowaste than onion compost in both years, by 30% in 2005, and 50% in 2006. Total N content is an important compost property, as the amount of compost land application is restricted according to the amount of total N applied to the soil (see paragraph 2.3.3.3).

The C: N ratio of the two types in 2005 was similar and equal to 16, whereas in 2006 biowaste C: N was 20 and onion was 10. The C: N ratio is considered an important indicator of compost N mineralization or immobilization potential, and subsequently of the N availability to plant growth. The soil application of organic materials with C: N ratio higher than 15 may result in limited N availability due to immobilization (Gutser et al., 2005).

Biowaste compost bulk density was found equal to 0.24 g ml^{-1}, significantly lower than in 2005. This lower value should have mainly resulted from the presence of the wood waste, which constitutes bulking agent and may reduce the density of the material. On the contrary, onion compost bulk density was significantly higher in 2006 (0.66 g ml^{-1}). This result should be mainly attributed to the smaller particles present in the 2006 compost. Composts pH was above 8 in both years; exception was the onion compost in 2005, with a pH of 7. EC was higher in 2006 for both composts. Onion compost even though it was characterized by higher EC, contained less Na in both years.

According to Zucconni & De Bertoldi (1987), compost NH_4^+ content should not exceed the level of 0.04% dry weight (or 0.031% NH_4-N), as the NH_4^+ is an indication of unstabilized materials. Both composts in both years were found below this limit. Findings of Cooperband et al. (2003) and Said-Pullicino et al. (2007) indicate that increase in soluble NH_4-N results from ammonification of easily mineralizable organic N. As the composting process proceeds NH_4-N concentration decreases as a result of immobilization, ammonification, and/or nitrification. After 200-250 days of composting, increased NO_3-N concentration has been reported (Cooperband et al., 2003; Said-Pullicino et al., 2007). The results of NH_4-N and NO_3-N content of the onion compost in 2006 indicate the nitrification of NH_4-N. Sánchez-Monedero et al. (2001) demonstrated that the highest NO_3-N concentration is produced at the end of maturation and can vary from 0.12% to 0.53%. Onion compost NO_3-N concentration in 2006 was 0.02%, under the assumption that NO_3-N was equal to TON-N.

It should be noted, however, that soluble nutrients, such as TON-N and NH_4-N, can vary widely within the same batch (as it was shown in Tables A.1.1, A.1.2 and A.1.3) and therefore their value for assessing compost quality may be limited. The same is true for soluble P. The measurement of total P, along with soluble P, could provide better indication of compost P content and the potential P availability. Compost soluble K content was found to be in comparable range between the two composts in both years, with the onion compost in 2005 to contain the highest amount of K.

3.4.3 Heavy metals

Table 3.2 summarizes the results of the indicative heavy metal analysis, which was carried out for the four compost types in 2005, and the onion and the biowaste composts in 2006. Table 3.3 presents the maximum allowed compost content of heavy metals according to certain compost quality standards (Table 3.3 is a copy of Table 2.2 in Chapter 2 and it is re-presented in this section to facilitate the comparisons between the measured values of heavy metals in the composts with the existing compost quality standards. More details about Table 3.3 can be found in paragraph 2.3.3.2).

The heavy metals concentrations of the biowaste compost in both years were within the limits of all standards, with the exception of the Eco-agric standards, which are the most limiting ones, and the Class 1 limit for Cd. Onion compost was within the limits of the Eco-agric, with the exception of Cd. The garden waste compost was found to contain elevated levels of Pb. Pb level of garden waste compost was only within the limits of Class 2 and BSI PAS 100 standards.

Table 3.2 Average heavy metal content of the four compost types: results for 2005 and 2006 samples.

Determinant (mg kg^{-1} DM)	**Composts**					
	2005				2006	
	Biowaste	**Onion**	**Garden**	**MBT**	**Biowaste**	**Onion**
Cadmium (Cd)	0.9	0.9	0.6	0.6	0.4	1.1
Chromium (Cr)	32	63	28	21	25	57
Copper (Cu)	56	29	60	196	79	41
Lead (Pb)	82	11	147	383	50	22
Mercury (Hg)	<0.1	<0.1	0.1	0.2	0.1	0.1
Nickel (Ni)	23	16	18	24	15	17
Zinc (Zn)	177	56	201	917	176	74
Arsenic (As)	9	14	8	4	5	13

Table 3.3 Maximum concentration of heavy metals allowed in composts according to the 2nd draft of the WD on biological treatment of biowaste (Class 1 and 2), the EU Eco-label, the EU Eco-organic, and the BSI PAS 100.

Determinant	**Compost standards**				
(mg kg^{-1} DM)	**Class 1**	**Class 2**	**Eco-label**	**Eco-agric**	**BSI PAS 100**
Cd	0.7	1.5	1	0.7	1.5
Cr	100	150	100	70	100
Cu	100	150	100	70	200
Pb	100	150	100	45	200
Hg	0.5	1	1	0.4	1
Ni	50	75	50	25	50
Zn	200	400	300	200	400

MBT compost was observed to have high levels of Pb, Zn and Cu. The MBT content of Pb and Zn was well above all the quality standards, and the Cu level was just below the BSI PAS 100 limit only. MBT compost was expected to have increased heavy metals content because it is produced from non separated MSW (Gomez, 1998; Brinton, 2000). The mechanical separation involved in the MBT process decreases the heavy metal content of the produced composts. Levels of Cu, Pb and Zn in MSW compost with mechanical separation were reported equal to 290, 740 and 870 mg kg^{-1}, respectively, whereas in MSW compost without separation equal to 512, 850 and 1640 mg kg^{-1}, respectively (Gomez, 1998).

Limitations about the As content of composts are established in the Netherlands. Composts in order to be classified as Class 1 should not contain more than 15 mg of As per kg dry compost, whereas for Class 2, the limit is 25 mg kg^{-1} dry compost (Brinton 2000). All composts were below 15 mg kg^{-1}, although onion compost was close to this limit.

Heavy metals were not expected to be a problem for the onion compost. However, the Cd and As content of the compost was relatively high. Onions generally are not characterised by high levels of either As or Cd in their plant-tissue, unless they have absorbed them from the water or soil. Onions, because it is a root crop, may accumulate high As levels when they grow in areas with elevated As concentrations in soil or water (Schalau, 2006). According to Alexander & Alloway (2007) onions can accumulate Cd and Pb. Phosphorus fertilizers may be common sources of Cd in agricultural soils (Page et al., 1987). Elevated concentrations of heavy metals in biowaste compost should probably be attributed to compost contamination with non-biodegradable MSW materials, especially in cases of low efficient source separation.

According to the heavy metals analysis mentioned above, the concentrations of heavy metals in the analyzed samples were within the quality standards. Despite that, it is suggested to monitor soil concentration of heavy metals in order to avoid excessive heavy metals accumulation especially after continuous and high dose compost application.

3.5 General discussion

3.5.1 Compost properties

The examined physical and chemical compost properties provided a useful insight into compost characteristics and also information about the potential effects of compost land application on soil quality, and subsequently on crop growth. Of all the examined properties the most relevant ones, which may require measurement before compost land application, are summarized in Table 3.4.

Compost soluble P and mineral N, even though they provide an indication of directly plant available nutrients, they can be widely variable within the same batch and therefore their value for assessing compost quality and potential effects of land application may be limited.

Table 3.4 Important compost properties and their potential implications for land application.

Property	Relative influence
Heavy metals	defines compost suitability for land application
Total N	determines the amount of compost to be applied to the soil
C: N ratio	indicates the N mineralization potential of composts: can be used along with the total N content to estimate N fertilizer requirements for crop production
DM	influences the land spreading of composts: application techniques and potential nuisance of odour problems
OM	indicates the potential increase of soil OM
EC_e and Na	indicates potential salinity and sodicity problems
K	indicates K plant availability: K fertilizer requirements

3.5.2 Compost types

The laboratory analysis of the four compost types in 2005 indicated that composts are variable materials. Differences in properties of the different compost types examined were expected mainly as a result of the different input materials and composting procedures used.

However, differences were also observed for the same compost type between different samples. These differences should be mainly attributed to different ratios of input materials used and also differences in compost maturity level. The finding that composts are variable materials indicates that, for sustainable compost utilization in agriculture, compost land application should depend on, and adjust according to the specific characteristics of composts, as also concluded by Verdonck et al. (1987). Alternatively, compost manufacturers should aim to produce compost with consistent characteristics, providing with a specific range of nutrients and OM. This option would allow for easier management of compost, but also for higher prices of the end product (due to increased production costs). The production of consistent compost is difficult to be achieved with simple composting techniques, such as the static pile method employed for the production of the onion compost on-farm.

The analysis of the four compost types in 2005 suggested that all composts may contribute to the increase of soil organic matter level and potentially enhance nutrient availability. However, compost application may also result in soil degradation mainly through the increase of electrical conductivity and Na levels, and heavy metal contaminants.

The onion compost was significantly variable in 2005 and 2006. This variation should be mainly attributed to different level of compost maturity. This difference in maturity was mainly due to the composting method involved. The static pile composting process is not uniform within the pile, and generally, the outer parts of the pile are usually not well composted. The land application of a compost with the characteristics of the onion compost in 2005 could raise problems associated with production of foul odour, and uneven distribution of nutrients because of its bulk nature; thus specific land-spreading practices are required.

Onion compost in both years was found to be a relatively good source of nutrients and organic matter (on average in DM basis: 29% OM, 1.1% total N, 0.9% soluble K, 0.03% soluble P). Its relatively high electrical conductivity may suggest potential salinity problems especially following continuous application. Potential soil Cd accumulation should also be taken into consideration, and therefore soil EC and Cd levels needs to be monitored.

The biowaste compost appeared to be rich in nutrients and organic matter (58% OM, 1.8% total N, 0.8% soluble K, 0.03% soluble P), and generally of good quality in both years of the study. However, EC_e and Na values were found elevated indicating potential salinity and sodicity problems. Heavy metal content of biowaste compost was at low levels both years, with only Cd to be of potential concern. Biowaste compost C: N ratio was higher than 20 in 2006, most probably due to the wood waste included, indicating potential problems with N immobilization following its land application.

Garden waste compost was also found to be rich in nutrients and organic matter (37% OM, 1.5% total N, 0.5% soluble K, 0.02% soluble P), without particularly high values of EC_e and Na, but with high levels of Pb. The results for the garden compost in comparison to those of the biowaste compost reflect the effect of the input materials in the properties of the end product.

The analysis indicated that the MBT compost should not be considered suitable for application to agricultural land. The land application of the MBT compost could lead to soil contamination with non-biodegradable materials, such as plastic, glass and metals. The nutrient content of the MBT compost was generally the lowest, indicating low fertilizing potential. Moreover, the dusty nature of the MBT compost makes it difficult to work with and spread it to land. However, the most important issue which should exclude the use of MBT compost in agriculture is the high levels of heavy metals (Pb, Zn and Cu).

Generally, the MBT compost is considered suitable for landfilling (Smith et al., 2001; Clemens et al., 2003). However, it should be noted that the MBT process is variable, as explained in paragraph 2.2.2.3. The findings and conclusions drawn from this analysis should be related to the MBT composts produced from non-separated MSW following the process described in paragraph 3.2.4, and not generalized for all MBT compost types.

3.6 Conclusions

The conclusions drawn from the compost laboratory analysis are summarized below:

a) Composts should be applied to agriculture as they may contribute to the increase of soil organic matter level and potentially enhance nutrient availability to the crop. However, in cases of saline or sodic soils, and/or in the production of saline sensitive crops compost application may be discouraged. Land application of non-separated waste compost should be restrained because of potential soil heavy metal contamination.

b) The onion and biowaste composts examined were suitable for application to agricultural land. So was the garden waste compost. The MBT compost, produced from non-separated MSW, was found unsuitable for agricultural utilization, mainly because of its high levels of heavy metal contaminants.

c) Composts are variable materials. Differences in properties of the different compost types studied were mainly attributed to the input materials (or different ratios of input materials), the composting process, and the level of compost maturity.

d) There is evidence to suggest that due to compost variability, prior to application to agricultural land, composts should be analysed for the following properties: heavy metal content, total N and C: N ratio, electrical conductivity and Na content, OM and available K, as they may influence soil quality parameters and subsequently crop production.

e) The variation of compost properties implies management issues for their sustainable utilization in agriculture. Thus, compost management should depend on, and adjust according to the specific characteristics of each compost type and batch produced.

4 Field experiment: effects on forage maize production and soil properties

4.1 Introduction

The laboratory analysis of the composts, discussed in Chapter 3, provided an indication of the composts utilization perspectives in agriculture. However, experimentation is necessary in order to assess the actual effects of compost application to agricultural land. From the four composts analyzed, biowaste and onion composts were chosen to be used in the experimentation, as they are representative of municipal and agricultural organic waste composts that can be beneficially applied to agriculture (for further details see paragraphs 3.2.5 and 3.5.2).

Chapter 4 focuses on the two years field trial which was undertaken to study the effects of biowaste and onion compost application to forage maize production on sandy loam soil, under actual farming conditions. This was essential to provide results which could be extended to practical recommendations for compost application to agricultural land, and also could allow for a more accurate assessment of the economic costs and benefits involved.

The objectives of the field experiment were:

1. To assess the forage maize yield and nitrogen availability to plants from a two years compost application.
2. To estimate the possible threats of compost use for excessive nitrate leaching.
3. To evaluate the effect of compost type and application rate on soil properties.

4.2 Materials and methods

4.2.1 Field experiment establishment

The experiment was conducted on a sandy loam soil at the experimental farm of Cranfield University at Silsoe in 2005 and 2006 (68.6% sand, 19.4% silt and 12.0% clay). The site used to be grassland until 1980, when the grass was ploughed in the soil (0.2 m). Since 1980 to the beginning of the experiment the site had been used for the production of potatoes in 1990 and dahlias in 1991 and 1992. The weeds were ploughed in the soil annually (0.2 m). Some fertility parameters of the topsoil (0-0.3 m) at the beginning of the experiment are presented in Table 4.1.

Table 4.1 Topsoil properties at the beginning of the experiment.

pH	**OM** (%)	**EC_e** ($dS\ m^{-1}$)	**ESP** (%)	**Total N** (%)	**C : N**	**Mineral N**	**Olsen P** ($kg\ ha^{-1}$)	**K**
6.7	3.01	0.8	0	0.12	10.36	54.6	369.7	944

The soil was amended with the biowaste and onion composts described in Chapter 3. The two composts were applied at three rates based on their total nitrogen (N) content. In the first year agronomic rates were used of 80, 165 and 250 kg total N ha^{-1}. In the second year the rates were increased as follows: 100, 250, and 400 kg total N ha^{-1} (the lower rate in the second year (100 kg total N ha^{-1}) was applied to the plots receiving the lower rate in the first year (80 kg total N ha^{-1}) and respectively for the other two rates). The compost application rates along with the corresponding amount of nutrients applied each year are summarized in Table 4.2.

The biowaste compost was surface applied and incorporated. The onion compost was only incorporated because of problems related to foul odour produced from the onion compost in the first year of the experiment. Prior to compost incorporation, the composts were spread onto the respective plots by hand, as evenly as possible. Then they were incorporated within the top approximately 0.15 m depth by a power harrow. Compost spreading and incorporation was on June 1, 2005 and June 8, 2006. After the sowing was completed, biowaste compost was surface applied to the respective plots (on June 9, 2005 and June 19, 2006).

Table 4.2 Biowaste and onion composts: application rates and respective amounts of nutrients and organic matter (OM) applied per ha, and composts C: N ratio for the two years of the field experiment.

Application rates (kg total N ha^{-1})	**Fresh material** (Mg ha^{-1})	**Soluble mineral N** (kg ha^{-1})	**Soluble K** (kg ha^{-1})	**Soluble P** (kg ha^{-1})	**OM** (Mg ha^{-1})	**C: N**
2005						
Biowaste compost						16.2
80	9.4	1.3	32.8	1.2	2.1	
165	19.0	2.6	66.0	2.4	4.2	
250	29.6	4.0	103.0	3.8	6.5	
Onion compost						16.5
80	20.0	2.1	70.4	1.4	2.5	
165	39.8	4.2	140.3	2.7	4.9	
250	62.4	6.6	219.7	4.2	7.7	
2006						
Biowaste compost						20.1
100	7.9	2.0	47.6	2.0	4.0	
250	19.8	5.0	119.0	5.0	10.1	
400	31.7	8.1	190.4	8.1	16.1	
Onion compost						10.5
100	20.8	2.0	70.5	5.5	2.1	
250	52.1	5.0	176.3	13.7	5.2	
400	83.4	8.0	282.1	22.0	8.4	

Mineral N fertilizer (ammonium nitrate) was applied to separate plots at rates ranging from 0 to 80 kg N ha^{-1}, in order to determine the N availability from the composts. These plots were also amended with fertilizer P and K as shown in Table 4.3. Details regarding the mineral fertilization can be found in paragraph A.3.1.2 in the Appendix. Mineral fertilizer was surface applied by hand after the crop sowing was completed (the same dates that the biowaste compost was surface applied). No mineral fertilizer was applied to plots which received composts in both years of the study.

Each treatment was applied on an individual plot of 2.5 m x 5 m using a complete randomized block design with three replicates. The field experiment layout is illustrated in Figure A.3.1.1 in the Appendix. Forage maize (*Zea Mays* L.), variety Vernal, was used as the monitoring crop in both years. Crop sowing was at a rate of about 96000 seeds ha^{-1}.

Table 4.3 Mineral fertilizers N, P and K applied to each reference treatment in 2005 and 2006.

Treatment	Mineral fertilizer N (kg ha^{-1})	 P (kg ha^{-1})	 K (kg ha^{-1})
2005			
Control	0	0	0
Mineral_80	80	0	61.8
2006			
Control	0	0	91.3
Mineral_40	40	0	91.3
Mineral_80	80	0	91.3

Figure 4.1 shows part of the field trial after the completion of the experiment establishment in 2005. Details about the experimental establishment and monitoring are given in paragraph A.3.1.1 in the Appendix, along with information regarding the climatic conditions during the experimental period.

Figure 4.1 Part of the field experiment in 2005. The plot shown in the front was amended by biowaste compost surface applied at 250 kg total N ha^{-1}. Forage maize was at about the 3rd leaf growth stage.

4.2.2 Crop and soil measurements and analysis

The effect of compost application to agricultural land was assessed by measuring crop and soil properties. Crop growth was monitored only in 2005 by measuring the plant height of 8 maize plants in the center of each plot. Plant height was defined as the distance from the back of the collar of the upper complete leaf to the soil surface. A leaf was deemed to be complete when its collar was completely formed (Easson & Fearnehough, 2000).

The crop was harvested in both years once it had completed its physiological maturity and its dry matter content was about 30% (Draper, 2003). At harvest, 3 whole maize plants were randomly selected from the centre of each plot and were harvested by hand. Crop above ground dry matter (DM) yield was determined by drying maize cobs at 60 ˚C until constant weight, and the rest of the plant at 105 ˚C for 26 hours. Maize total N concentrations were determined separately for the cobs and the rest of the maize plant by dry combustion using the Vario EL elemental analyzer. The DM yield of maize cobs and of the rest maize plant were multiplied by the respective total N content, and then summarized to crop N uptake.

Maize total P and K concentrations were determined in 2005 for the whole maize plant according to MAFF (1986). Crop P and K uptake were also evaluated by multiplying the whole maize DM yield by the measured P and K concentrations, respectively.

Soil quality properties were determined after harvest for the soil layer 0-0.3 m depth. Organic matter content (OM) was determined by the loss on ignition method, pH in the 1/2.5 soil/water suspension, total N, total C and C: N ratio by the use of elemental analyzer. Available P was estimated by the Olsen method and measured by spectrophotometry. Extractable K and Na were extracted by ammonium nitrate and measured by flame photometry (MAFF, 1986). Soil salinity and sodicity were estimated by measuring the electrical conductivity in the saturated paste extract (EC_e), and the exchangeable sodium percentage (ESP) from the sodium adsorption ratio (SAR) according to Richards (1954).

Soil ammonium (NH_4) and total oxides of nitrogen (TON) -nitrogen fluctuations were monitored during the period of September 2005 to October 2006 within the top 0.6 m of the soil profile in the field. Below the 0.6 m depths there was sandstone. Samples were taken for the following soil layers: 0-0.3 m and 0.3-0.6 m. Mineral N (TON-N and NH_4-N) was measured in potassium chloride extracts by segmented flow analysis. The amount of soil residual TON-N after the crop harvest each year was used to estimate potential environmental threats due to excessive nitrate leaching. Analytical results expressed as concentrations were converted to kg per ha assuming a bulk density of 1.44 g l^{-1}.

Also, the soil water release curve (WRC) was produced by using the sand table method (Harst et al., 1975) and the pressure membrane apparatus (Richards, 1947; Salter & Haworth, 1961). Plant available water was determined as the water retained between the states of field capacity (FC) and permanent wilting point (PWP). Soil FC was taken at -10 kPa and PWP at -1500 kPa.

Details of the methods involved in the soil and crop sampling and analysis can be found in Appendix A.2.

4.2.3 Statistical analysis

The effects of each treatment and the influence of compost type and application rate on the measured variables were assessed by analysis of variance (5% probability level) and the relation between variables was established by regression analysis (5% probability level) using the statistical package GenStat (GenStat, 2006).

The accuracy of the statistical analysis was checked graphically by the residual plots. When the residuals did not follow the normal distribution then the data were normalized by applying a square root or a natural logarithm transformation. Also, in cases of an extreme value in the data set, this value was removed from the analysis when this data point caused the residuals to be not normal.

4.3 Results and discussion

4.3.1 Forage maize production and nutrient uptake

4.3.1.1 Crop growth, dry matter yield and nutrient uptake in 2005

Forage maize growth was vigorous in all treatments. Erisken et al. (1999) found that the application of compost with C: N ratio 40:1 resulted in impaired maize growth (stunted and pale plants) at the early stages of the growing season only, due to early season immobilization of soil N. Throughout the experiment, biowaste and onion compost (C: N ratio 16:1) application was not observed to impair forage maize growth, as the plants did not show any sign of stress or nutrient deficiency. Pictures of crop production at different growth stages are shown in Appendix A.3.1.3.

The height of forage maize plants produced on the control soil was found not to be significantly different (P>0.05) from the height of the plants produced on the compost or mineral fertilizer amended soil (Figure 4.2). Some interactions between compost type and application rate were observed to be significant with respect to plant height. However, the effect of compost type or application rate was not clear. Mineral fertilizer application resulted in similar plant height to the control treatment.

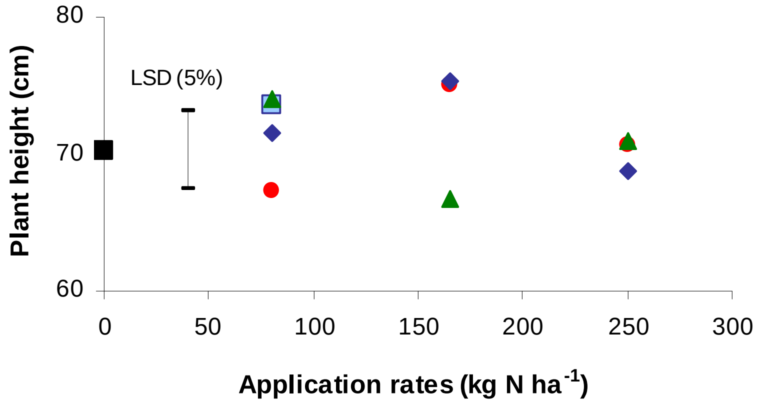

Figure 4.2 The forage maize height response to compost and mineral fertilizer application at different rates (means of the whole growing season in 2005). (■ : Control, □ : Mineral, ◆: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated).

The forage maize production obtained in 2005 was plotted as dry matter (DM) yield and N uptake response to the different rates of compost and mineral fertilizer application in Figures 4.3 and 4.4, respectively.

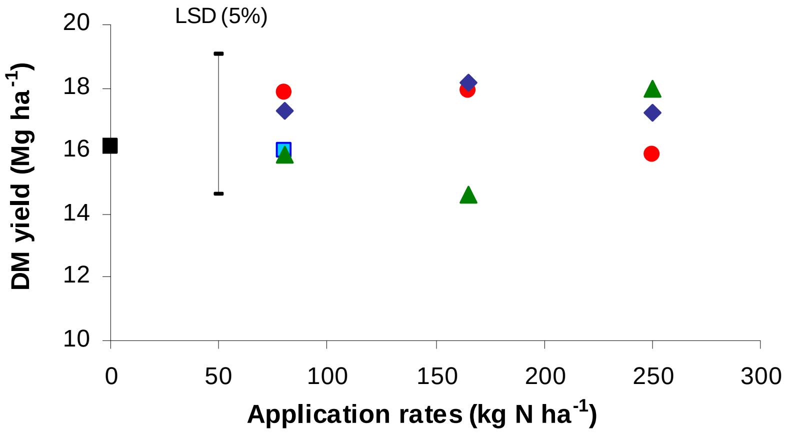

Figure 4.3 Forage maize DM yield in the first year of compost application (2005) (■ : Control, □ : Mineral, ◆: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated).

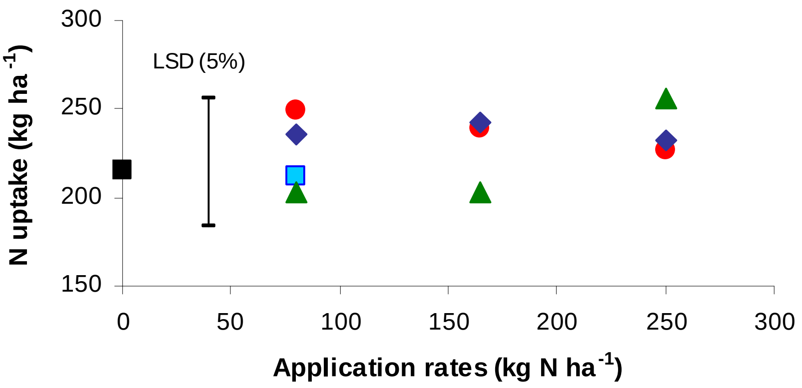

Figure 4.4 Forage maize N uptake in the first year of compost application (2005) (■ : Control, □ : Mineral, ◆: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated).

There are a number of features to note for Figures 4.3 and 4.4. The first is that the N uptake response to compost and mineral fertilizer application followed similar patterns with the DM yield. The second is that the differences between the treatments were not significant (P>0.05), which suggests that neither the application of composts, nor the mineral fertilization increased the crop yield significantly. The increase of onion compost application rate showed a trend for higher crop DM yield and N uptake, although the increase was not significant.

The control (non-amended soil) treatment resulted in 16 Mg ha^{-1} DM yield. This value is in good agreement with forage maize dry matter yields reported for forage production under mineral fertilization at rates higher than 120 kg N ha^{-1} (Pain & Phipps, 1974; Eriksen et al., 1999), which suggests that the soil itself was capable of providing the necessary nutrients for forage maize production, without any further fertilizer or compost application to be necessary.

The ratio of total N input (from the composts) to N output (N uptake by aboveground forage maize biomass) was lower for the highest N input (Figure 4.5). This lower N use efficiency at higher rates could indicate increased environmental risks due to N losses. However, at the highest application rate, almost all the applied N appeared to have been taken up by the plant (N output: N input ratio is approximately equal to 1), whereas at the lower rates outputs are greater than the inputs (ratio > 1). These findings indicate the net mineralization of soil inherent N during the growing season, since low release of compost N is expected during the first season after application, due to N immobilization by the soil microbial population (Sullivan et al., 1998; Nevens & Reheul, 2003). Work by Richards et al. (1999) has shown a net mineralization potential of about 140 kg ha^{-1} mineral N in different soils in the UK in the 7-10 weeks following maize drilling.

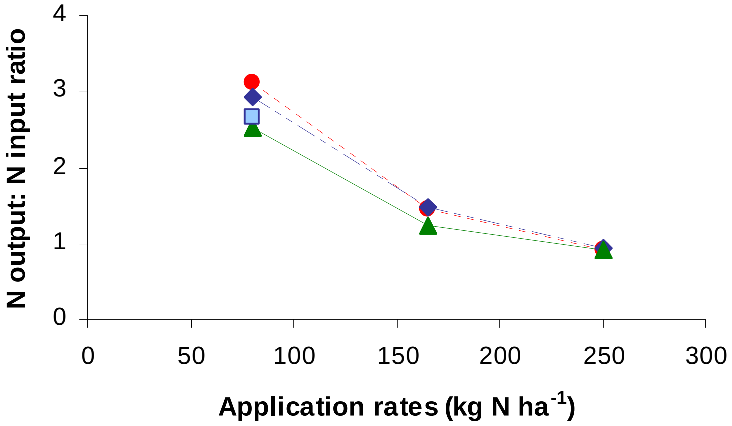

Figure 4.5 N output: N input ratio on forage maize production (□ : Mineral, ◆---◆: Biowaste surface applied, ●---●: Biowaste incorporated, ▲—▲ : Onion incorporated)

Phosphorus and potassium uptake were found not to be significantly influenced ($P>0.05$) by compost or mineral fertilization (Figure 4.6). K uptake was at similar levels for all treatments, whereas P uptake showed a trend to increase with the increase of compost application rate. Soil P was in high levels as at the beginning of the experiment (soil P index was equal to 6 according to MAFF (2000) classification), capable of supporting forage maize production with no further P addition to be necessary (MAFF, 2000). Therefore the trend for increased P uptake with increased compost application rate may indicate increased P availability from compost treatment with higher rates. Similar trends were not observed for the K uptake. The finding that the control treatment was not significantly different from the mineral fertilizer or compost treatments shows that the soil was capable of covering the crop P and K requirements.

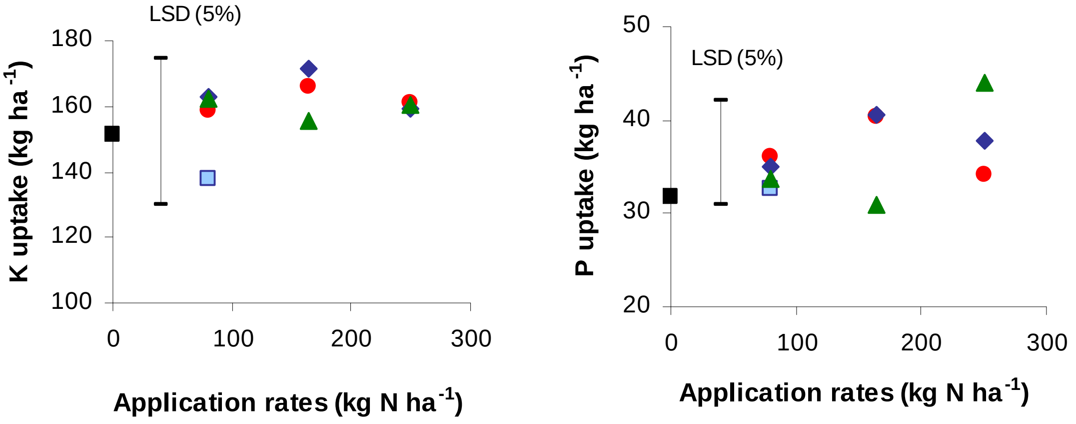

Figure 4.6 Forage maize K uptake (left) and P uptake (right) in the first year of compost application (2005) (■ : Control, □ : Mineral, ◆: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated).

4.3.1.2 Dry matter yield and nitrogen uptake in 2006

Due to the limited forage maize response to compost application and mineral fertilization in the first year of the experiment and the evidence of the high soil fertility of the site, only crop dry matter yield and nitrogen uptake were determined in 2006. [The experiment was continued in 2006, despite the poor crop and soil response to compost and mineral fertilizer application in 2005, aiming to study the effects of a 2-year compost application on soil properties and quantify the compost N mineralization potential through measurements predominantly of the soil mineral N content, as limited crop response was expected (see paragraph A.3.1.2.1 in the Appendix)]. Figures 4.7 and 4.8 present the DM yield and N uptake response to the different rates of compost and mineral fertilizer application, respectively.

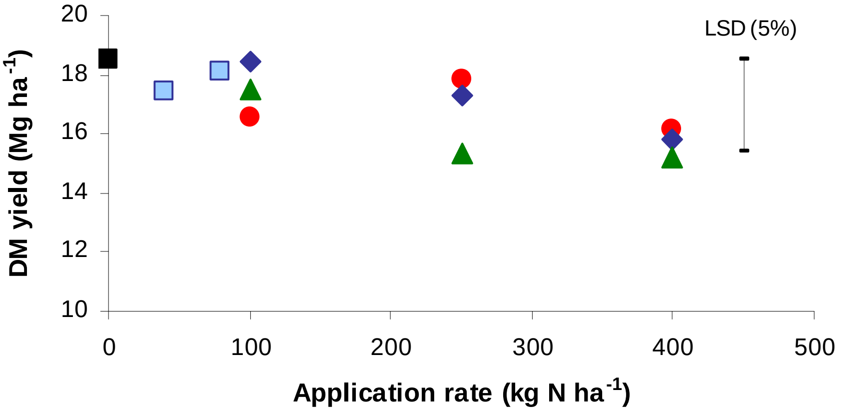

Figure 4.7 Forage maize DM yield in the second year of compost application (2006) (■ : Control, ☐ : Mineral, ◆: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated).

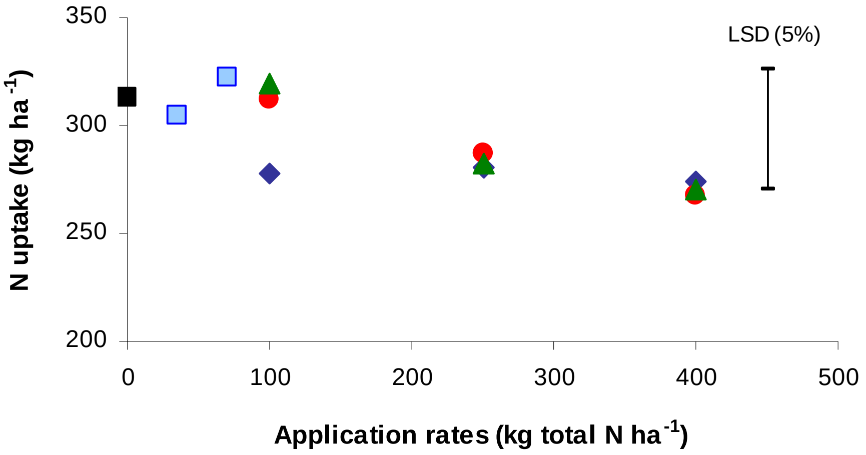

Figure 4.8 Forage maize N uptake in the second year of compost application (2006) (■ : Control, ☐ : Mineral, ◆: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated).

It can be seen that the control treatment resulted in high DM yield and N uptake in 2006. DM yield and N uptake from the control treatment were higher in 2006 compared to 2005, reflecting the warmer and sunnier weather conditions during the crop growing season in 2006 (see Table A.3.1.2 in the Appendix), and indicating higher soil N mineralization. Compost application or mineral fertilization had no significant effect (P>0.05) on crop dry matter yield and N uptake in the second year of the study.

A trend of decreasing N uptake with increasing rates of onion and biowaste composts incorporated was observed. However, the feature to note is that in 2006 N uptake was not following closely the DM yield results (as was shown for 2005). Also, the crop yield response to the type and rate of compost application was not consistent within the two years of the study. These finding along with the high LSD values suggest that the field experiment could not provide a clear indication of the effect of compost application on forage maize production. The reason for this was mainly attributed to the mineralization of soil inherent N and its utilization by the crop; the control treatment resulted in the same crop N uptake as the mineral fertilizers, indicating that the soil itself could cover the crop N requirements without any further N addition to be necessary. As a result, it was not possible to evaluate the N availability of compost N to the crop in both years of the experiment. However, the compost N mineralization patterns were estimated by monitoring the soil mineral N levels.

4.3.2 Soil mineral nitrogen

Figures 4.9 and 4.10 present the results of the soil TON-N and NH_4-N monitoring from September 2005 to October 2006 for the soil layer of 0-0.6 m depths, respectively, grouped by the treatment type.

NH_4-N was generally in low levels (less than 5 kg N ha^{-1}) during the monitoring period for all treatments. Only in September 2005, soil NH_4-N levels reached about 20 kg N ha^{-1} for the control and the mineral fertilizer treatments, whereas the compost treatments were at levels lower than 10 kg N ha^{-1}. The differences between the treatments were found to be not significant (P>0.05).

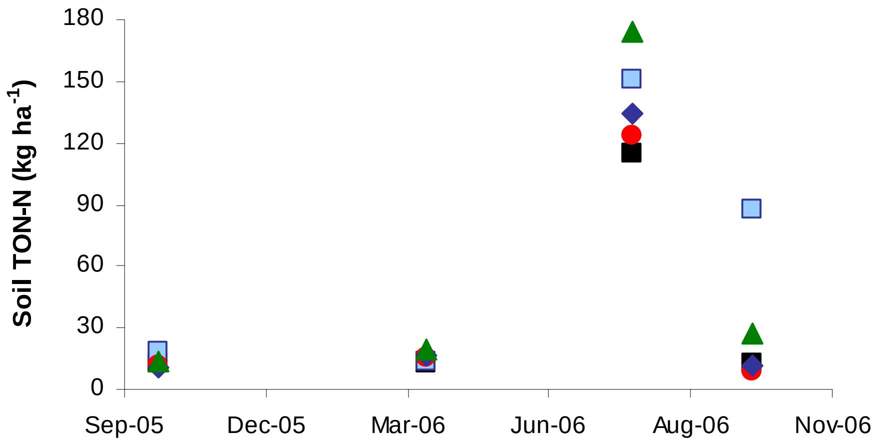

Figure 4.9 TON-N fluctuations within the soil layer of 0-0.6 m depths during the period of September 2005 to October 2006 (■ : Control, □ : Mineral, ◆ : Biowaste surface applied, ● : Biowaste incorporated, ▲ : Onion incorporated).

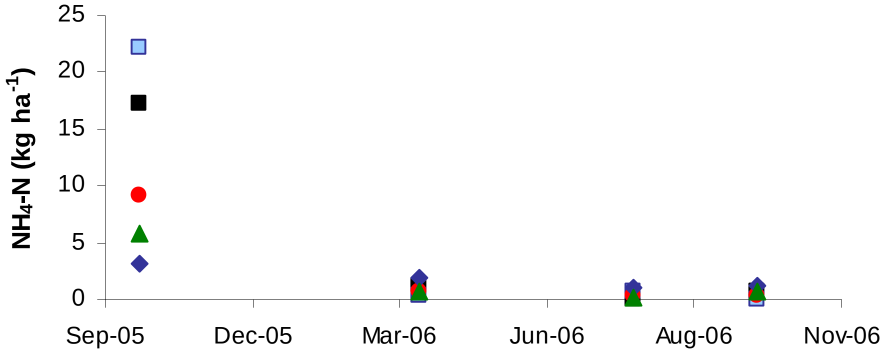

Figure 4.10 NH_4-N fluctuations within the soil layer of 0-0.6 m depths during the period of September 2005 to October 2006 (■ : Control, □ : Mineral, ◆ : Biowaste surface applied, ● : Biowaste incorporated, ▲ : Onion incorporated).

Soil TON-N was generally found to be in higher levels compared to NH_4-N, ranging between about 15 to 180 kg N ha^{-1} during the measuring period, thus indicating that this was the predominant form of mineral N in the soil profile of 0-0.6 m depths.

Figure 4.11 presents the soil TON-N levels after the crop harvest in September 2005. No significant differences (P>0.05) were observed for the different treatments within both sampling depths. The effect of compost application rate was found to be not significant (P>0.05). The decrease of TON-N concentration downwards the soil profile indicates low N losses by run-through and efficient N uptake by the crop during the growing season. Since all treatments are in similar levels with the control, excessive nitrate leaching during the winter was not expected from any of the different treatments.

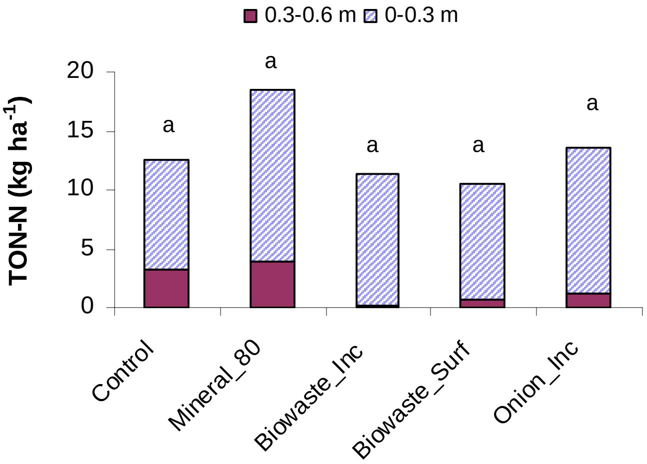

Figure 4.11 Soil TON-N levels within the soil layer of 0-0.6 m depth measured in September 2005. Columns labelled with the same letter are not significantly different (P>0.05). (Inc: incorporated, Surf: surface applied; 80: application rate in kg N ha^{-1}).

The soil TON-N content was measured again in the middle of March 2006 (Figure 4.12), in order to estimate possible nitrate leaching losses from the different treatments during the winter. The treatment 'Mineral 40*' at this stage of the experiment is the same as the control treatment, since the second rate of mineral fertilization was applied for the first time in June 2006, after the crop sowing. (During the first year of the experiment the respective plots of the treatment 'Mineral 40' received exactly the same tillage as the control).

Comparing the Figures 4.11 and 4.12 it can be seen a decrease in the TON-N levels within the 0-0.3 m soil layer and a subsequent increase within the 0.3-0.6 m layer. Despite this downwards movement of TON-N throughout the soil profile, soil TON-N content of the 0-0.6 m depth remained at levels close to those measured in September and therefore no or limited leaching losses were expected to have occurred during winter.

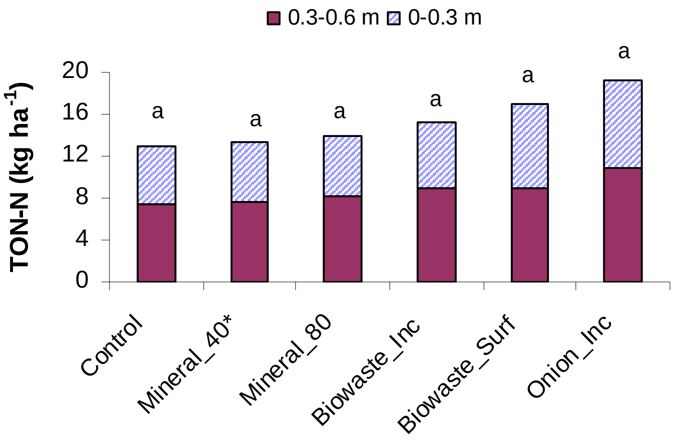

Figure 4.12 Soil TON-N content within the soil layers of 0-0.6 m depths in March 2006. Columns labelled with the same letter are not significantly different (P>0.05). (Inc: incorporated, Surf: surface applied; 40, 80: application rates in kg N ha^{-1}).

The effect of the compost application rate was found to be significant (P<0.05) in March 2006, with the lower rate (80 kg total N ha^{-1}) to result in significantly less soil nitrate-N content compared to the other two higher rates (Figure 4.13). The effect of compost application rate on soil TON-N content was not clear in any other sampling date. Petersen (2003) also found the effect of household waste compost application rate to be insignificant, over a three-year field experiment on sandy loam and loamy sand soil.

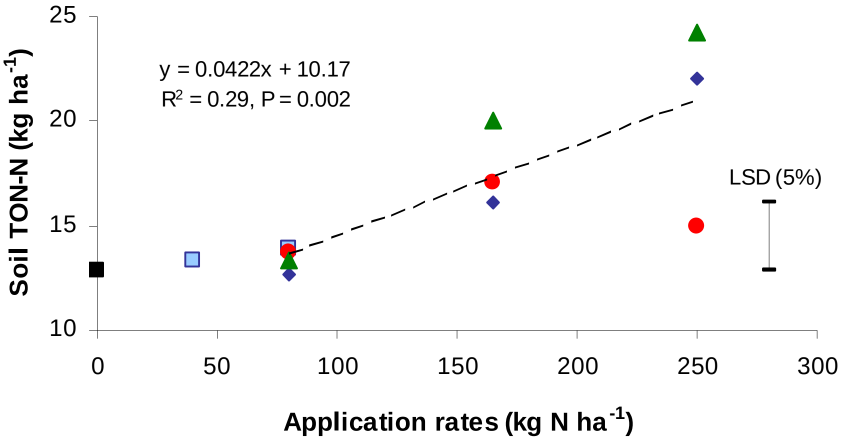

Figure 4.13 The effect of compost application rate on soil TON-N levels within the top 0.6 m depths in March 2006 (■ : Control, □ : Mineral, ◆ : Biowaste surface applied, ● : Biowaste incorporated, ▲ : Onion incorporated).

The next measurement of soil TON-N content took place in July 25, 2006, about 1 month after the application of the composts and the mineral fertilizer. At this sampling date soil TON-N levels for all treatments were higher compared to the levels measured the previous two sampling dates (Figure 4.9). This finding may be attributed to increased mineralization of soil and compost N due to the increased temperature during the summer months. The high value of TON-N measured for the non-amended soil treatment (control) indicates that the high TON-N levels were also supplied from the inherent soil N mineralization. This finding is in accordance with finding of Richards et al. (1999) who measured soil TON-N levels in different soil types in the UK, and supports the results of the high crop production from the control treatment presented in paragraph 4.3.1.

The forage maize plants at the time of sampling in July were between the 6th and 9th leaf growth stage, which means that the N uptake by the plants had already started by then. However, after the 9th leaf growth stage, and for about another month, the soil N is in greater demand to meet the needs of the increased crop growth rate (Ritchie & Benson, 1993). The high TON-N content measured at this time of year could justify the crop N uptake values measured at harvest.

Figure 4.14 presents the differences in soil TON-N observed between the different treatment types in July 2006. It is shown that the onion compost application resulted in significantly higher soil TON-N levels compared with the control and the biowaste compost treatments, and also in the same levels of TON-N as the mineral fertilizers. This finding indicates mineralization of onion compost N. Onion compost application resulted in significantly higher soil TON-N levels in comparison to the biowaste compost treatments.

Figure 4.15 illustrates the soil TON-N levels for the different treatments and rates of application in July 2006. The calculated LSD value at the 5% probability level is 66.3 kg TON-N ha^{-1}, which is a generally high value when considering soil TON-N levels and it is indicative of the high variability of TON-N content in the soil at the time of the measurement. Using a probability level of 20% reduces the LSD in 42.2 kg ha^{-1} and indicates more clearly that the mineralization of biowaste compost N was at levels similar to those from the control or the mineral fertilizer amended soil at 40 kg N ha^{-1}. This finding may indicate limited mineralization or immobilization of biowaste compost N during the crop growing season. Onion compost is shown to be more efficient in increasing the soil TON-N levels than the biowaste compost.

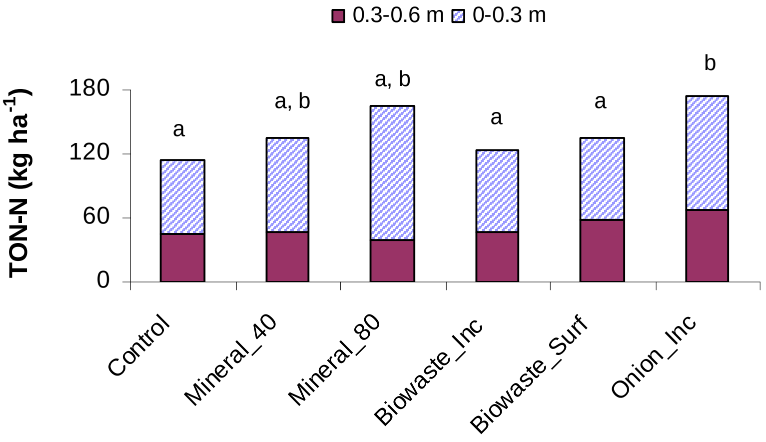

Figure 4.14 Soil TON-N levels within the top 0.6 m depths in July 2006. Columns labelled with the same letter are not significantly different (P>0.05). (Inc: incorporated, Surf: surface applied; 40-80: application rates in kg N ha^{-1}).

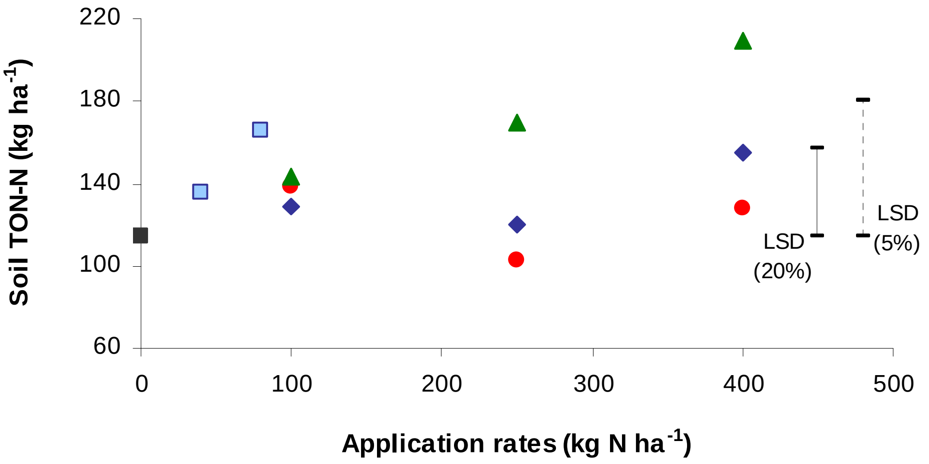

Figure 4.15 The effect of compost application rate on soil TON-N levels within the top 0.6 m depths in July 2006 (■ : Control, □ : Mineral, ◆ : Biowaste surface applied, ● : Biowaste incorporated, ▲ : Onion incorporated).

The last measurement of soil TON-N was on October 10, 2006, about 1 week after the crop harvest, in order to estimate the residual soil TON-N following two years of compost application (Figure 4.16).

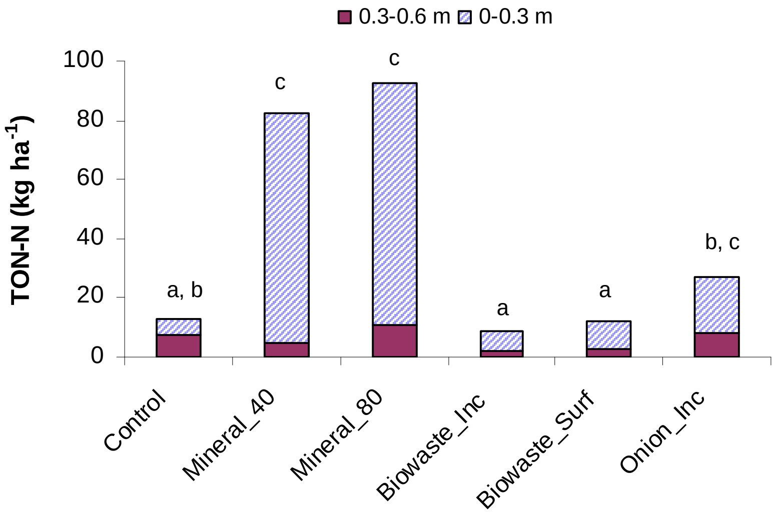

Figure 4.16 Soil TON-N content within the top 0.6 m depths in October 2006. Columns labelled with the same letter are not significantly different ($P>0.05$). (Inc: incorporated, Surf: surface applied; 40-80: application rates in kg N ha^{-1}).

Figure 4.16 shows that both rates of mineral fertilization resulted in significantly higher residual soil TON-N content than the control, indicating that increased nitrate leaching losses could be expected during the winter from the mineral fertilizer treatments. The compost amended soil resulted in similar levels of residual TON-N to the control, indicating that increased nitrate leaching should not be expected during the winter from the compost amended soil. The TON-N content of the onion compost amended soil was significantly ($P<0.05$) higher than the biowaste amended one.

The decrease of the soil TON-N content from July to October 2006 for all treatments shows N utilization by the crop. However, leaching losses during the growing season might also have occurred, especially from the onion compost treatment. This can be derived from the difference of the soil TON-N content between the onion compost and the control treatments in July and October. Since the N uptake by the crop was the same for all treatments and the residual soil TON-N content in October was the same for the onion and control treatments, then the extra TON-N measured in July for the onion compost treatment is possible to have leached during the growing season, as it was in excess of the crop N requirements.

The non-amended soil (control) was found capable of meeting the crop N requirements in both years of the field study, mainly through the mineralization of inherent soil N. Therefore the N applied by the mineral fertilizers and mineralized from composts should have been in excess to the crop N requirements and thus susceptible to leaching. The inherent N fertility of the experimental site prevented the determination of compost N availability to plants and also resulted in overestimating the environmental impact of nitrate leaching from the onion compost and mineral fertilizer application.

During the monitoring period of the soil TON-N content, the effect of the application method of the biowaste compost was observed to be not significant ($P>0.05$). The second year of the field experiment has shown evidence of limited mineralization or immobilization of biowaste compost N, and mineralization of onion compost N. The C: N ratio of the biowaste compost used in the 2nd year field experiment was 20, whereas the C: N ratio of the onion compost was equal to 10.5. These results support the finding that organic materials with C: N ratio between 9 and 15 prevent negative yield effects because of high N immobilization (Eriksen et al., 1999; Nevens & Reheul, 2003).

4.3.3 Other soil properties

The two years of compost application was observed to have limited influence to most of the rest soil properties. The following properties were not significantly affected ($P>0.05$) by the compost application: organic matter content (mean soil OM content of all treatments was 3.7%), pH (mean pH was 7.1), total N (was 0.15%), total C (15%), C: N ratio (10.0), and extractable P (375.0 kg ha^{-1}).

In the second year of compost application (2006), it was observed that the onion compost significantly increased soil K content compared with the biowaste and the control ($P<0.05$). Figure 4.17 illustrates the soil extractable K levels in October 2006 as affected by the different treatments and application rates.

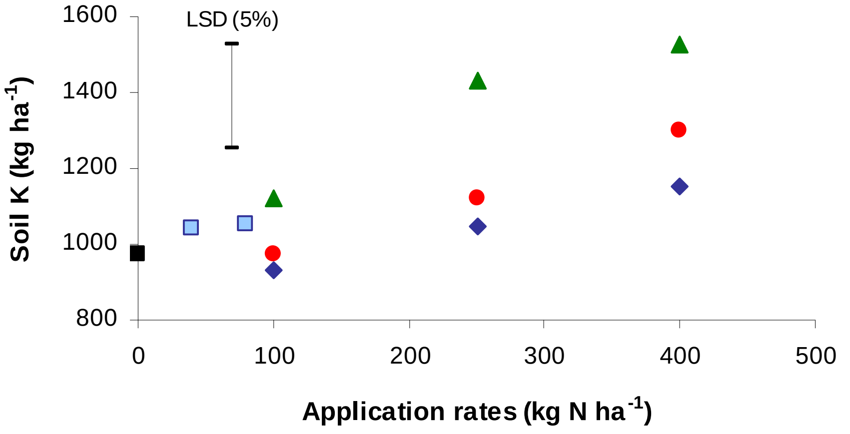

Figure 4.17 Soil extractable K levels following two years of compost application in comparison to mineral fertilization (■ : Control, ☐ : Mineral, ◆ : Biowaste surface applied, ● : Biowaste incorporated, ▲ : Onion incorporated).

The increase of compost application rate significantly increased soil K content ($P<0.05$). The reason why the onion compost resulted in higher soil K levels was mainly related to the higher amounts of soluble K applied by the onion compost in both years of the study (see Table 4.2). Figure 4.18 presents the relationship between compost K added (sum of soluble compost K in 2005 and 2006) and the soil K levels in 2006. It can be seen that the soil K content increases linearly with the addition of higher amounts of compost K.

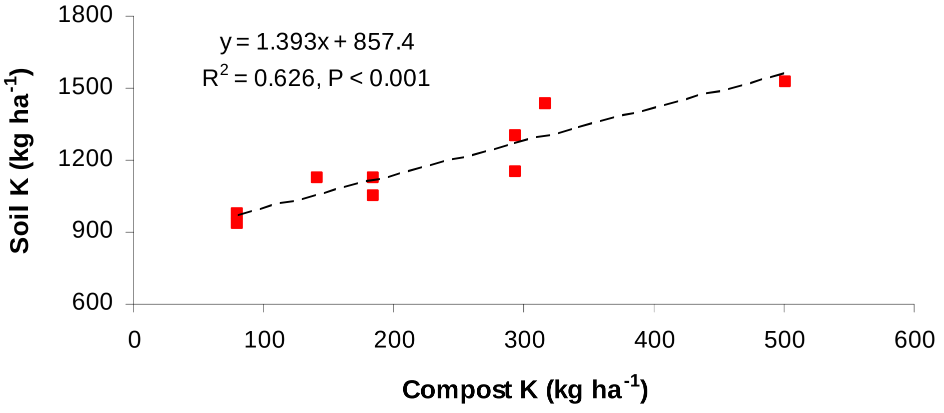

Figure 4.18 Soil extractable K levels as affected by two years of compost K addition.

The 2 years of compost application resulted in significant increase ($P<0.05$) of soil extractable Na content, as shown in Figure 4.19. Onion incorporated and biowaste surface applied composts at rates lower than 250 kg total N ha^{-1} did not increase significantly ($P>0.05$) the soil Na levels compared with the mineral fertilizer. The incorporation of biowaste compost even at the lowest rate significantly increased ($P<0.001$) soil Na in comparison with the reference treatments.

Considering a given rate of application, the biowaste compost applied on the soil surface and the onion compost incorporated resulted in similar levels of soil Na ($P>0.05$). With the exception of the lowest application rate, the biowaste compost incorporated resulted in significantly ($P<0.05$) higher soil Na than when surface applied. This finding indicates that the surface application of the biowaste compost resulted in lower availability of compost Na.

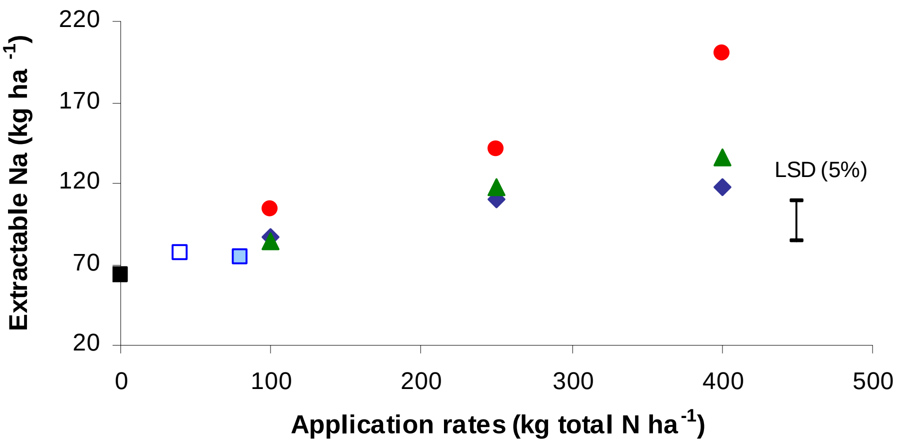

Figure 4.19 Soil extractable Na levels following two years of compost application (■ : Control, □ : Mineral, ◆: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated).

The effect of compost application rate was observed to be significant. The increase of the compost application rate resulted in significant increase ($P<0.001$) of the soil Na content. Figure 4.20 shows the relationship between compost Na added (sum of soluble compost Na in 2005 and 2006) and the soil Na levels in 2006. It can be seen that the soil Na content increases linearly with the addition of higher amounts of compost Na.

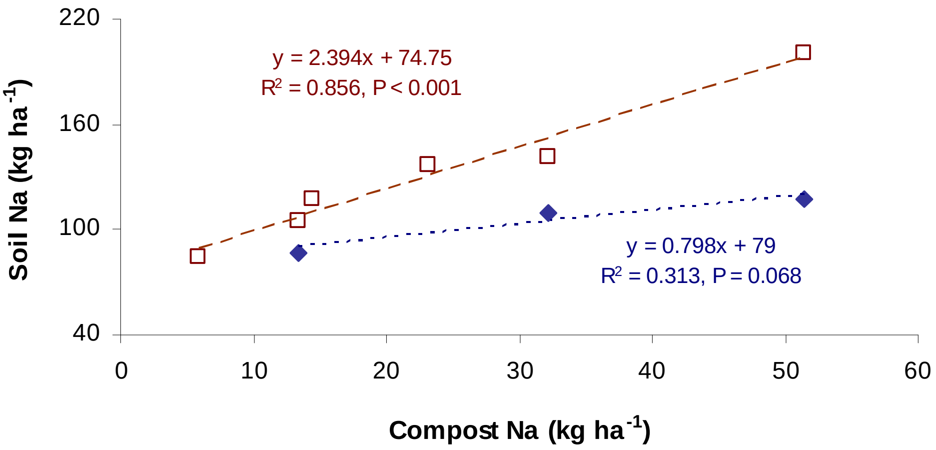

Figure 4.20 Soil extractable Na levels as affected by two years of compost Na application (☐ : Incorporated biowaste or onion compost, ◆: Surface applied biowaste).

The relationship between compost Na and soil Na is not significantly different for the incorporated biowaste and onion composts (P>0.05). However, the relationship between compost Na and soil Na is significantly different for the surface applied biowaste compost in comparison with the biowaste and onion composts incorporated (P<0.05). The two lines in Figure 4.20 represent these different relationships. The lower value of the slope of the surface applied compost line indicates the reduced response to compost Na when it is surface applied.

The increase of the soil Na content following the compost application could possibly raise problems with soil salinity and sodicity, especially in a year-after-year application scheme. After the first year of compost application the soil EC_e was not influenced by the different treatments. The average for all treatments was equal to 0.5 dS m^{-1}. The soil ESP was zero for all treatments.

After the second year of compost application, the analysis of the high application rates of the different treatments showed that compost application and mineral fertilization significantly increased soil EC_e ($P<0.05$). The onion compost resulted in the highest soil EC_e ($P<0.10$), which was equal to 1.4 dS m^{-1}, while the control was 0.6 dS m^{-1}. An EC_e value of 1.4 dS m^{-1} is below the limit of 4 dS m^{-1} which is critical for most crops.

However, this result may indicate potential salinity problems following year-after-year compost application at high rates.

Soil ESP was found significantly increased ($P<0.05$) by the biowaste compost incorporation only, supporting the findings shown in Figure 4.19. The ESP of the control, onion and mineral fertilizer treatments remained equal to zero. The incorporated biowaste compost resulted in ESP equal to 0.4%, whereas the surface applied biowaste to 0.1%, which are both very low ESP values, considering that the ESP limit for sodicity problems is 15%. Despite that, these results indicate that soil salinity and sodicity should be monitored when composts applied to agriculture in consequent years to avoid deteriorating soil quality.

The soil volumetric moisture content was not significantly affected ($P>0.05$) by the compost application, as it can be seen by the soil-water characteristic curves shown in Figure 4.21. Nor was the available water content. The mean available water content of all treatments was calculated equal to 15.9% in volumetric basis. However, the soil volumetric moisture content expressed as the average of all soil water potentials was found significantly influenced ($P<0.05$) by compost application (Figure 4.22).

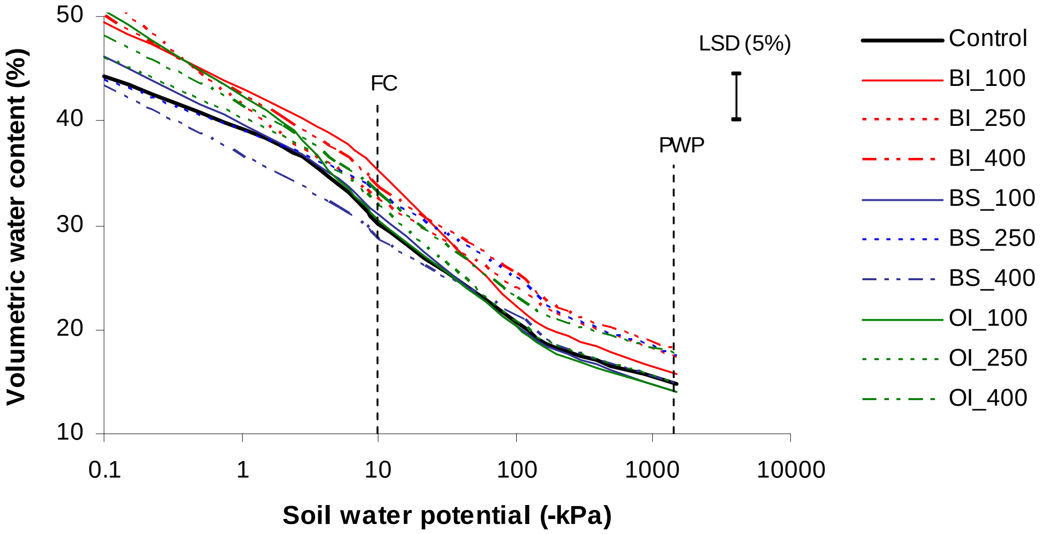

Figure 4.21 Water release curves (B: biowaste compost, O: onion compost; I: incorporated, S: surface applied; 100, 250, 400: application rates in kg total N ha^{-1}).

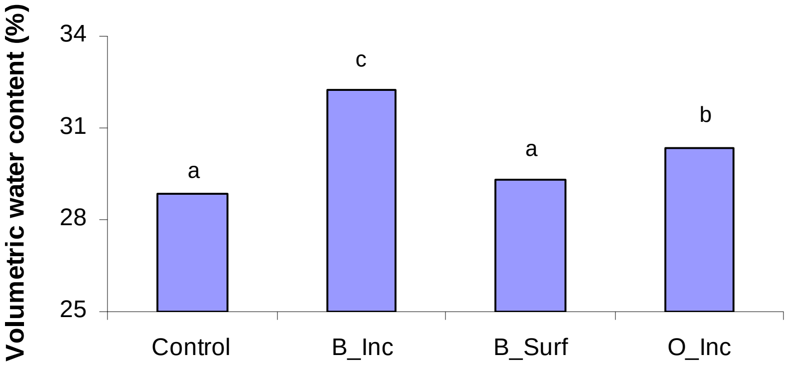

Figure 4.22 Average volumetric soil water content (B: biowaste compost, O: onion compost; Inc: incorporated, Surf: surface applied). Columns labelled with the same letter are not significantly different (P>0.05).

The surface application of biowaste compost had no significant effect (P>0.05), but the incorporation of biowaste and onion composts significantly increased (P<0.05) the average soil water content, indicating therefore that compost incorporation within the topsoil may increase soil water retention in the long term. This increase may be attributed to the improvement of soil structure, as shows the work undertaken by Aggelides & Londra (2000), who also found that compost incorporation increased soil water retention.

It needs to be noted that under field conditions, evaporation is a very important factor which affects soil water retention. In the method used to study the soil water retention, evaporation is controlled and does not influence the soil water content. The surface application of composts is shown to increase soil water retention mainly through the reduction of evaporation (Pinamonti, 1998; Agassi et al., 2004). It may be suggested, therefore, that compost incorporation may increase soil water retention mainly through improving soil structure, whereas compost surface application may increase soil water retention through the reduction of the evaporation.

4.4 General discussion

Compost application or mineral fertilization were found to have no significant effect ($P>0.05$) on forage maize yield and nutrient uptake, mainly due to the high soil fertility. The field was rich in phosphorus and potassium and the total nitrogen content was high and mineralized in rates that were found capable of covering the crop N requirements. The climatic conditions during the study period were positively influencing forage maize growth.

The total N content of the soil was 0.14% within the top 0.3 m depths, which is a high value of total N content of sandy loam soils, but within common limits. Total N content in sandy loam soils is reported to range between 0.10% and 0.15% (Gagnon & Simard, 1999; Chadwick et al., 2000; Nevens & Reheul, 2003; Chaves et al., 2005a). The increased N mineralization observed during the study is associated with the low C: N ratio of the soil (C: N =10). Soil C: N ratio of 15 is suggested to be a critical limit separating soil groups with higher or lower N release (Springob & Kirchmann, 2003).

The high mineralization of the soil inherent N may be explained taking in consideration the history of the site, which involves ploughing up grassland. Increase in N mineralization following ploughing up grassland has been reported, especially the first years following the ploughing. More N release is likely from grassland of increasing age (Johnston et al., 1994; Eriksen et al., 2006).Whitmore et al. (1992) showed that approximately 2000 kg N ha^{-1} were mineralized within the first 5.5 years following permanent grassland, whereas a total of 4000 kg N ha^{-1} were mineralized within 20 years. The increase in N mineralization, however, may persist for 25 years (Whitmore et al., 1992) or more than 27 years (Richter et al., 1989). The field, where this experiment took place, was permanent grassland and it was ploughed up in 1980. Since then it was left predominantly in fallow. The weeds grown were added back to the soil by annual ploughing, hence possibly enhancing the easily mineralizable soil N. The low C: N ratio of the soil could indicate that soil N mineralization may persist for more years.

Compost application on such a good quality soil, with high N content and mineralization potential, may introduce risks of increased N leaching. As the soil itself is capable of covering the crop N requirements, the compost nitrogen potentially mineralized may be in excess of crop requirements and thus susceptible to leaching. Care, therefore, should be taken to adjust compost application so as the soil available N during the crop growing season does not exceed the crop N requirements. The use of low rates and/or composts with low mineralization potential (e.g. the biowaste compost with C: N of 20) might reduce the possibility of increased N leaching following compost application to soils with high mineralization potential of inherent N.

Furthermore, compost application in high rates was shown to increase the electrical conductivity and Na content of a light textured (sandy loam) soil. Although the ESP of the soil was low, this finding suggests that consecutive compost application is capable of introducing salinity and sodicity problems, which needs to be taken into consideration when composts are to be applied to salinity sensitive soils and crops. Another important feature to be taken in consideration is that long term compost application is likely to increase water retention. Increasing the soil water retention is of great significance in light textured soils, and arid or semi-arid areas, as it is likely to result in both improved crop growth and water conservation.

4.5 Conclusions

The conclusions drawn from the two years field study can be summarized as follows:

a) Biowaste and onion compost application had no significant effect on forage maize production, mainly due to the mineralization of background levels of soil inherent N and its utilization by the crop. The increased soil N mineralization following ploughing up grassland was shown to persist for more than 26 years.
b) Mineralization of onion compost N, indicating the N fertilizer potential of composts with C: N ratio equal to 10.
c) Limited mineralization or immobilization of biowaste compost N, indicating that composts characterised by a C: N ratio of 20 should not be considered as N fertilizer alternatives or supplements.

d) High rates of compost application increases soil extractable K levels, thus increasing soil fertility and reducing the mineral fertilizer K requirements for crop production.

e) There is evidence to suggest that long term compost application is likely to increase soil water retention, hence increasing crop production and water conservation, especially in arid areas.

f) Consecutive biowaste and vegetable waste compost application to agricultural soils in high rates is likely to result in salinity and/or sodicity problems in sensitive soils (such as heavy soils, or coastal areas), or reduced yields of salinity sensitive crops.

5 Pot experiment: effects on nitrogen leaching, crop production and soil properties

5.1 Introduction

Chapter 5 describes the pot trial, which was undertaken as a supplementary study to the field experiment. The pot trial aimed predominantly at providing a more representative estimate of the nitrogen leaching potential of the biowaste and onion composts, following their application on sandy loam soil.

In the field experiment the environmental risk of nitrogen leaching was estimated by monitoring the soil nitrogen levels. In the pot experiment the nitrogen leaching from compost amended soil was assessed by the measurement of the amount of nitrogen actually leached, under similar conditions to those of the field study.

The pot experiment aimed to identify the effect of compost type, application rate and method on:

1. The nitrogen availability to crop production from two years of biowaste and vegetable waste compost application,
2. The environmental risk from excessive nitrate leaching through measurements of the mineral N leached, and
3. The soil properties.

5.2 Materials and methods

5.2.1 Pot experiment

The pot experiment was carried out in two periods. The first period was in 2005 and involved the production of forage maize on compost amended sandy loam soil. The study was conducted over about 4 months under greenhouse environment and irrigation conditions, corresponding to the growing season of forage maize in the field (from May to September).

The experimental design of the pot study involved the use of two compost types: biowaste and onion composts; three application rates: corresponding to 80, 165 and 250 kg total N ha^{-1}; and two application methods: incorporation and surface application. A control (non-amended soil) and a mineral fertilizer treatment (30 kg N ha^{-1} (ammonium nitrate) and 74.4 kg K_2O ha^{-1} (sulphate of potash)) were used as reference treatments. The pot trial was established on May 9, 2005 as a complete randomized block design with three replicates.

Composts were applied to 10 L pots, filled with 10.2 kg sandy loam soil collected from the field site. A piece of cloth was placed at the bottom of each pot in order to prevent soil losses. The rates of compost and mineral fertilizer applied per pot were calculated on a volumetric basis assuming a soil depth of 0.15 m and a bulk density of 1.44 kg l^{-1}. Tables 5.1 and 5.2 present the amount of composts (fresh material, N, P, K and OM) and mineral fertilizer (N, P, K) applied per pot. Some soil fertility properties at the beginning of the experiment were presented in Table 4.1 in Chapter 4. Forage maize (*Zea Mays* L.) was used as the monitoring crop.

Crop sowing was at a rate of 3 seeds per pot. The crop was thinned to 2 plants per pot whilst at the 3rd leaf stage (on June 1, 2005). Compost incorporation was prior to seeding and it was carried out within the top 4.5 kg of soil. After the sowing, composts and mineral fertilizers were applied to the soil surface of the respective pots. The pots were placed on weld mesh to allow for free drainage and easy collection of leachate. The leachate was collected in saucers placed under the weld mesh below each pot.

Drip irrigation with tap water was applied to all pots during the crop growing season, aiming to meet the water requirements of the crop (so water was not limiting crop production). The same amount of water was applied to all pots and care was taken to avoid/minimise leachate production between the leachate samplings. The soil water content was kept close, but below the FC, and the amount of the water added by irrigation was increased in the date of leachate sampling in order to produce the leachate sample. Leachate samples were collected by sub-sampling from the total amount of leachate produced. Leachate volume was determined on the remaining amount of leachate using a volumetric cylinder.

Crop harvest was on August 23, at the plants physiological maturity. After the completion of the study (on September 6, 2005) the pots were left intact on site and the soil was left to air-dry. Figure 5.1 presents the pot trial in 2005, whilst forage maize was at the 4th-5th leaf growth stage.

Figure 5.1 Pot trial on June 11, 2005: Forage maize production (at 4th-5th leaf stage) on compost amended sandy loam soil under greenhouse conditions. Leachate collection was from the saucers below each pot.

The second period of the pot trial was in 2006 and involved the production of annual ryegrass on the previously compost amended sandy loam soil, which was left to air-dry. The methodology of the pot study in 2006 was adjusted in respect to the results obtained from the pot and the field experiment in 2005.

Smaller pots were used (3L) in 2006, which were filled with 2.75 kg soil sub-sampled from the respective 10L pots used in 2005 (The air-dried soil left in each 10L pot was thoroughly mixed, sub-sampled and used to fill in the respective 3L pot. This was necessary as significant amounts of soil were removed for soil analysis following maize harvest). The soil was placed over a thin layer of gravel (250 g) to facilitate drainage. On the bottom of the pots a piece of cloth was placed to prevent soil and gravel losses. The experimental design in the second period of the pot trial was modified to include another rate of mineral fertilizer N (the pots receiving the extra mineral fertilizer N rate were filled with soil from the control pots).

The application rates of both composts and mineral fertilizer were changed. Compost application rates in 2006 were increased to 100, 250 and 400 kg total N ha^{-1} (the lower rate in 2006 (100 kg total N ha^{-1}) was applied to the pots receiving the lower rate in 2005 (80 kg total N ha^{-1}), and respectively for the other two rates). Mineral fertilizer N rates were also increased to 0, 180 and 360 kg N ha^{-1}, which are very high rates. Table 5.1 summarizes the compost application rates and the respective amounts of macro-nutrients (N, P, K) and organic matter applied each year. Table 5.2 shows the amounts of mineral N, P and K applied to the mineral fertilizer treatments each year.

The second period of the pot trial started on May 12, 2006 and it was conducted over 7 months. The trial was carried out outdoors during the summer months and in the greenhouse during the winter months. The crop used was annual ryegrass (*Lolium multiflorum* Lam.), which was selected as it can allow for the monitoring of N availability through harvesting at regular inervals. Compost incorporation was prior to sowing and it was carried out within the top 1 kg of soil. Ryegrass was sown at a rate of 1.50 g per pot. Composts and mineral fertilizers were applied to the soil surface of the respective pots the day following ryegrass sowing (June 1, 2006). The crop was harvested 4 times. The first harvest was on August 16, 2006, when the crop was enough developed (average ryegrass height was 0.16 m) to ensure that after the harvest it would continue its growth. The latest harvest was on December 12, 2006.

Table 5.1 Biowaste and onion composts: application rates and respective amounts of nutrients and organic matter applied per pot, and composts C: N ratio for the two years of the pot experiment.

Application rates (kg ha^{-1})	**Fresh compost** (g pot^{-1})	**Total N** (g pot^{-1})	**Soluble P** (mg pot^{-1})	**Soluble K** (g pot^{-1})	**Soluble mineral N** (mg pot^{-1})	**OM** (g pot^{-1})	**C: N**
2005 - 10 L pots							
Onion compost							15.9
80	120.0	0.38	3.59	0.38	1.63	12.9	
165	247.5	0.78	7.41	0.78	3.35	26.5	
250	375.0	1.18	11.22	1.19	5.08	40.2	
Biowaste compost							14.6
80	37.3	0.38	2.86	0.15	5.62	9.4	
165	77.1	0.78	5.91	0.30	11.61	19.5	
250	116.6	1.18	8.93	0.46	17.56	29.5	
2006 - 3 L pots							
Onion compost							10.5
100	30.1	0.13	7.56	0.10	2.76	2.9	
250	75.3	0.34	18.91	0.24	6.91	7.2	
400	120.4	0.54	30.25	0.39	11.05	11.6	
Biowaste compost							20.1
100	11.0	0.13	2.66	0.06	1.48	5.3	
250	27.5	0.34	6.64	0.16	3.71	13.3	
400	44.0	0.54	10.63	0.25	5.93	21.3	

Table 5.2 Mineral fertilizers N, P and K applied to each reference treatment in 2005 and 2006 (30, 180, 360: mineral fertilizer N application rates in kg N ha^{-1}).

Treatment	**Mineral fertilizer**		
	N (g pot^{-1})	**P** (mg pot^{-1})	**K** (g pot^{-1})
2005 -10L pots			
Control	0	0	0
Mineral_30	0.14	0	0.29
2006- 3 L pots			
Control	0	0	0.09
Mineral_180	0.23	0	0.09
Mineral_360	0.46	0	0.09

The pots were placed again on weld mesh to allow for free drainage and easy collection of leachate. The leachate was collected in open buckets, placed under the weld mesh below each pot. Each leachate sample represented a bulk sample of the leachate produced between two consecutive sampling dates. Drip irrigation with tap water was applied to all pots during the experiment, aiming to keep the soil moisture close to field capacity, as ryegrass is sensitive to drought. Whilst the experiment was held outdoors a polyethylene sheet was used to ensure that the leachate collected in the buckets is not contaminated with rain water (Figure 5.2).

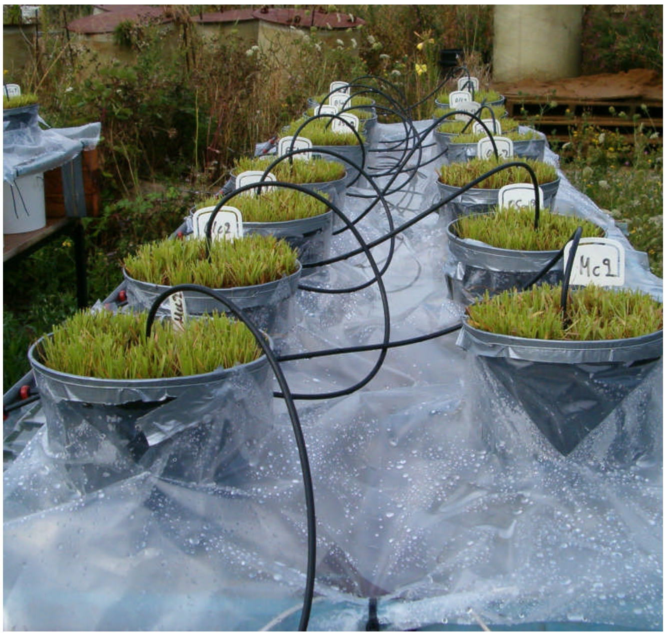

Figure 5.2 The second block of the pot trial after the first harvest of ryegrass (August 16, 2006). The polyethylene sheet prevents rainfall to contaminate the leachate.

Further details regarding the pot experiment establishment, layout, management and temperature conditions in 2005 and 2006 can be found in Appendix A.3.2.

5.2.2 Leachate, crop and soil measurements and analysis

5.2.2.1 Leachate

In 2005, leachate sampling was undertaken every 10 days starting from May 11 to August 20. Mineral N (TON-N and NH_4-N) determination in the leachate was carried out by segmented flow analysis (SFA). In 2006, leachate sampling was carried out every 2 weeks, starting from June 15, until December 11. Measurements of mineral N (TON-N and NH_4-N) in the leachate were undertaken on all samples, whereas for total dissolved N (TDN) on the samples collected after the 13th July 2006. Determination of TDN in the leachate was carried out by SFA.

The amount of mineral N leached was estimated as a percentage of the total amount of N applied according to Equation 5.1.

$$N\ leached\ (\%) = 100 \times \frac{Fertilizer\ or\ Compost\ N\ leached\ - Control\ N\ leached}{Total\ Ferilizer\ or\ Compost\ N\ applied} \qquad (5.1)$$

The amount of dissolved organic N (DON) in the leachate was estimated as the difference between the TDN and the mineral N measured for each treatment and sampling date (Equation 5.2).

$$TDN = DON + Mineral\ N \qquad (5.2)$$

5.2.2.2 Crop

5.2.2.2.1 Forage maize

Forage maize growth was monitored by measuring the number of complete leaves, the plant height and the shoot diameter. A leaf was considered to be complete when its collar was completely formed (Easson & Fearnehough, 2000). The plant height was defined as the distance from the back of the collar of the upper complete leaf to the soil surface. Shoot diameter was measured at a distance of 0.06 m from the soil surface.

At harvest, crop above ground dry matter (DM) yield was determined by drying maize cobs at 60 ˚C until a constant weight and the rest of the plant at 105 ˚C for 26 hours. Maize total N concentrations were determined separately for the cobs and the rest of the maize plant by dry combustion using Vario EL elemental analyzer. The DM yield of maize cobs and of the rest maize plant were multiplied by the respective total N content, and then summarized to crop N uptake.

Maize total P and K concentrations were determined for the whole maize plant according to MAFF (1986). Crop P and K uptake were also evaluated by multiplying the whole maize DM yield by the measured P and K concentrations, respectively. Crop DM yield and N uptake were also measured for the plants which were removed at thinning to achieve more accurate calculation of the N budget. N recovery from compost and mineral fertilizer was calculated according to the Equation 5.3 (Hartl & Erhart, 2005).

$$N\ re\mathrm{cov}ery\ (\%) = 100 \times \frac{Fertilizer\ or\ Compost\ N\ uptake\ -\ Control\ N\ uptake}{Total\ Ferilizer\ or\ Compost\ N\ applied} \qquad (5.3)$$

5.2.2.2.2 Annual Ryegrass

The annual ryegrass was harvested four times: at 9, 15, 22 and 26 weeks after crop emergence (4 cuts: August 16, September 28, November 14, and December 12, 2006) by cutting the plant tops 0.03 m above the soil surface. At each harvest time, crop dry matter (DM) yield was determined by drying the ryegrass tops at 60 ˚C until constant weight. Crop total N concentration was determined by dry combustion using Vario EL elemental analyzer. The crop DM yield was multiplied by the respective total N content, and then summarized to crop N uptake. Ryegrass N recovery from compost and mineral fertilizer was calculated according to the Equation 5.3.

5.2.2.3 Soil

Soil quality properties were determined after the crop harvest in both years of the study. Organic matter content (OM) was determined by the loss on ignition method, pH in the 1/2.5 soil/water suspension, total N, total C and C: N ratio by the use of Vario EL elemental analyzer. Available P was estimated by the Olsen method and measured by spectrophotometer. Extractable K and Na were extracted by ammonium nitrate and measured by flame photometry (MAFF, 1986). Soil ammonium (NH_4) and total oxides of nitrogen (TON) -nitrogen content was measured in potassium chloride extracts by segmented flow analysis. The amount of soil residual TON-N after the crop harvest each year was used to estimate potential environmental threats due to excessive nitrate leaching. Soil salinity and sodicity were estimated by measuring the electrical conductivity in the saturated paste extract (EC_e), and the exchangeable sodium percentage (ESP) from the sodium adsorption ratio (SAR) according to Richards (1954).

Details of the methods involved in the soil and crop sampling and analysis can be found in Appendix A.2.

5.2.3 Statistical analysis

The effects of each treatment and the influence of compost type, application rate and application method on the measured variables were assessed by analysis of variance (at 5% probability level) and the relation between variables was established by regression analysis (at 5% probability level) using the statistical package GenStat (GenStat, 2006).

The accuracy of the statistical analysis was checked graphically by the residual plots. When the residuals did not follow the normal distribution then the data were normalized by applying a square root or a natural logarithm transformation. Also, in cases of an extreme value in the data set, this value was removed from the analysis when this data point caused the residuals to be not normal.

5.3 Results and discussion

5.3.1 First year of compost application (2005)

5.3.1.1 Forage maize production

5.3.1.1.1. Growth measurements

In general, the growth measurements of the forage maize plants showed that forage maize growth was significantly affected by the application of composts and mineral fertilizer. Figure 5.3 presents the mean number of complete leaves, plant height, and shoot diameter of all measuring dates, grouped by the type and method of compost application, as the co-effect of compost type and application method was found significant ($P<0.05$) for all three growth parameters. Figure A.3.2.5 (in the Appendices) shows the number of complete leaves, plant height and shoot diameter (on average of all measuring dates) as affected by each treatment.

Figure 5.3 shows that the control treatment tends to result in lower growth than the mineral fertilizer treatment. However, the differences between these two treatments were found significant ($P<0.05$) only for the plant height.

An interesting feature shown in Figure 5.3 is that the effect of compost application on crop growth is not consistent for the different growth parameters. In particular, the surface applied in comparison to the incorporated onion compost, resulted in significantly lower number of complete leaves ($P<0.001$) and shoot diameter ($P<0.05$), but also in significantly higher plant height ($P<0.001$). The effect of the method of biowaste compost application on the shoot diameter and the number of complete leaves was found not to be significant ($P>0.05$). However, the incorporated biowaste compost resulted in significantly higher ($P<0.05$) plant height than when surface applied.

Although there were significant differences ($P<0.05$) between the treatments, the effect of compost application on crop growth was not clear, due to the variation of the effect on the different growth parameters. Consequently, in order to properly monitor the growth of a crop, different growth parameters need to be considered (e.g. height and diameter), especially when plants are grown under limiting conditions (e.g. pots, where the soil volume is limited).

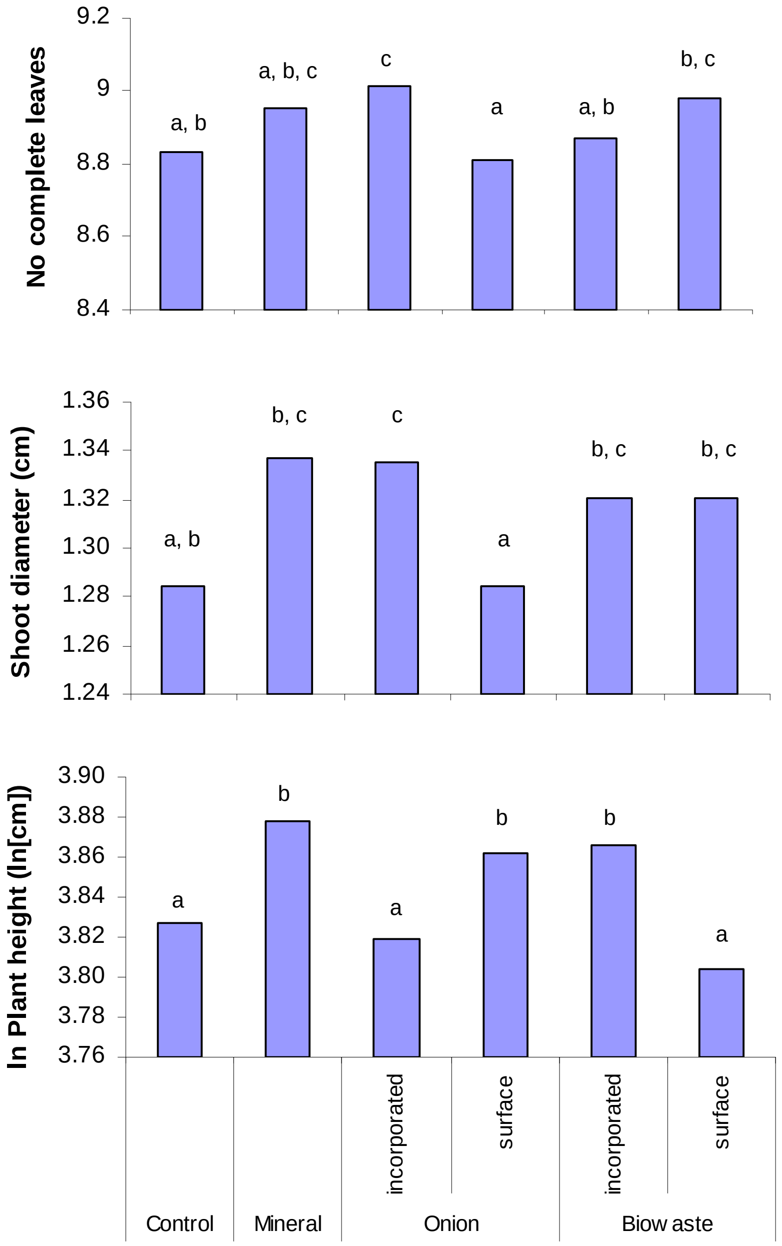

Figure 5.3 The effect of type and method of compost application on forage maize number of complete leaves (top), shoot diameter (middle) and the natural logarithm of plant height (bottom) in the first year of compost application, in comparison to the reference treatments. The values shown are the means for the whole measuring period. Columns labelled with the same letter are not significantly different (P>0.05).

5.3.1.1.2 Forage maize dry matter yield and nutrient uptake

Figure 5.4 presents the crop dry matter yield as affected by the compost and mineral fertilizer application. It can be seen that the control treatment resulted in similar (P> 0.05) DM yield to all other treatments. The effects of compost type and application rate on maize DM yield were found not to be significant (P>0.05). However, it was shown that the incorporation of composts resulted in significantly higher (P<0.05) DM yield than the surface application.

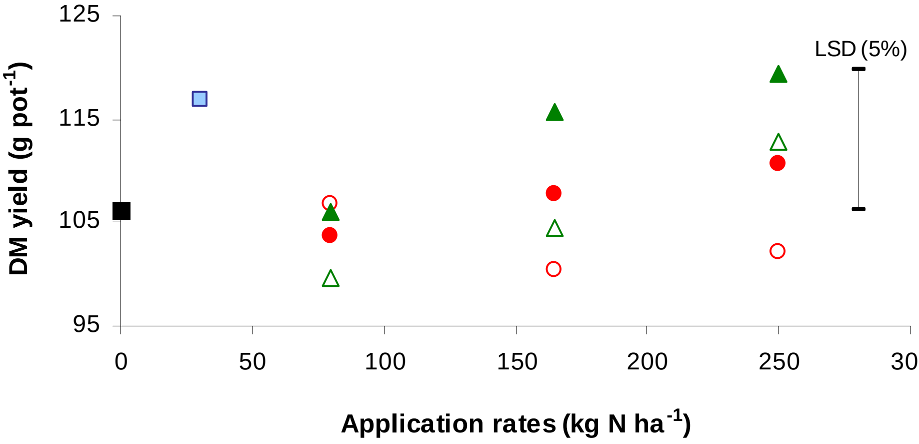

Figure 5.4 Forage maize dry matter yield response to compost and mineral fertilizer application at different rates (■ : Control, ◻ : Mineral, ○: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated, △: Onion surface applied).

N uptake was not significantly influenced (P>0.05) by compost or mineral fertilizer application (Figure 5.5). The mean N uptake value was found equal to 0.51 g pot^{-1}.

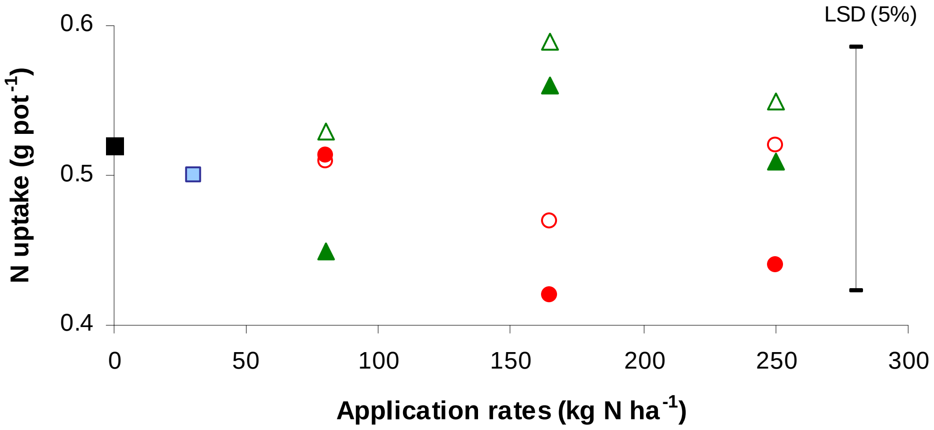

Figure 5.5 Forage maize N uptake resulting from compost and mineral fertilizer application at different rates (■ : Control, ▢ : Mineral, ○: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated, △: Onion surface applied).

P and K uptake were not significantly influenced (P>0.05) by compost or mineral fertilizer application, in comparison to the control. However, there was a trend for higher K uptake resulting from the application of mineral fertilizer K (0.35 g pot^{-1}, see Table 5.2) in comparison to the control (Figure 5.6).

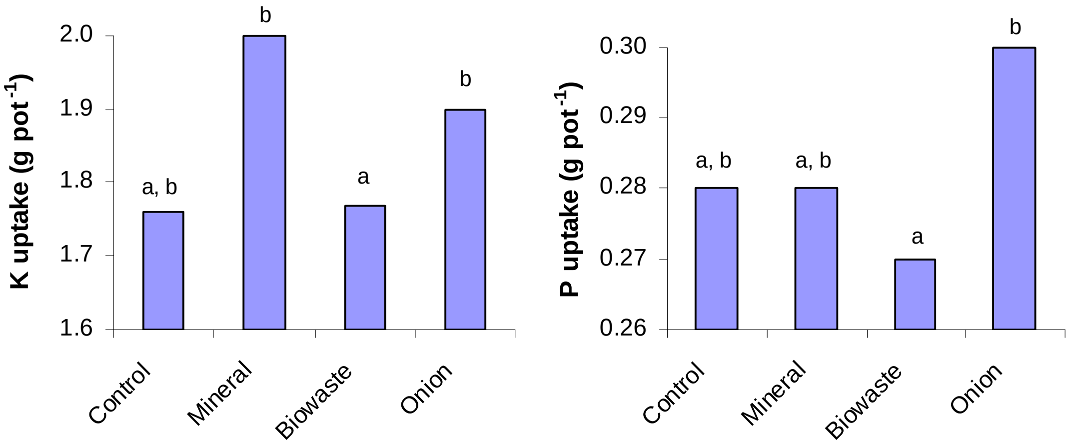

Figure 5.6 Forage maize K uptake (left) and P uptake (right) in the first year of compost application. Columns labelled with the same letter are not significantly different (P>0.05).

No fertilizer P was applied to the mineral fertilizer treatment and it can be seen that the P uptake from the control and mineral fertilized soil are almost the same. P and K uptake were found to increase significantly ($P<0.05$) with the application of onion compost in comparison to the biowaste compost (Figure 5.6). K uptake was also influenced by the method of application used. Compost incorporation significantly increased ($P<0.05$) crop K uptake, in comparison to the surface application, by 5.8%.

The crop DM yield and N uptake results are in good agreement with the field experiment results which demonstrated that the crop yield was not significantly influenced by the compost or mineral fertilizer application, and also that the control soil was capable of supplying the necessary N to cover the crop N requirements.

5.3.1.2 Nitrogen leaching

The mineral N leaching losses resulted from the different treatments over time are presented in Figure 5.7. The majority of N losses occurred during the first month of the experiment for all treatments. After this period, all treatments produced minimal amounts of mineral N in the leachate.

Figure 5.8 shows the total amount of mineral N leached during the experiment from the different treatments. The application of the mineral fertilizer significantly increased ($P<0.05$) the mineral N losses compared to the control. The application of biowaste compost, either with incorporation or surface application, resulted in similar amounts of mineral N in the leachate with the control soil ($P>0.05$). The onion compost incorporated did not significantly increase the amount of mineral N leached compared to the control ($P>0.05$), whereas the surface application of onion compost resulted in significantly higher ($P<0.05$) mineral N losses when applied at rates higher than 80 kg total N ha^{-1}.

Increasing the rate of onion compost application resulted in higher mineral N losses ($P<0.05$). The biowaste compost application rate did not significantly increase the amount of mineral N leached ($P>0.05$). The surface application of the onion compost at 250 kg total N ha^{-1} significantly increased ($P<0.05$) the mineral N leached in comparison to all treatments.

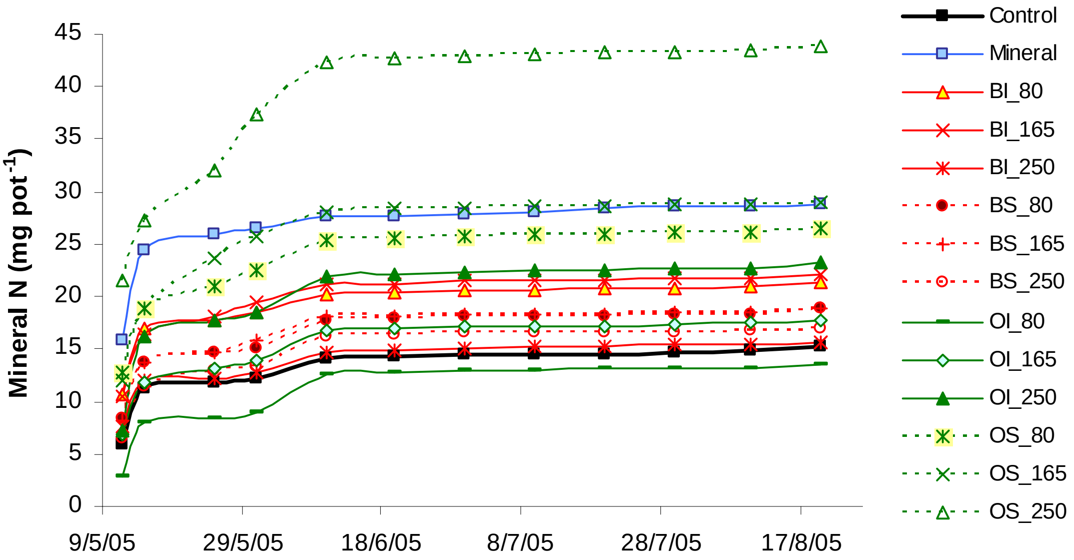

Figure 5.7 Cumulative mineral N leached from all 14 treatments (BI: biowaste compost incorporated, BS: biowaste surface applied, OI: onion compost incorporated, OS: onion surface applied; 80, 165, 250: application rates in kg total N ha^{-1}).

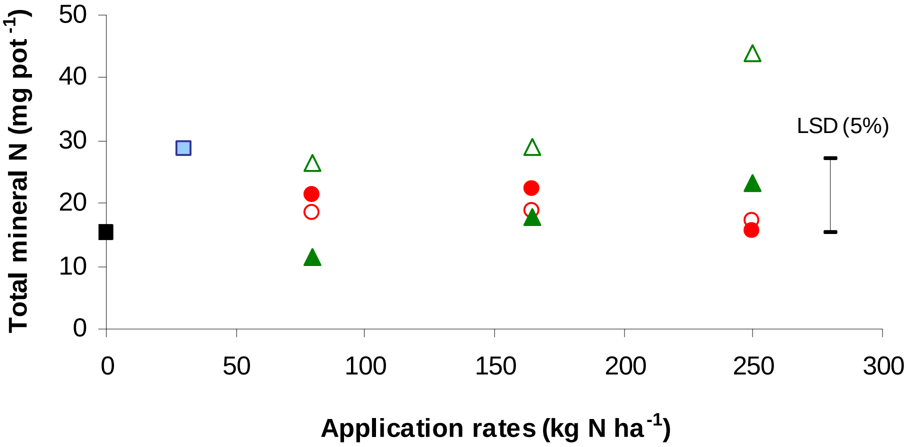

Figure 5.8 The effect of compost type on the total amount of mineral N leached in comparison to the reference treatments (■ : Control, □ : Mineral, ○: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated, △: Onion surface applied).

The amount of mineral N leached from the mineral fertilizer amended soil calculated as a percentage of the mineral fertilizer N applied was found equal to 9.6 %. The amount of the mineral N leached from the compost amended soil calculated as a percentage of the total compost N applied was found to range from -1.0% to 3.0% for the onion compost, and from 0% to 1.6% for the biowaste compost, depending on the compost application method and rate.

5.3.1.3 Soil properties

The soil organic matter (OM) content was also significantly increased ($P<0.05$) following the compost incorporation, by 8.2% in comparison to the control soil. This is an important finding as it shows the potential of compost incorporation, even in relatively low rates, to increase soil OM content. The surface application of composts did not increase the soil OM ($P>0.05$) compared to the control soil (Figure 5.9).

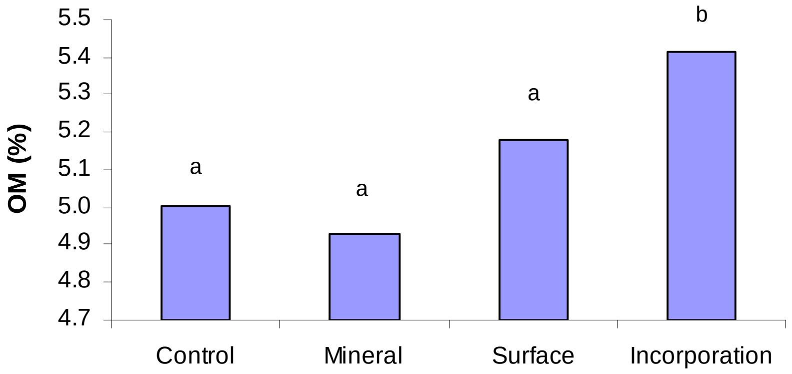

Figure 5.9 The effect of compost application method on the soil OM content. Columns labelled with the same letter are not significantly different ($P>0.05$).

Compost incorporation significantly increased ($P<0.05$) the soil total N and C content, in comparison to the control and the compost surface application. As it is shown in Figure 5.10, the increase of total N follows the same pattern with the increase of total C. Consequently, soil C: N ratio was not significantly influenced ($P>0.05$) by any treatment (mean C: N ratio was 10.4).

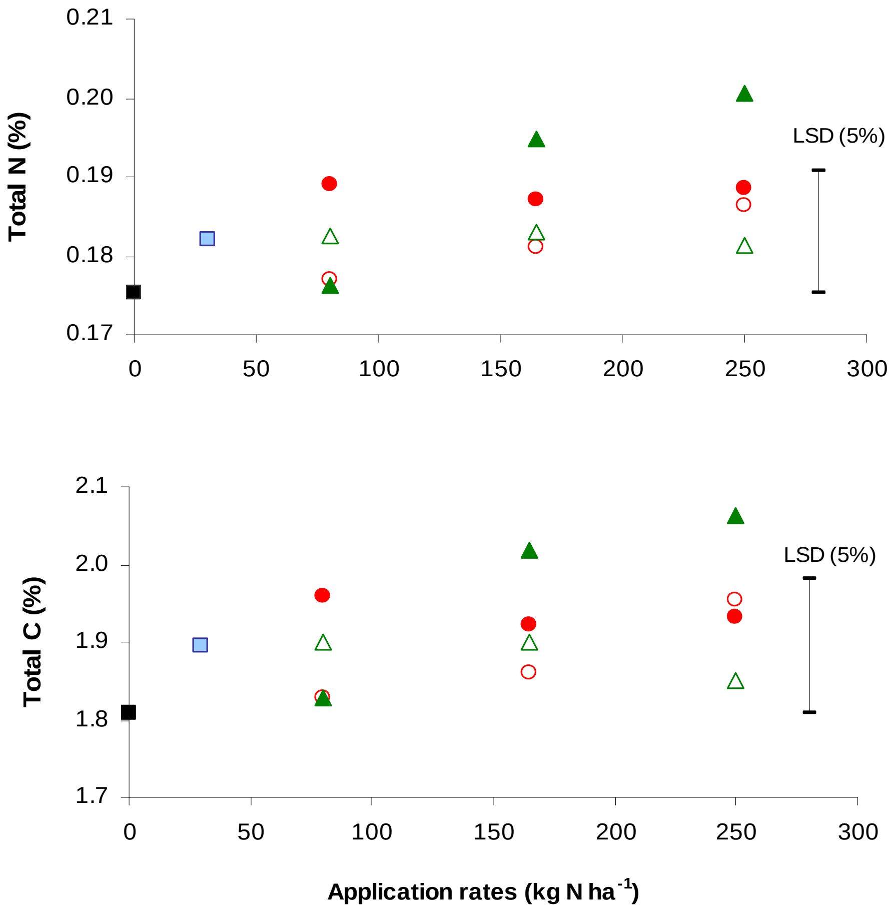

Figure 5.10 Soil total N (top) and total C (bottom) content following the first year of compost application (2005) (■ : Control, □ : Mineral, ○: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated, △: Onion surface applied).

Soil extractable P was not significantly influenced by compost application (P>0.05), but the extractable K content of the soil was found to increase with the compost application (P<0.05). Figure 5.11 presents the soil extractable K content for the different treatments.

Onion compost application significantly increased (P<0.05) the soil extractable K levels in comparison to the control. Biowaste compost application resulted in similar levels of K with the control (P>0.05), except when surface applied at the highest application rate. Soil K content was significantly increased (P<0.05) with the increase of compost application rate.

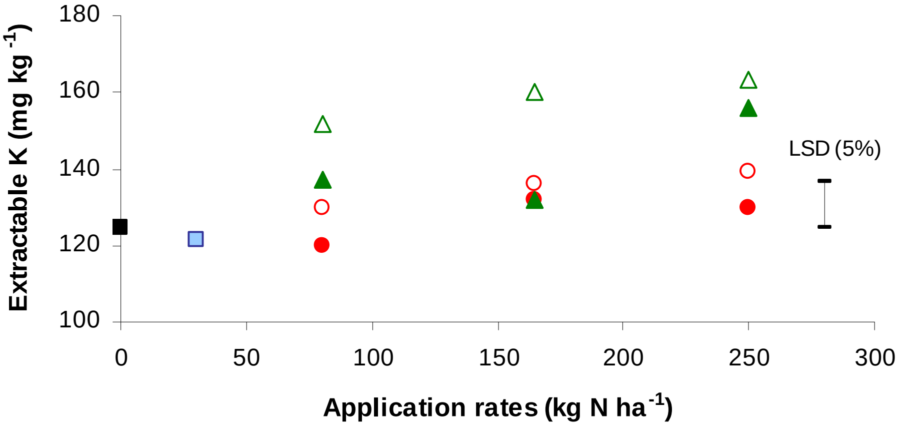

Figure 5.11 Soil extractable K content following the first year of compost application (■ : Control, □ : Mineral, ○: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated, △: Onion surface applied).

Surface application of composts resulted in significantly higher ($P<0.05$) soil K content than composts incorporation. Comparing the maize K uptake and soil K results indicates that the reason why the surface application of composts resulted in higher soil K content is because the crop was more efficient at taking up the K from the soil where the composts were incorporated. Figure 5.12 shows the soil extractable K levels response to the application of compost K. The sum of K uptake and soil extractable K was assumed to be indicative of the total compost K availability, and it was also plotted in respect of compost K applied (Figure 5.12).

Soil pH and EC_e were not significantly affected ($P>0.05$) by compost application (mean pH was 6.7 and mean EC_e was 1.7 dS m^{-1}). The ESP of the compost amended and mineral fertilized soil was not significantly different ($P>0.05$) from the control soil ESP (mean ESP was 2.3%). Despite that, the surface applied compost resulted in significantly lower ($P<0.05$) ESP than the incorporated compost and the mineral fertilizer treatment.

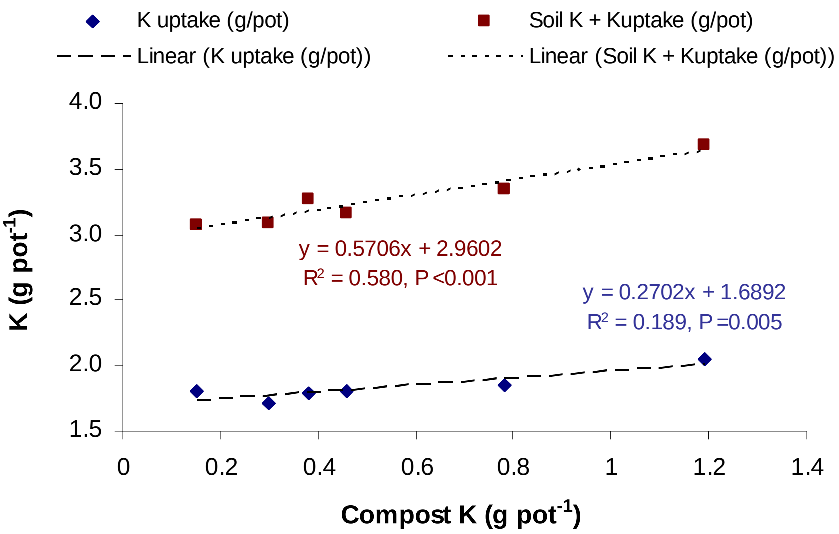

Figure 5.12 K uptake response to compost K application, and K uptake plus soil K response to compost K application ($P<0.05$).

Residual soil TON-N content was used as an indication of possible nitrate leaching losses after the crop harvest. TON-N content of all treatments was found to be at low levels, ranging from 0.14 to 1.13 mg kg^{-1}. The mean TON-N content of the compost amended soil was 69.3% higher than the control soil, but the difference was not significant ($P>0.05$). The effect of compost type, application rate and method were found to be not significant ($P>0.05$). The mineral fertilizer resulted in significantly lower ($P<0.05$) residual TON-N content than the composts. The soil NH_4-N content was found to be not significantly different ($P>0.05$) between the different treatments (mean NH_4-N content was 3.3 mg kg^{-1}).

During the whole experiment, all pots received the same amount of water. Consequently the moisture content of the soil at the time of sampling was considered indicative of the water retention capability of the soil. Soil moisture content (gravimetric) was found to be significantly increased ($P<0.05$) by the onion compost application. The surface application of composts also significantly increased ($P<0.05$) the soil moisture content in comparison to compost incorporation (Figure 5.13). The effect of compost application rate was not clear.

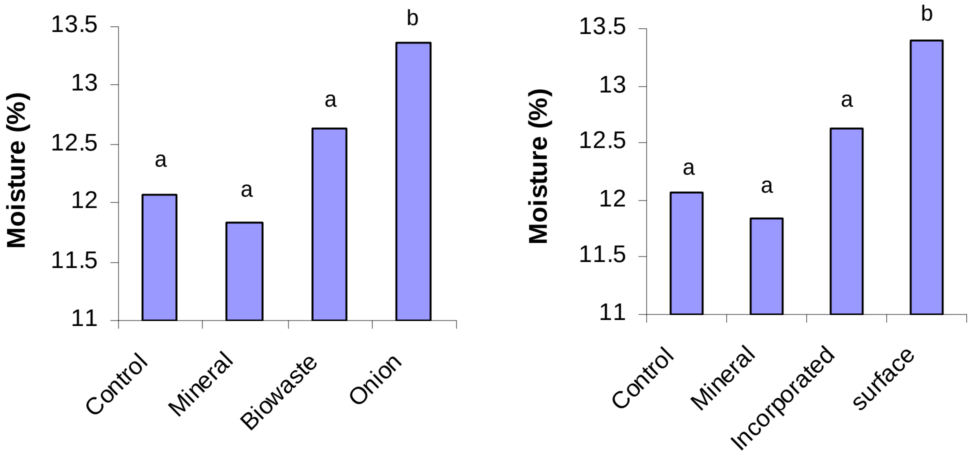

Figure 5.13 The effect of compost type (left) and application method (right) on the soil gravimetric moisture content. Columns labelled with the same letter are not significantly different (P>0.05).

5.3.1.4 General discussion of the first year results

The monitoring of forage maize growth did not clearly indicate the effects of compost and mineral fertilizer application on forage maize production, due to the variation of the effect on the different growth parameters. Forage maize dry matter yield was statistically similar for all treatments. However, the application of mineral fertilizer and compost incorporated at the higher rates tended to increase yield in comparison to the control soil. Onion compost resulted in higher P and K availability than the biowaste compost, as suggested by the P and K uptake results. However, statistically, all treatments resulted in similar P and K uptake to the control.

Compost N recovery was not possible to be calculated as the N uptake from the compost amended soil was at similar levels with the control. Crop N uptake was similar for all treatments, despite the amount of N which was added. The control soil was found capable of covering the crop N requirements because of the mineralization of inherent levels of soil N, as it was suggested by evidence obtained from the field experiment where the same soil was used. Since the control soil was capable of meeting the crop N requirements, the compost N which was mineralized and the mineral fertilizer N was in excess of the crop N requirements, and thus susceptible to leaching.

The majority of N leaching losses occurred during the first month following compost and mineral fertilizer application. After this period, all treatments produced minimal amounts of mineral N in the leachate. The surface application of the onion compost resulted in higher mineral N losses than the control, when applied at rates higher than 80 kg total N ha^{-1}. The application of mineral fertilizer N (30 kg N ha^{-1}) also increased the N leaching compared to the control. All the other treatments resulted in similar N leaching to the control. The surface application of the onion compost at 250 kg total N ha^{-1} significantly increased ($P<0.05$) the mineral N leached in comparison to all treatments.

The onion compost surface application resulted in higher amounts of available mineral N, in comparison to incorporation, as suggested by the higher amounts of N leached at the higher rates of compost application. One possible reason why the surface compost application resulted in higher N availability is likely to be related to the fact that surface applied organic materials tends not to be incorporated within the soil OM and consequently they mineralize faster (Smith et al., 1997). The soil total N and total C content were higher for the incorporated composts than the surface applied ones, thus providing support to the theory that the surface applied materials mineralize faster.

However, the N availability from the biowaste compost was not significantly affected by the method of application. The C: N ratio of the biowaste compost was lower than the onion compost one (14.6 and 15.9, respectively), and therefore higher amounts of mineral N were expected to be produced from the biowaste compost, as the C: N ratio is a commonly used indicator of the compost N mineralization potential (Sullivan et al., 2002; Wolkowski, 2003; Flavel & Murphy, 2006).

A factor which possibly affected the N mineralization of the composts is the quality of the compost C. Flavel & Murphy (2006) have shown that C quality parameters, such as the lignin content, also affect the compost N mineralization. The biowaste compost, as it contained garden waste, should have higher lignin content than the onion compost, and possibly this resulted in lower N mineralization than the onion compost. Actual measurements of the lignin content or other C quality parameters of the compost were not carried out. The mineral N content of both composts was very low and therefore it is unlikely to have influenced the amount of mineral N leached or taken up by the plant (the C: N_{org} ratio of the materials was not different from the C: N ratio).

There is evidence to suggest that the soil OM content increases even after the first year of compost application by incorporation. Soil water holding capacity appears to improve when composts are surface applied, as it was suggested by the measurement of the gravimetric soil moisture content at the time of sampling. This should be attributed to the lower evaporation losses resulting from the surface application (mulching effect), as it was demonstrated by Pinamonti (1998) and Agassi et al. (2004). Compost application generally increased the soil K content in comparison to the control soil. It was shown that the surface application of composts increased the soil extractable K levels, but at the same time decreased the crop K uptake, in comparison to compost incorporation. This finding indicates better use efficiency of compost K with the compost incorporation.

In general, the effects of compost application on forage maize production shown in the pot experiment are in good agreement with the results shown in the field. However, the effects of compost application on soil properties were not obvious in the field experiment in the first year of the application. This was mainly because the effect of compost application on soil properties was studied within the top 0.3 m depths in the field, whereas in the pot experiment it was studied within the top 0.15 m depths, and within the relatively short term application of composts during this study it was shown that soil properties within the top 0.3 m depths are slightly affected, whereas more changes are obvious within the top 0.15 m depths. Furthermore, due to limited soil volume in the pots there is less variability and the effects of the different treatments were clearer.

5.3.2 Second year of compost application (2006)

5.3.2.1 Annual ryegrass production

The drought and high temperature conditions during June delayed the emergence and the establishment of the crop. Ryegrass emergence first started on the pots where composts were surface applied. The crop was well established in all treatments towards the end of June. Ryegrass dry matter yield as affected by mineral fertilizer, biowaste and onion compost application is presented in Figure 5.14, in response to the rates applied in the second year of the experiment.

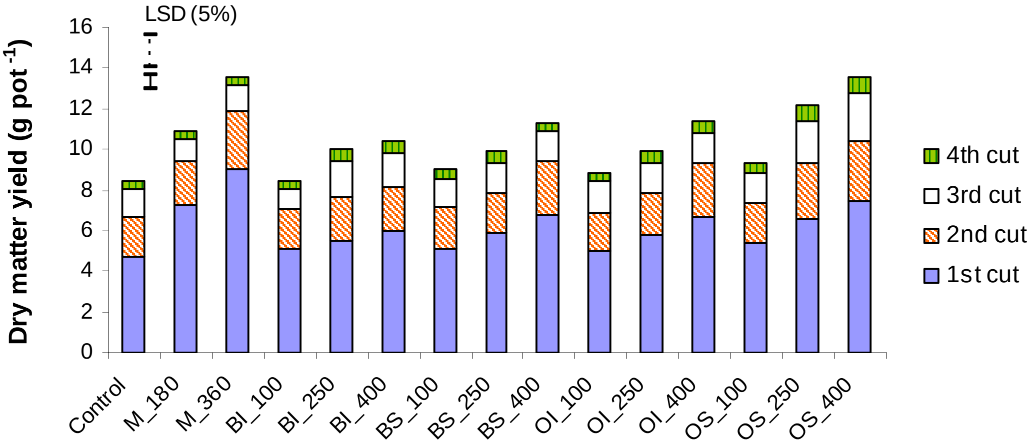

Figure 5.14 Ryegrass dry matter yield for the 15 treatments. The dashed-line LSD bar is for the comparison between the whole columns (sum of all cuts), whereas the solid-line LSD bar is between the different cuts (M: mineral fertilizer, BI: biowaste incorporated, BS: biowaste surface applied, OI: onion incorporated, OS: onion surface applied; 180, 360: application rates in kg N ha^{-1}; 100, 250, 400: application rates in kg total N ha^{-1}).

In all four harvests, the ryegrass DM yield was significantly affected by the type, method and rate of compost application. Crop DM yield was significantly increased ($P<0.001$) with the increase of compost application rate. Compost application at 100 kg total N ha^{-1} resulted in similar crop DM yield to the control in all harvests ($P>0.05$). Compost incorporation resulted in significantly lower ($P<0.05$) crop DM yield in comparison to the surface application. Onion compost significantly increased crop DM yield compared to biowaste compost ($P<0.05$, except in the 1st harvest: $P<0.10$).

In the first harvest, application of mineral fertilizer N significantly increased ($P<0.001$) the crop DM yield compared to the control. The increased rate of mineral fertilizer N resulted in a significant increase ($P<0.05$) in crop DM yield. Mineral fertilizer N applied at the rate of 360 kg N ha^{-1} resulted in the highest yield of all treatments. In the second harvest, only the highest rate of mineral fertilizer N significantly increased ($P<0.05$) the crop DM yield compared to the control. The highest yield was produced from the mineral fertilizer at the rate of 360 kg N ha^{-1}, the onion surface applied at 250 and 400 kg N ha^{-1}, the biowaste surface applied and the onion incorporated at the rate of 400 kg total N ha^{-1}. In the following harvests, both rates of mineral fertilizer N resulted in similar levels of crop DM yield to the control ($P>0.05$). The highest yield was from the onion compost surface applied at 250 and 400 kg N ha^{-1}.

Considering the total ryegrass DM yield production during the study (Figure 5.15), the application of mineral fertilizer N significantly increased the crop DM yield. Increased mineral fertilizer N rate resulted in significant increase in DM yield ($P<0.001$). Compost application resulted in similar levels of total DM yield to the mineral fertilizer application at the rate of 180 kg N ha^{-1} ($P>0.05$). However, onion compost surface applied tended to increase ($P<0.10$) ryegrass DM yield in levels higher than those of fertilizer N at 180 kg N ha^{-1}.

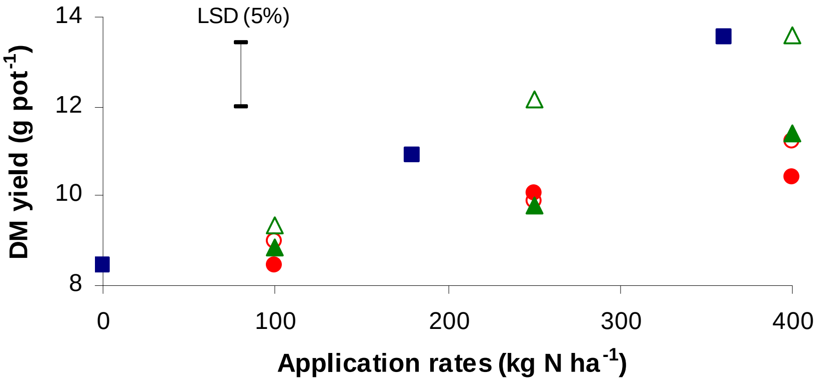

Figure 5.15 Ryegrass DM yield resulted from the compost and the mineral fertilizer amended soil (■ : Mineral fertilizer, ○: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated, △: Onion surface applied).

Ryegrass DM yield significantly increased ($P<0.001$) with an increase in compost application rate (Figure 5.16). The application of all composts at 100 kg total N ha^{-1} resulted in similar crop DM yield to the control ($P>0.05$). Compost surface application significantly increased ($P<0.05$) the crop DM yield compared to compost incorporation (Figure 5.16).

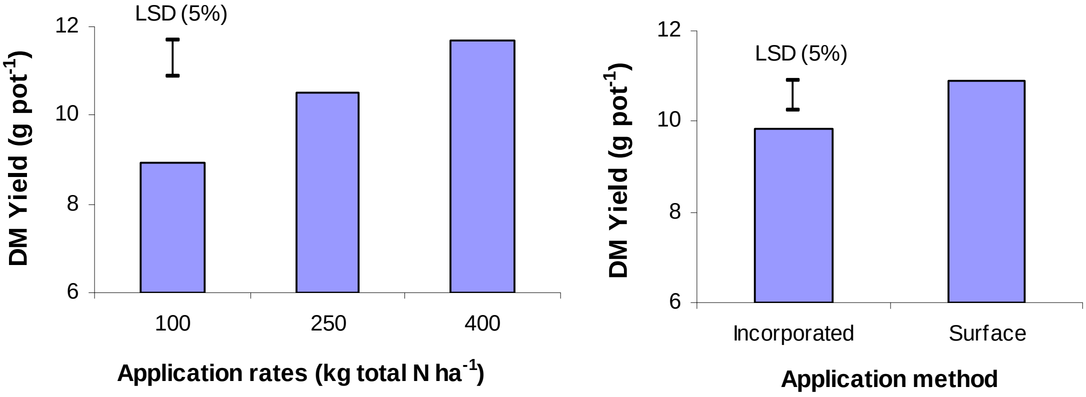

Figure 5.16 The effect of compost application rate (left), and method (right) on ryegrass DM yield.

Ryegrass N uptake response to different compost application and mineral fertilization treatments is presented in Figure 5.17. In the first cut, the N uptake from the mineral fertilized soil significantly increased ($P<0.001$) compared with the control and the compost amended soil. The increase in mineral fertilizer application rate significantly increased ($P<0.001$) the crop N uptake. Compost application at rates higher than 100 kg total N ha^{-1} significantly increased ($P<0.001$) crop N uptake compared with the control treatment. The application of composts at 400 kg total N ha^{-1} resulted in similar N uptake to the mineral fertilizer applied at 180 kg N ha^{-1} ($P>0.05$). The effects of compost type and application method on ryegrass N uptake were found not to be significant ($P>0.05$).

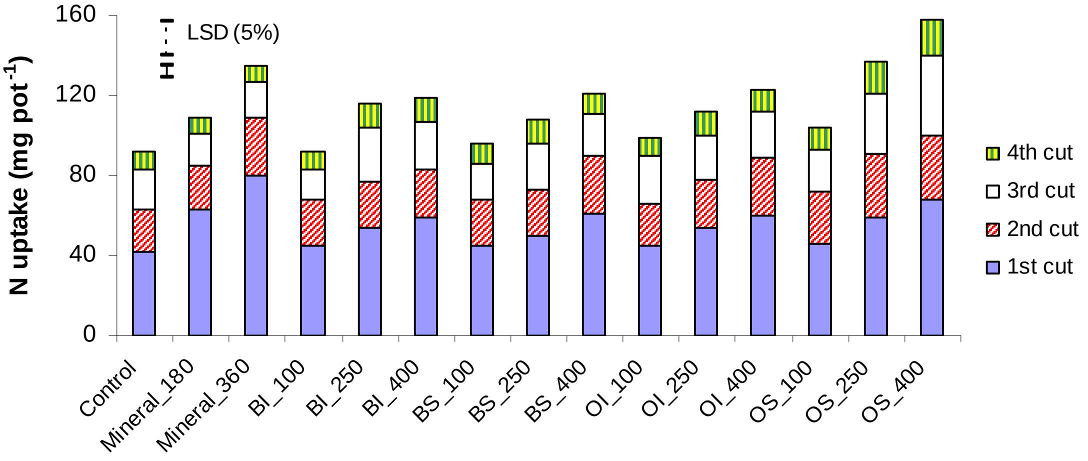

Figure 5.17 Ryegrass N uptake for the 15 treatments. The dashed-line LSD bar is for the comparison between the whole columns (sum of all cuts), whereas the solid-line LSD bar is between the different cuts (M: mineral fertilizer, BI: biowaste incorporated, BS: biowaste surface applied, OI: onion incorporated, OS: onion surface applied; 180, 360: application rates in kg N ha^{-1}; 100, 250, 400: application rates in kg total N ha^{-1}).

In the second harvest, mineral fertilization at a rate of 180 kg N ha^{-1} resulted in similar levels of crop N uptake with the control, whereas mineral fertilizer applied at a rate of 360 kg N ha^{-1} significantly increased the N uptake ($P<0.05$), in comparison to control and mineral fertilizer at 180 kg N ha^{-1}. Application of composts at rates higher than 100 kg total N ha^{-1} significantly increased ($P<0.05$) the crop N uptake, in comparison to the control soil, and it resulted in an N uptake not significantly different ($P>0.05$) from mineral fertilizer at 360 kg N ha^{-1}.

In the following two harvests, all treatments resulted in N uptake not significantly different from the control ($P>0.05$), with the exception of the onion compost surface applied at 250 and 400 kg total N ha^{-1}, which significantly increased the crop N uptake ($P<0.05$).

Figure 5.18 shows the total ryegrass N uptake as affected by compost and mineral fertilizer application at different rates.

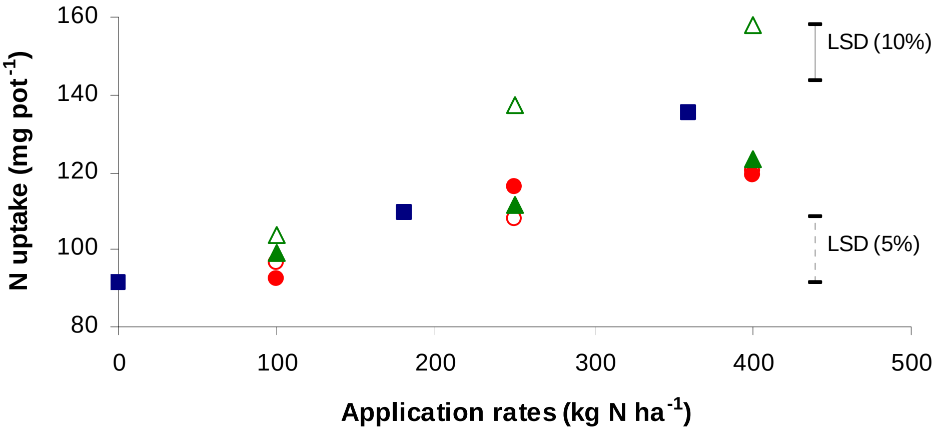

Figure 5.18 Ryegrass N uptake resulted from the compost and the mineral fertilizer amended soil (■ : Mineral fertilizer, ○: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated, △: Onion surface applied).

Mineral fertilization significantly increased ($P<0.001$) total ryegrass N uptake in comparison to the control. Increasing the application rate of mineral N resulted in a significant increase ($P<0.001$) in crop N uptake.

The increase in compost application rate also significantly increased ($P<0.001$) crop N uptake. Compost application at the rate of 100 kg total N ha^{-1} resulted in crop N uptake not being significantly different ($P>0.05$) from the control. Biowaste compost, surface applied or incorporated, and onion compost incorporated resulted in similar N uptake ($P>0.05$) to that of mineral fertilizer applied at 180 kg N ha^{-1}. Onion compost surface applied resulted in crop N uptake not significantly different ($P>0.05$) to mineral fertilizer applied at 360 kg N ha^{-1}. Onion compost surface applied at 400 kg total N ha^{-1} tended to result in higher ($P<0.10$) crop N uptake, in comparison to all other treatments.

Figure 5.19 shows the ryegrass N uptake response to compost total N application. Ryegrass N uptake increases linearly with increased amounts of compost total N applied. Figure 5.19 shows that the N use efficiency of onion compost surface applied was higher than from biowaste compost surface applied or incorporated, or onion compost incorporated.

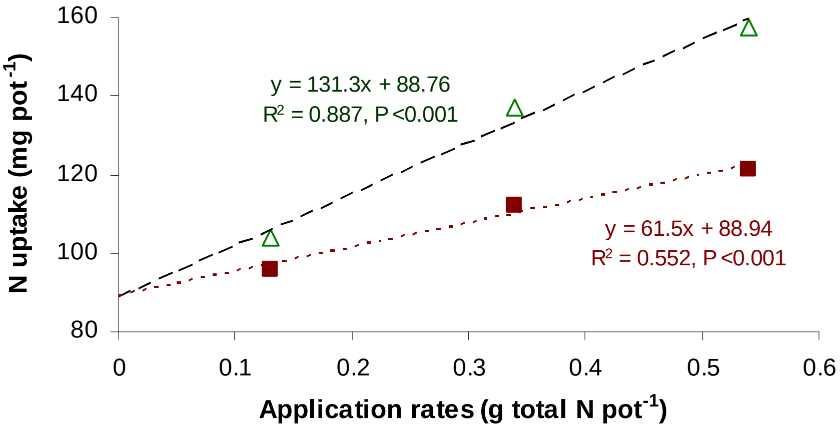

Figure 5.19 Ryegrass N uptake response to compost application (■ : Mean of biowaste surface applied, biowaste incorporated and onion incorporated, △: Onion surface applied).

The crop N recovery of onion compost N ranged between 4.9% and 12.0%, whereas biowaste compost N was between 0.6% and 6.5%. Table 5.3 shows the crop N recovery of compost N for the different compost application rates and methods used.

Table 5.3 Ryegrass N recovery of the total onion and biowaste compost N applied at the different rates and methods.

Rates	**N recovery (%)**			
	Biowaste		Onion	
(kg total N ha^{-1})	Incorporated	Surface	Incorporated	Surface
100	0.6	3.3	4.9	8.3
250	6.5	4.3	5.3	12.0
400	4.5	4.7	5.2	10.7

The N recovery of the mineral fertilizer N was 7.8% at the rate of 180 kg N ha^{-1}, and 9.5% at the rate of 360 kg N ha^{-1}. The low N recovery of the mineral fertilizer N is attributed to N losses by leaching (paragraph 5.3.2.2), denitrification or NH_3 volatilization. The later case being very possible as the fertilizer was applied at high rates on the soil surface, the soil was kept at or near field capacity, and slow drying conditions existed for several days (Wild, 1988; Tisdale et al., 1999).

The ryegrass N concentration was low in all treatments. The mean ryegrass N concentration ranged from 10 to 11.6 g N kg^{-1} DM, whilst the minimum level of N required by productive grazing animals is 20 g N kg^{-1} DM (ARC, 1980). This finding, suggests that both mineral fertilizer and compost application did not meet the crop N requirements for optimum production. The N application at rates lower than the optimum for grass production, results in increased grass growth, with little or no increase in N concentration. As the rate of N increases, both yield and N concentration increase, until the yield reaches a maximum level. Further increase in N rates is likely to increase N concentration but the change in yield is limited (Whitehead, 1995). However, the late establishment of the crop (in June) can also have resulted in the low response of N uptake by ryegrass. Salazar et al. (2005) have also reported limited response of N uptake by ryegrass established in June in the UK. Ryegrass N concentrations less than 20 g N kg^{-1} DM have also been reported by Beckwith et al. (2002) following ammonium nitrate application at 150 kg N ha^{-1} to sandy loam/loamy sand soil.

5.3.2.2 Nitrogen leaching

As mentioned in paragraph 5.2.1, in order to ensure efficient crop growth, the soil moisture was kept close to field capacity by almost daily irrigation. In practice, leachate was produced daily. Due to high temperatures and drought conditions during the summer of 2006, losses of leachate by evaporation in the collection buckets occurred and also losses of N by denitrification, from the soil or the leachate, should have taken place. Therefore, the amounts of N leached shown below should be regarded as indicative ones.

The main N leaching losses were observed during the first month of the experiment (Figure 5.20), when the ryegrass N requirements were low. The mineral N (sum of nitrite-, nitrate-, and ammonium- N) concentration in the leachate of all treatments was almost zero after the end of June. This finding indicates efficient utilization of available N by the crop from the end of June and onwards.

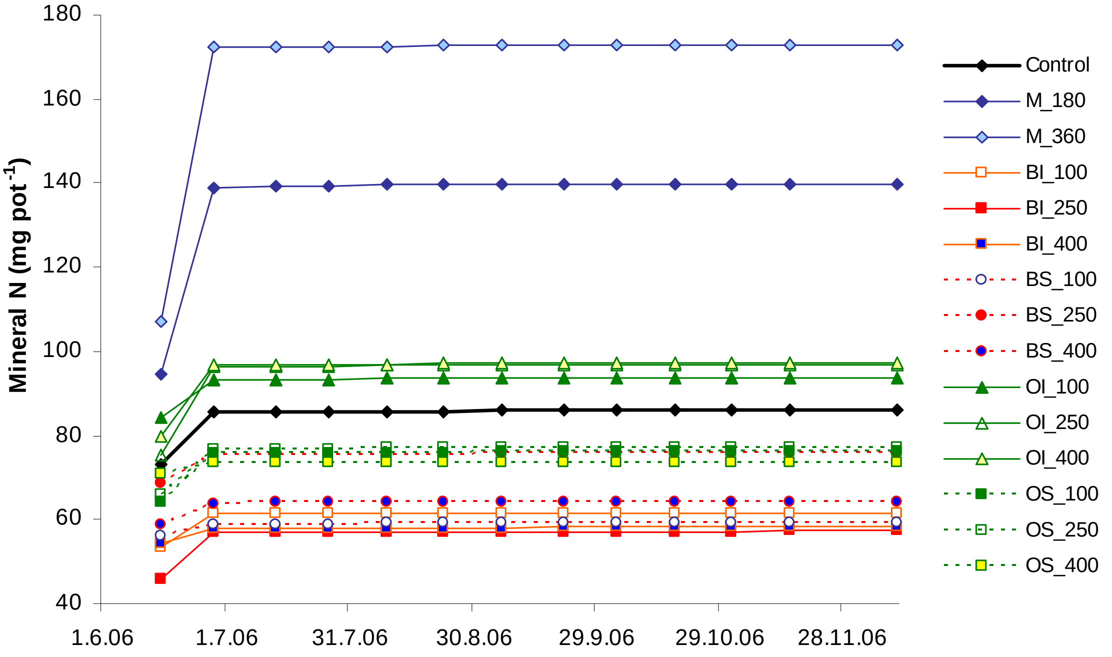

Figure 5.20 Cumulative mineral N leached from the different treatments (M: mineral fertilizer, BI: biowaste incorporated, BS: biowaste surface applied, OI: onion incorporated, OS: onion surface applied; 100, 250, 400: compost application rates in kg total N ha^{-1}; 180, 360: application rates in kg N ha^{-1}).

The application of composts did not increase the total amount of mineral N leached, in comparison to the control soil (Figure 5.21). Mineral fertilizer application significantly increased ($P<0.001$) the amount of mineral N leached, compared to control and compost amended soil. Biowaste compost application resulted in significantly lower ($P<0.001$) amount of mineral N in the leachate than the onion compost and the control soil. The effect of compost application rate on the total amount of mineral N leached was not significant ($P>0.05$). The amount of mineral N leached from the biowaste compost amended soil was not significantly affected ($P>0.05$) by the method of application. However, it was observed that onion compost incorporation significantly increased ($P<0.05$) the amount of mineral N leached in comparison to surface application.

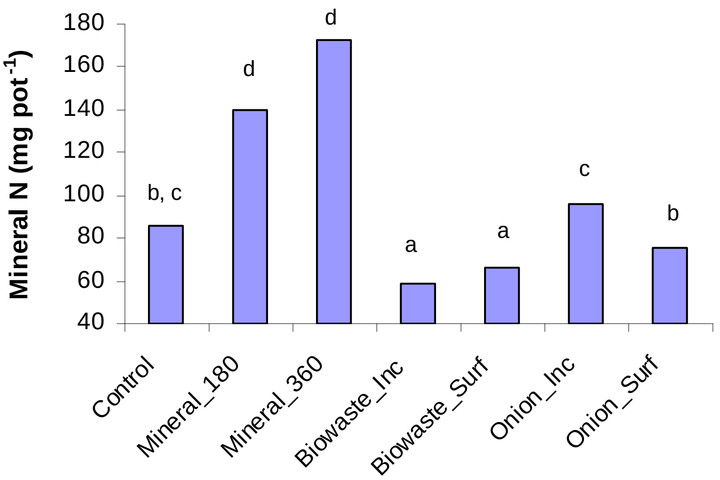

Figure 5.21 The effect of compost type and application method on the total amount of mineral N leached, in comparison to the reference treatments. Columns labelled with the same letter are not significantly different ($P>0.05$).

The mineral N leaching losses calculated as a percentage of the mineral fertilizer N applied (Equation 5.1) was 23.4 % at the rate of 180 kg N ha^{-1}, and 18.9% at 360 kg N ha^{-1}. The amount of the mineral N leached from the compost amended soils was lower than those of the control, with the exception of the onion compost incorporated. Therefore, the N leaching losses calculated as a percentage of the total N applied were negative for the biowaste and the surface applied onion compost treatments. Table 5.4 shows the mineral N leaching losses from the compost amended soil as a percentage of the total N applied.

Table 5.4 Mineral N leaching losses as a percentage of the total onion and biowaste compost N applied at the different rates and methods.

Rates	**N leaching losses (%)**			
	Biowaste		Onion	
(kg total N ha^{-1})	Incorporated	Surface	Incorporated	Surface
100	-18.7	-20.5	5.9	-7.5
250	-8.4	-3.1	3.2	-2.6
400	-5.1	-4.0	2.1	-2.3

After the 13th of July, the amount of total dissolved N (TDN) in the leachate was measured in order to determine whether the lack of mineral N leaching after this date was due to efficient crop N uptake or to N leaching in organic forms. Figure 5.22 shows the amount of TDN leached over the measuring period.

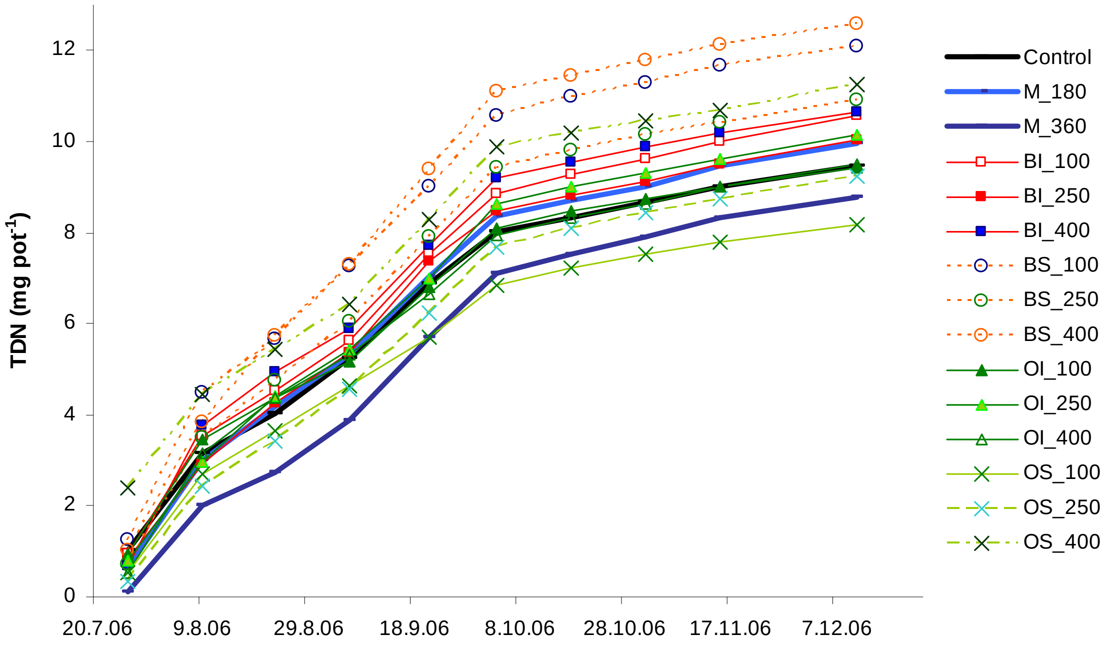

Figure 5.22 Cumulative TDN leached during the period of 14 July to 11 December (BI: biowaste incorporated, BS: biowaste surface applied, OI: onion incorporated, OS: onion surface applied; 100, 250, 400: compost application rates in kg total N ha^{-1}; 180, 360: application rates in kg N ha^{-1}).

The mean total amount of mineral N leached after the beginning of the TDN measurements was 0.3 mg N per pot, whereas the mean total amount of TDN was 10.2 mg per pot. Figure 5.23 shows the effect of compost type on the total amount of TDN leached during the period of 14 July to 11 December 2006, in comparison to the control and mineral fertilizer treatments. The application of biowaste compost significantly increased (P<0.05) TDN in the leachate in comparison to the onion compost and control.

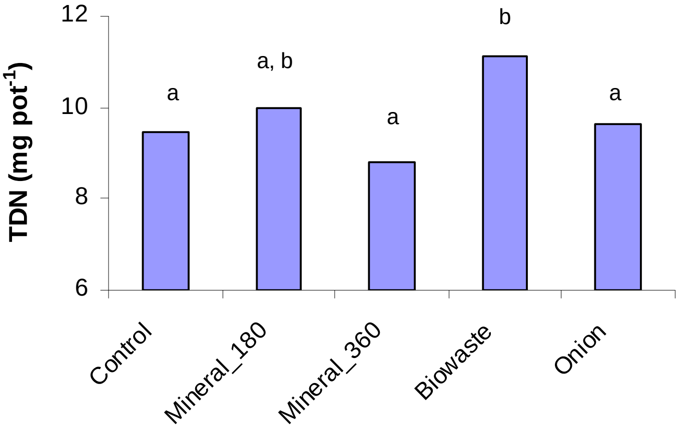

Figure 5.23 The effect of compost type on the total amount of TDN leached, in comparison to the reference treatments. Columns labelled with the same letter are not significantly different (P>0.05).

The effect of compost application rate on the total TDN leached was not found to be significant (P>0.05). The method of compost application was found to influence the TDN losses in the case of biowaste compost. The surface applied biowaste resulted in higher (P<0.10) amounts of TDN in the leachate than the biowaste incorporated. The mineral fertilized soil at 180 kg N ha^{-1} resulted in no significant difference (P>0.05) in the amount of TDN in the leachate from the biowaste compost amended soil.

The amount of dissolved organic N (DON) in the leachate was taken as the difference between the TDN and the mineral N measured for each treatment and sampling date (Equation 5.2). Figure 5.24 shows the amount of DON leached over the measuring period.

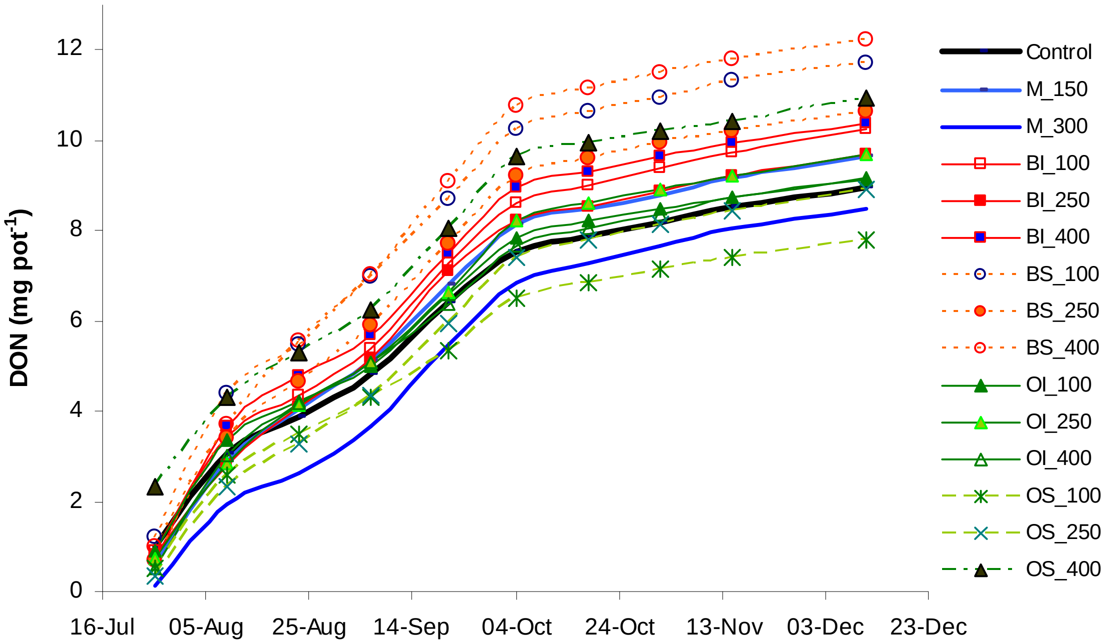

Figure 5.24 Cumulative DON leached during the period of 14 July to 11 December (BI: biowaste incorporated, BS: biowaste surface applied, OI: onion incorporated, OS: onion surface applied; 100, 250, 400: compost application rates in kg total N ha^{-1}; 180, 360: application rates in kg N ha^{-1}).

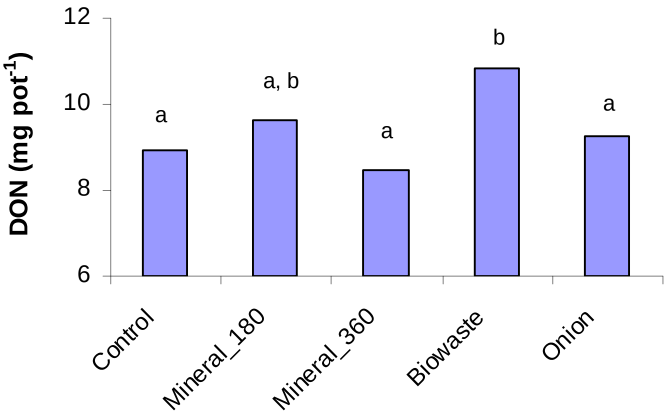

Figure 5.25 The effect of compost type on total amount of DON leached, in comparison to the control and mineral fertilizer treatments. Columns labelled with the same letter are not significantly different (P>0.05).

As shown in Figures 5.22 to 5.25, the TDN measured in the leachate for the period July 14 to December 11, was mainly DON. Taking into consideration that the DON was similar for the mineral fertilizer and compost treatments it should be concluded the lack of mineral N leaching after July 13, was due to efficient crop N uptake, not a result of excessive N leaching in organic forms. The amount of dissolved organic N in the leachate for the different treatments indicates that the N losses in organic forms were low and they should not be considered as an important factor influencing the mineral N leaching from compost amended soil, at least as shown during the period of July 14 to December 11 (after the first four weeks of compost application and during the crop growth).

5.3.2.3 Soil properties

Most soil properties were significantly influenced by the compost application. Compost application at rates higher than 100 kg total N ha^{-1} significantly increased ($P<0.001$) soil total N content, compared with the reference treatments. Soil total N and C content was not significantly affected ($P>0.05$) by the compost type or the application method used. Compost application, even at the lower rates, significantly increased ($P<0.001$) soil total C content compared with the reference treatments. Increasing the compost application rate resulted in a significant increase ($P<0.001$) in soil total N and C content (Figure 5.26). The total N and total C content of the control soil was 0.17% and 1.74%, respectively. The soil C: N was not significantly influenced ($P>0.05$) by the 2 years compost application (mean C: N ratio was 10.5).

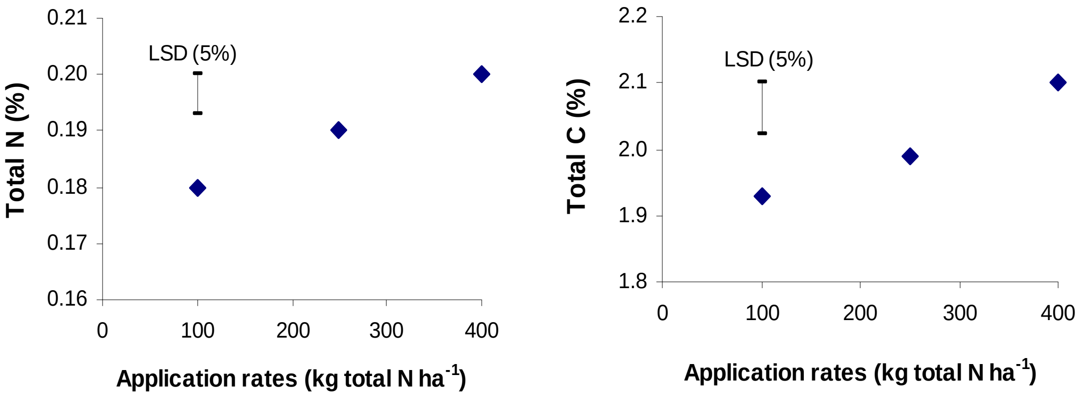

Figure 5.26 Soil total N (left) and total C (right) content following the second year of compost application (2006).

The soil organic matter content also increased significantly ($P<0.05$) following compost application (Figure 5.27). The effect of compost type on soil OM levels was found not to be significant ($P>0.05$). Increasing the compost application rate resulted in a significant increase ($P<0.001$) in soil OM content. Compost incorporation resulted in significantly higher ($P<0.05$) soil OM content compared with surface application.

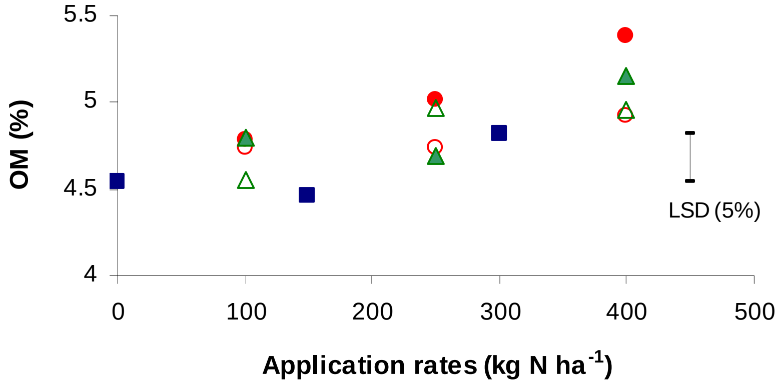

Figure 5.27 Soil OM content resulted from the compost and the mineral fertilizer amended soil (■ : Mineral fertilizer, ○: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated, △: Onion surface applied).

The soil residual TON-N was equal to zero for all treatments. However, there were small quantities of NH_4-N in the soil. This finding indicates that the nitrification process was minimized or ceased at that time of year (December), which was expected due to the low temperatures. Figure 5.28 shows the effect of compost type and application method on the soil NH_4-N content, in comparison to the reference treatments. The effect of compost application rate on soil NH_4-N was not significant ($P>0.05$). Figure 5.29 shows that the surface applied onion compost resulted in significantly higher ($P<0.05$) levels of mineral N in the soil than the control. This finding indicates higher N availability for plant uptake, during the following winter months, which suggests positive effects on the ryegrass yield, or for N leaching in the case there was no covering crop during the winter months. However, it needs to be noted that the soil NH_4-N from all treatments corresponds to less than 4 kg NH_4-N ha^{-1}.

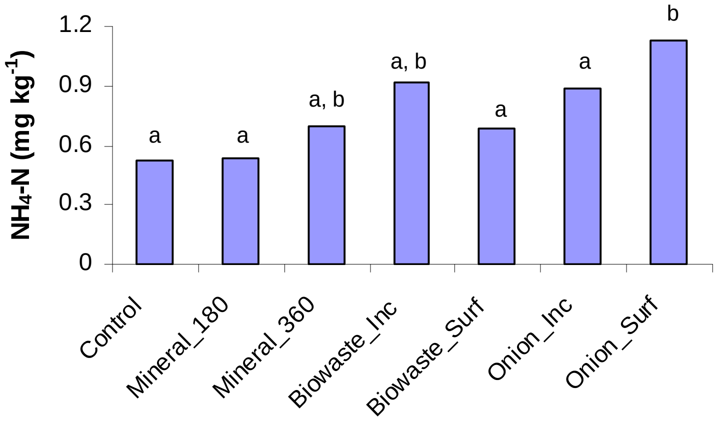

Figure 5.28 The effect of compost type and application method on soil residual NH_4-N levels, in comparison to the control and mineral fertilizer treatments (Inc: incorporated, Surf: surface applied). Columns labelled with the same letter are not significantly different (P>0.05).

Figure 5.29 presents the soil extractable K content as affected by the different treatments. Onion compost resulted in significantly higher (P<0.001) soil K levels in comparison to biowaste compost. Onion compost application at rates higher than 100 kg total N ha^{-1} and biowaste compost application at rates higher than 250 kg total N ha^{-1} significantly increased (P<0.001) the soil extractable K content, compared with the reference treatments. Increasing the composts application rate resulted in a significant increase (P<0.001) in soil extractable K levels. The surface application of both composts resulted in significantly higher (P<0.001) K levels than incorporation.

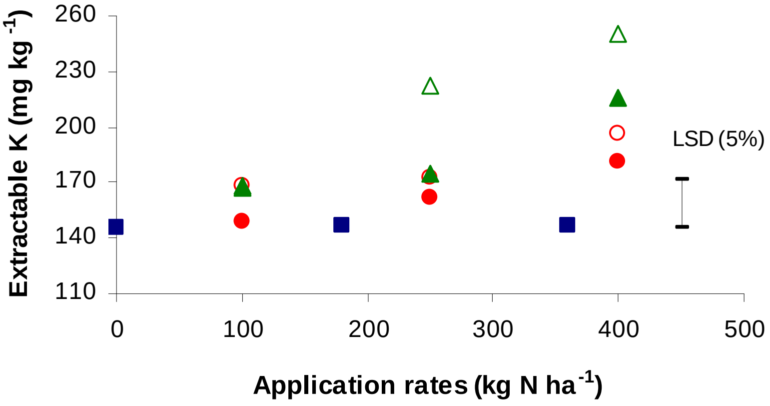

Figure 5.29 Soil extractable K content resulted from the compost and the mineral fertilizer amended soil (■ : Mineral fertilizer, ○: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated, △: Onion surface applied).

An important feature to note is that all reference treatments were amended with fertilizer K at the rate of 0.09 g pot^{-1} in the second year of the experiment. Consequently the results shown in Figure 5.29 suggest that compost application even at the lower rate resulting in adding about 0.09 g pot^{-1}, which is in good agreement with the amounts of soluble K in the compost (see Table 5.1). Figure 5.30 presents the soil extractable K response to the compost K applied. Onion compost application resulted in higher soil K content because of the higher soluble K content.

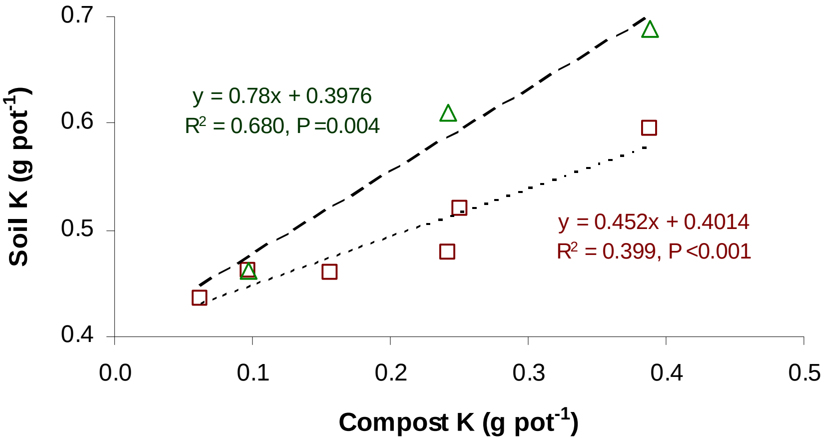

Figure 5.30 Soil extractable K response to the application of compost soluble K (□ : Onion incorporated, Biowaste incorporated and surface applied, △: Onion surface applied).

Compost application increased the soil available P levels compared to the reference treatments (Figure 5.31). Onion compost application resulted in significantly higher ($P<0.001$) P levels compared to the biowaste compost. The increase in compost application rate also significantly increased ($P<0.001$) the soil P content. The effect of the compost application method was observed not to be significant ($P>0.05$).

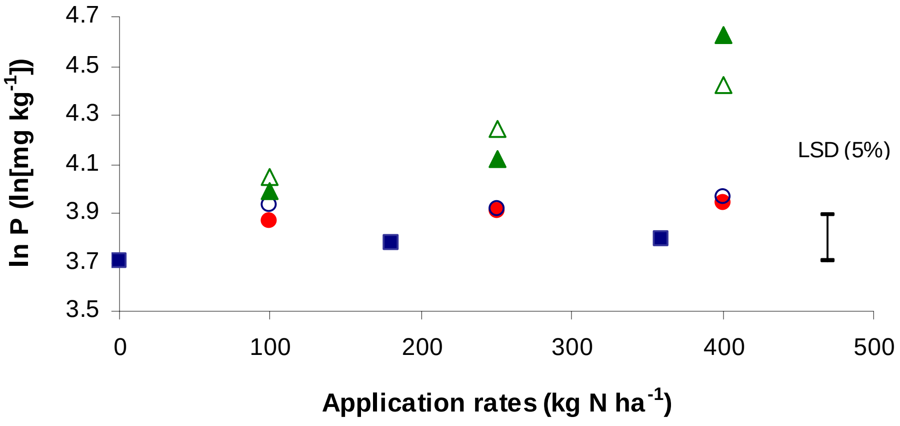

Figure 5.31 Soil available P for the different treatments and application rates. The values shown are the natural logarithm of the original values (■ : Mineral fertilizer, ○: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated, △: Onion surface applied).

The extractable Na content of compost treated soil was not significantly different ($P>0.05$) from mineral fertilized soil. Compost application resulted in similar levels of Na with control, with the exception of compost incorporation at 400 kg of total N ha^{-1}, which significantly increased the soil extractable Na content in comparison to the control (Figure 5.32). The effect of compost type was found not to be significant ($P>0.05$).

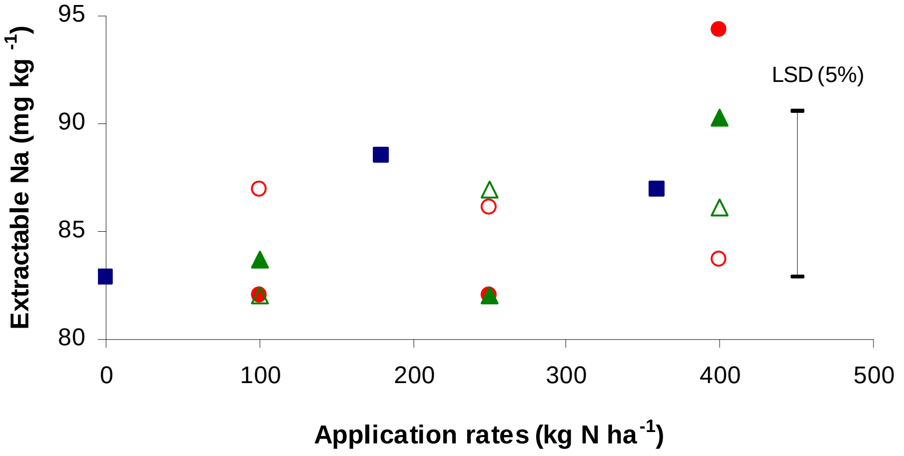

Figure 5.32 Soil extractable Na for the different treatments and application rates (■ : Mineral fertilizer, ○: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated, △: Onion surface applied).

The effect of compost and mineral fertilizer application on soil pH, was not significant (P>0.05) (mean pH was 6.5). The effect of compost application on soil moisture content (gravimetric) is shown in Figure 5.33. Compost application significantly (P<0.05) increased the soil moisture content in comparison to the control soil, when applied at rates higher than 100 kg total N ha^{-1}. However, compost application even at the higher rate resulted in similar levels (P>0.05) of soil moisture with the mineral fertilizer applied at the highest rate.

Onion compost surface applied at 400 kg total N ha^{-1} tended to result in higher soil moisture (P<0.10). The surface application of composts also significantly increased (P<0.05) the soil moisture content in comparison to compost incorporation. The effect of compost application on soil moisture was not as clear as in the first year due to the different crop being used and the time of sampling. Ryegrass is a cover crop and therefore the ryegrass production interferes with the soil moisture content. The better the crop is developed the less moisture losses by evaporation are expected. This is indicated by the increase of soil moisture with the increase of mineral fertilizer N.

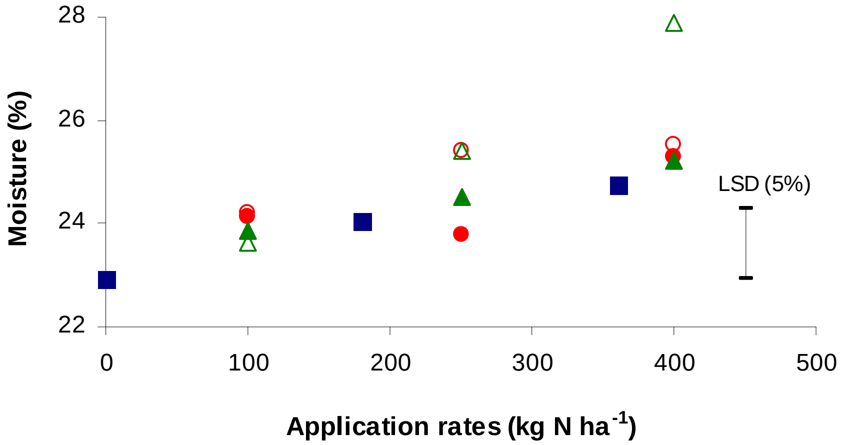

Figure 5.33 Soil moisture content (gravimetric) for the different treatments and application rates (■ : Mineral fertilizer, ○: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated, △: Onion surface applied).

5.3.2.4 General discussion of the second year results

Compost application at rates of 250 kg total N ha^{-1} or higher, resulted in increased ryegrass DM yield and N uptake, in comparison to the control soil. This increase was comparable to the mineral fertilized soil. Compost application at rates of 250 kg total N ha^{-1} or higher, resulted in similar levels of DM yield to mineral fertilizer applied at the rate of 180 kg N ha^{-1}. However, onion compost surface applied at 400 kg total N ha^{-1} tended to increase the ryegrass yield in levels similar to mineral fertilizer application at 360 kg N ha^{-1}.

The application of mineral fertilizer N increased ryegrass DM yield and N uptake in comparison to the control soil, but the low N concentration of the crop suggested that optimum production was not achieved. The ryegrass DM yield obtained from the mineral fertilized soil is in good agreement with other studies (Whitehead, 1995; Wilkins, 2000). However, the crop N concentration was low in all treatments (mean N concentration ranged from 10 to 11.6 g N kg^{-1} DM, whilst the minimum level of N required by productive grazing animals is 20 g N kg^{-1} DM (ARC, 1980)). These findings indicate that the N supplied by both the mineral fertilizer and the composts was adequate to increase DM yield, but not enough to increase the N concentration of the plant.

The ryegrass N recovery of the onion compost N ranged between 4.9% and 12.0%, whereas of the biowaste compost N between 0.6% and 6.5%, depending on the different compost application rates and methods used. Residual N availability from composts applied in the first year, however, needs also to be taken into consideration. The N recovery of the mineral fertilizer N was 7.8% at the rate of 180 kg N ha^{-1}, and 9.5% at the rate of 360 kg N ha^{-1}. The low N recovery of mineral fertilizer N applied is indicative of the high N losses occurred from the mineral fertilizer treatments. Also, it may explain the low N concentration of the crop, despite the high rates of mineral fertilizer N used.

Onion compost application increased crop N uptake during all four ryegrass harvests, which indicates the continuous release of compost N with time. The single application of mineral fertilizer N only increased the N uptake in the first two harvests, despite the high rates used. The fact that high rates of mineral fertilizer N did not result in optimum ryegrass production under irrigated sandy loam soil conditions, suggests that compost application before crop sowing followed by supplementary mineral N fertilization may be the best practise for maximising yield and N concentration.

Compost application increased ryegrass production without increasing N leaching. The majority of the N leaching losses from all treatments occurred during about the first month of the experiment (up until two weeks following crop emergence), when the crop N requirements were low. After the crop was well established in all treatments, there was limited N leaching.

The mineral fertilizer treatments resulted in increased N leaching. The N leaching from the mineral fertilized soil was enhanced by the fact that it was a single application of fertilizer N before the establishment of the crop, and the irrigation applied. The mineral N leaching losses calculated as a percentage of the mineral fertilizer N applied was approximately 21%. However, this percentage is expected to actually have been higher because of the losses of leachate (in volume) from the collection buckets by evaporation, and possibly losses of N leached from the open buckets by denitrification.

Compost application resulted in N leaching losses similar or lower than the control. As already shown in Table 5.4, with the exception of onion incorporated, the application of composts resulted in lower N leaching than the control. This finding can be explained as an effect of the compost type and the application method used.

The initial immobilization of soil N and the limited mineralization of biowaste compost N are likely to be the main reasons why the biowaste compost (C: N of 20) application resulted in lower N leaching than the control soil. The application of composts with C: N ratio of 20, are likely to result in N immobilization (Sullivan et al., 2002). Ryegrass yield from the biowaste compost amended soil in the first harvest was, however, higher than the control. This finding suggests that the application of the biowaste compost resulted in contemporary soil N immobilization at the initial stages only after application, thus decreasing the N leaching in comparison to control.

In the case of the onion compost, the main feature to consider is that the method used to apply the compost significantly affected the N leached. The surface application of onion compost resulted in less N leaching than incorporation. Compost surface application is likely to result in faster mineralization of N (Smith et al., 1997) and consequently in higher N availability than incorporation. The surface application of the onion compost, however, also resulted in earlier establishment of ryegrass. This is attributed to the higher soil moisture content following onion compost surface applied (soil moisture is likely to have been a limiting factor for crop establishment because of the warm and dry conditions following crop sowing). The N mineralized was more efficiently taken up by the better established plants at the earlier days of the experiment and therefore the N leaching losses were lower.

The low amount of dissolved organic N in the leachate for the different treatments indicates that the N losses in organic forms were low and they should not be considered as an important factor influencing the mineral N leaching from compost amended soil, at least as shown during the period following the first four weeks of compost application and during the crop growth.

Soil total N, total C and OM content were increased following the compost application (mean increase in comparison to control soil was 11.8%, 15.5% and 7.8%, respectively). Increasing the compost application rate increases the soil OM, total N and C content. The method of compost application only significantly affected the soil OM, with compost incorporation to significantly increase soil OM compared to surface application.

Soil P and K content increased with compost application. Onion compost having a higher soluble P and K content, resulted in higher levels of P and K in comparison to biowaste compost. In general, compost soluble K content was shown to give a good estimate of the K availability following compost application.

The surface application of both composts resulted in higher residual K levels than incorporation. This finding is in good agreement with the first year results. Compost surface application in the first resulted in lower maize K uptake than incorporation and consequently in higher residual soil K levels. The increased levels of soil K in the second year of compost application may be attributed to the already higher soil K levels from the first year or/and to higher ryegrass K uptake from the incorporated composts (ryegrass K uptake was not determined).

Soil Na content was only increased at the highest rate of compost application. However, sodicity problems in light textured soils are not likely to occur, especially under irrigation. Potential problems may rise, however, with sodicity sensitive crops. Compost application at rates of 250 kg total N ha^{-1} or higher, either surface applied or incorporated, increased the soil gravimetric moisture content, indicating potential benefits to the water holding capacity of the soil.

5.4 Conclusions

The conclusions drawn from the two periods of the pot experiment are summarized below:

g) Biowaste and onion compost application to forage maize production on sandy loam soil under irrigation had no significant effect on crop yield and N uptake, mainly due to the mineralization of background levels of soil inherent N and its utilization by the crop.

h) The second year of biowaste and onion compost application to the same soil (under irrigation) increased annual ryegrass production, when applied at rates of 250 kg total N ha^{-1} or higher. The increase was at levels comparable with a single mineral fertilization at 180 kg N ha^{-1}. However, optimum crop production was not achieved neither with the mineral fertilization nor the compost application.

i) Compost application is not likely to increase N leaching, not even when applied at high rates, as the compost N mineralization is slow and the amount of N mineralized is low. Care, however, should be taken in soils with high mineralization potential of inherent N, so as the compost N mineralized is not in excess of the crop N requirements.

j) Compost C: N ratio and C quality is influencing N mineralization.
    a. Compost with C: N ratio of 20 results in limited mineralization or immobilization of compost N.
    b. There is evidence to suggest that compost N mineralization depends also on the forms of compost C (such as lignin) and consequently composts with similar C: N ratios but different carbon quality can differ on their mineralization potential.

k) Compost application to sandy loam soils is likely to improve soil quality, by increasing the soil OM, total N, total C, and extractable K content, and water retention, and consequently increase crop production.

l) The surface application of composts may result in faster mineralization of compost N and organic matter, and increased soil water retention than compost incorporation.

6 Lysimeter trial: Nitrogen leaching and plant uptake from compost amended sandy soil

6.1 Introduction

This Chapter focuses on the lysimeter experiment, which took place during the period of May 2006 to November 2006. The lysimeter trial was required to study the potential nitrogen leaching of the biowaste and vegetable waste composts under high drainage rate conditions, and the agronomic benefits of compost application to poor quality soil.

The pot experiment, described in Chapter 5, contributed towards the assessment of the nitrogen losses from a compost amended fertile soil, rich in nitrogen and organic matter. The lysimeter experiment involved the use of sand as the growing medium aiming to ensure poor soil quality, low N levels, and also high drainage rate conditions, which would enhance nitrate leaching and subsequently would provide an indication of environmental risks from excessive N leaching.

The specific objectives of the lysimeter trial are summarized below:

1. To evaluate the fertilizing and soil conditioning potential of biowaste and vegetable waste composts under poor sandy soil conditions, and
2. To determine the environmental risks of excessive nitrogen leaching from the application of these two compost types to a well drained soil.

6.2 Materials and methods

6.2.1 Lysimeter experiment

The lysimeter study was undertaken at the experimental site of Cranfield University at Silsoe, UK. The lysimeters were filled with sand (commercially available ‘Garside Sands’) over a thin layer of gravel to facilitate drainage, as illustrated in Figure 6.1.

Leachate collection was from a drain pipe at the base of the lysimeters. Sand properties are presented in Table 6.1.

Figure 6.1 Lysimeters layout.

Table 6.1 Sand properties at the beginning of the experiment.

	Sand	Silt	Clay	pH	OM	Total N	Mineral N	P	K	Mg	Ca
	(%)				(%)		(mg kg^{-1})				
Fine sand	99.4	0.6	0.0	7.4	1.36	< 0.03	0	0.5	27.9	16.7	362.6
Coarse sand	98.5	0.8	0.7	7.6	1.33	< 0.03	0	0.5	20.7	13.2	257.4

The sand was amended with the two compost types: the biowaste and the vegetable waste composts. The two composts were incorporated within the top 15 cm of the sand at four rates corresponding to the application of 100, 250, 400 and 600 kg of compost total N ha^{-1}. The compost application rates along with the corresponding amount of nutrients applied are summarized in Table 6.2.

Forage maize was used as the monitoring crop. Mineral fertilizer (ammonium nitrate) treatments (0, 40, 80 and 120 kg N ha^{-1}) were used as reference (the treatment resulting in the application of 0 kg N ha^{-1} is referred to as ‘control’). The application of 120 kg N ha^{-1} corresponds to the highest amount of fertilizer N recommended by MAFF (2000) for the production of forage maize on poor soils.

The sand in all lysimeters was amended with mineral P, K and Mg in the beginning of the experiment, in order to supply the necessary macro-nutrients (apart from N) for maize growth in such a poor quality soil. The application quantities of P, K, and Mg (110, 230 and 60 kg ha^{-1}, respectively) were the maximum amounts recommended for forage maize production in poor light soils by MAFF (2000). Each treatment was applied on an individual lysimeter using a completely randomized design with three replicates.

Table 6.2 Biowaste and onion composts: application rates and respective amounts of nutrients and organic matter (OM) applied per lysimeter, and composts C: N ratio.

Application rates (kg total N ha^{-1})	**Fresh material** (g)	**Total N** (g)	**Soluble mineral N** (g)	**Soluble K** (g)	**Soluble P** (g)	**OM** (g)	**C: N ratio**
Biowaste compost							20.1
100	188	2.4	0.0	1.1	0.0	96	
250	471	5.9	0.1	2.8	0.1	239	
400	754	9.5	0.1	4.5	0.2	383	
600	1131	14.2	0.2	6.8	0.3	575	
Onion compost							10.5
100	495	2.4	0.0	1.7	0.1	50	
250	1238	5.9	0.1	4.2	0.3	124	
400	1980	9.5	0.2	6.7	0.5	199	
600	2970	14.3	0.3	10.1	0.8	299	

The experiment was carried out outdoors. The lysimeters were both rain-fed and irrigated. Drip irrigation was applied to all lysimeters, aiming to cover the water requirements of the crop. Two drippers were employed per lysimeter in order to ensure better water distribution within the soil profile.

The study was initiated on May 18, 2006, when the compost incorporation took place. The composts were incorporated within the top 15 cm of sand. Forage maize was sown on June 1 (at a rate of 6 seeds per lysimeter), and the application of the mineral fertilizer N was on the following day. On June 26, the maize was thinned to 3 plants per lysimeter. Supplementary fertilization with mineral P and K was applied on July 16, at the same rates as at the beginning of the experiment. Maize harvest was on September 11.

Table A.3.3.1 (Appendix A.3.3) summarizes the experimental establishment and monitoring by showing the dates of the experiment establishment practices, along with the soil, crop and leachate sampling dates. Figure A.3.3.1 illustrates the layout of the experiment and Figure A.3.3.4 presents the lysimeter trial on site. The leachate collection system employed is shown in Figure A.3.3.5. Information regarding the climatic conditions during the experimental period is presented in Tables A.3.3.2 and A.3.3.3. Figures A.3.3.2 and A.3.3.3 illustrate the total amount of water (rainfall and irrigation) applied per lysimeter, and the amounts of leachate produced.

In parallel to the lysimeter experiment, another three lysimeters were established. These three lysimeters were filled with sand following the same procedure described above, but without any addition of nutrients. Forage maize was sown, thinned and harvested at the same dates with the lysimeter experiment. The same amount of irrigation was also applied. Sand samples from the top 25 cm depths of these lysimeters were used at the end of the study as reference when assessing the effect of the initial and supplementary fertilization on certain sand properties. The average value of these three lysimeters is referred to as ‘check’. Data obtained from these three lysimeters were not included in the statistical analysis.

6.2.2 Leachate, forage maize and sand soil measurements and analysis

The first leachate sampling was on May 29. Leachate samples were taken on a 2-weeks basis until the crop harvest and after that on a monthly basis for another two months. Each leachate sample represents a bulk sample of the leachate produced between two consecutive sampling dates. Measurements of mineral N (total oxides of nitrogen-N (TON-N) and ammonium-N (NH_4-N)) in the leachate were undertaken on all samples by segmented flow analysis (SFA). The amount of mineral N leached was estimated as a percentage of the total amount of N applied according to Equation 6.1.

$$N\ leached\ (\%) = 100 \times \frac{Fertilizer\ or\ Compost\ N\ leached\ - Control\ N\ leached}{Total\ Fertilizer\ or\ Compost\ N\ applied} \tag{6.1}$$

Measurements of phosphate-P (PO_4-P) and total dissolved N (TDN) were carried out by SFA on the samples collected after June 29. The amount of dissolved organic N (DON) in the leachate was estimated as the difference between the TDN and the mineral N measured for each treatment and sampling date (Equation 6.2). Leachate samples were collected also in summer 2007 in a monthly basis (covering the period from May 1 to August 19, 2007) to study the residual effects of compost application on N leaching. These samples were analyzed for mineral N and TDN.

$$TDN = DON + Mineral\ N \tag{6.2}$$

Crop growth was monitored by measuring the plant height, which was defined as the distance from the back of the collar of the upper complete leaf to the soil surface. At harvest, crop above ground dry matter (DM) yield was determined by drying maize cobs at 60 ˚C until constant weight, and the rest of the plant at 105 ˚C for 26 hours. Maize total N concentrations were determined separately for the cobs and the rest of the maize plant by dry combustion using the Vario EL elemental analyzer. The DM yield of maize cobs and the rest of the maize plant were multiplied by the respective total N content, and then summarized to crop N uptake.

Maize total P and K concentrations were determined for the whole maize plant according to MAFF (1986). Crop P and K uptake were also evaluated by multiplying the whole maize DM yield by the measured P and K concentrations, respectively. Crop DM yield and N uptake were also measured for the plants which were removed at thinning to achieve more accurate calculation of the N budget. N recovery from compost and mineral fertilizer was calculated according to the Equation 6.3 (Hartl & Erhart, 2005).

$$N\ re\mathrm{cov}ery\ (\%) = 100 \times \frac{Fertilizer\ or\ Compost\ N\ uptake\ - Control\ N\ uptake}{Total\ Fertilizer\ or\ Compost\ N\ applied} \tag{6.3}$$

Sand fertility properties were assessed at thinning and harvest. At harvest, organic matter content (OM) was determined by the loss on ignition method, pH in the 1/2.5

soil/water suspension, total N, total C and C: N ratio by the use of elemental analyzer. Available P was estimated by the Olsen method and total P by the aqua regia method, and both were measured by spectrophotometry. Extractable K and Na were extracted by ammonium nitrate and measured by flame photometry (MAFF, 1986). Soil salinity and sodicity were estimated by measuring the electrical conductivity in the saturated paste extract (EC_e), and the exchangeable sodium percentage (ESP) from the sodium adsorption ratio (SAR) according to Richards (1954). The above mentioned parameters were determined for the layer of 0-0.25 m depth. Mineral N (TON-N and NH_4-N) was measured in potassium chloride extracts by segmented flow analysis. Samples were taken for the following depths: 0-0.25 m, 0.25-0.45 m and 0.45-0.65 m.

Also, the soil water release curve (WRC) was produced by using the sand table method (Harst et al., 1975) and the pressure membrane apparatus (Richards, 1947; Salter and Haworth, 1961). Plant available water was determined as the water retained between the states of field capacity (FC) and permanent wilting point (PWP). Soil FC was taken at -10 kPa and PWP at -1500 kPa.

At thinning, soil available P, extractable K and mineral N were determined for the top 0.25 m depth, following the same methodology described above for the samples taken at harvest.

Details of the methods involved in the soil, crop and leachate analysis can be found in Appendix A.2.

6.2.3 Statistical analysis

The effects of each treatment and the influence of compost type and application rate on the measured variables were assessed by analysis of variance (5% probability level) and the relation between variables was established by regression analysis (5% probability level) using the statistical package GenStat (GenStat, 2006).

The accuracy of the statistical analysis was checked graphically by the residual plots. When the residuals did not follow the normal distribution then the data were normalized by applying a square root or a natural logarithm transformation. Also, in cases of an extreme value in the data set, this value was removed from the analysis when this data point caused the residuals to be not normal.

6.3 Results and discussion

6.3.1 Forage maize production and nutrient uptake

6.3.1.1 Crop growth

Forage maize growth was significantly influenced (P<0.001) by compost type and application rate, as indicated by the changes in plant height shown in Figure 6.2.

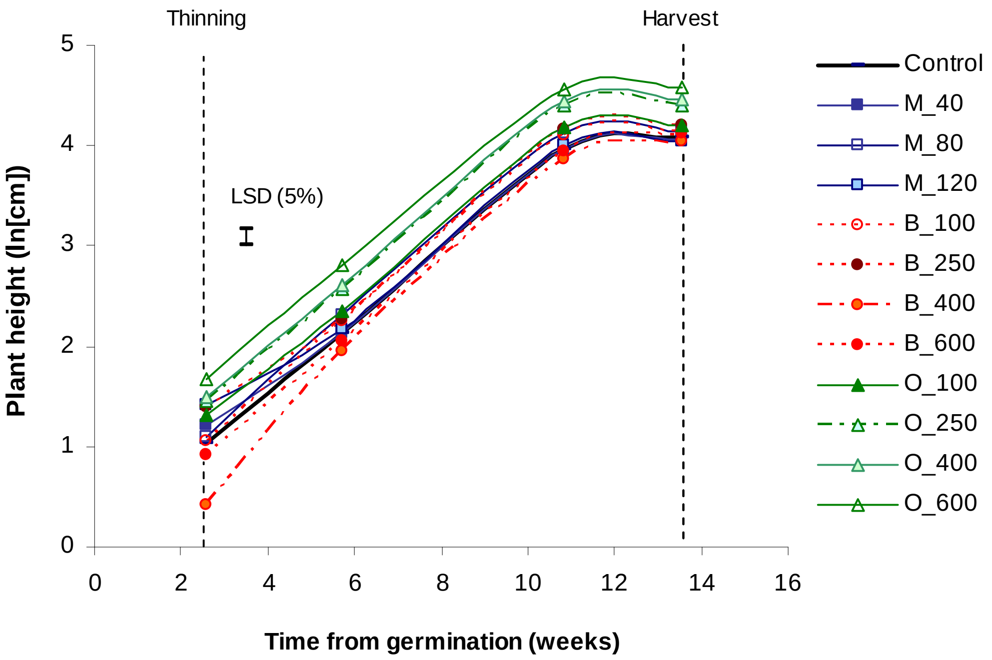

Figure 6.2 Forage maize height at the different growth stages as affected by mineral fertilizer and compost application at different rates. The values shown are the natural logarithms of the measured values (M: mineral fertilizer, B: biowaste compost, O: onion compost; 40, 80, 120: application rates in kg N ha^{-1}; 100, 250, 400, 600: application rates in kg total N ha^{-1}).

The application of biowaste compost at rates higher than 400 kg total N ha^{-1} resulted in restriction of maize growth during the first 6 weeks of the experiment. Maize height for these treatments was at levels lower or similar to control. This finding can be attributed to the higher amount of total C that was applied by the biowaste compost application at these high rates, which might have resulted in early season immobilization of soil N. After the first 6 weeks, however, no significant differences ($P>0.05$) were observed in maize plant height between the lower and higher rates of biowaste compost application, suggesting that the early season N deficiency did not persist through out the season and did not affect the final crop production. Eriksen et al. (1999) also reported an early season immobilization of soil N following the application of about 310 kg total N ha^{-1} of MSW compost with a C: N ratio of 40.

In general, after the first 6 weeks of crop growth, control, mineral fertilizer and biowaste treatments resulted in similar levels of crop height ($P>0.05$), whereas onion compost resulted in significantly higher ($P<0.05$) plant height, when applied at rates higher than 100 kg total N ha^{-1}. The increase of onion compost application rate resulted in increased plant height.

6.3.1.2 Dry matter yield and nitrogen uptake

Figures 6.3 and 6.4 demonstrate the effect of the different rates of biowaste and onion compost application on crop DM yield and N uptake, respectively, at the early stages of plant growth (3rd leaf stage). The measurements were carried out on the thinned plants.

At this early stage of plant growth, both mineral fertilizer and onion compost showed a trend to increase DM yield with increasing application rates. The onion compost and the mineral fertilizer applied at the rate of 120 kg N ha^{-1} significantly increased ($P<0.05$) the crop DM yield compared to the control. On the contrary, biowaste compost application at rates higher than 400 kg total N ha^{-1} resulted in less DM yield than the control, following the trends shown for the plant height (Figure 6.2).

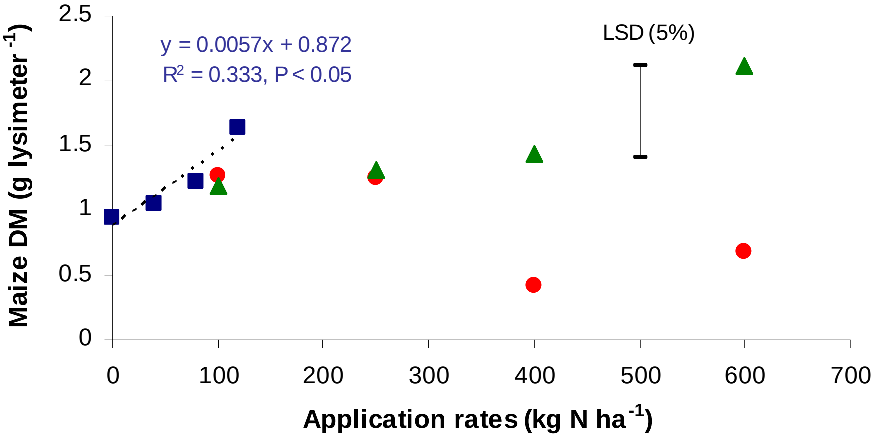

Figure 6.3 Forage maize DM yield at the 3rd leaf stage resulted from biowaste, onion compost, and mineral fertilizer amended soil (■ : Mineral, ●: Biowaste, ▲: Onion).

Forage maize N uptake followed the same pattern as the DM yield results; although a better response to mineral fertilization was noticed. N uptake increased significantly ($P<0.05$) with the mineral fertilizer and the onion compost when both applied at the highest rate. The biowaste compost application was not observed to have any benefit on crop DM yield and N uptake. N uptake was lower for the higher biowaste compost application rates supporting the early season N deficiency scenario discussed in paragraph 6.3.1.1.

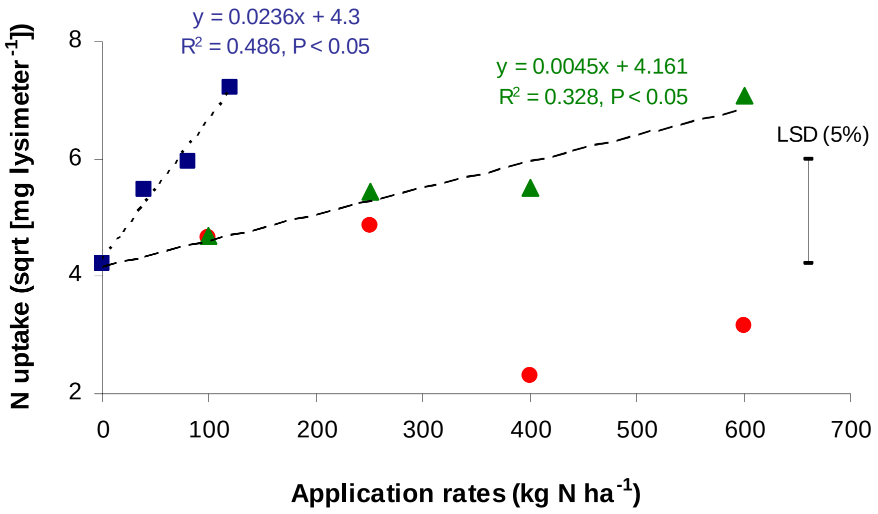

Figure 6.4 Forage maize N uptake at the 3rd leaf stage resulted from biowaste and onion compost, and mineral fertilizer amended soil. The values shown are the square roots of the measured values (■ : Mineral, ●: Biowaste, ▲: Onion).

The crop DM yield and N uptake at harvest are presented in Figures 6.5 and 6.6, respectively. Figure 6.5 shows that the mineral N fertilization, the biowaste compost application and the onion compost applied at the rate of 100 kg total N ha^{-1} resulted in similar (P>0.05) DM yield to the control treatment.

The initial reduction of maize yield due to the high rates of biowaste compost application was compensated at harvest, as was also shown for plant height. Onion compost application at rates higher 250 kg total N ha^{-1} significantly increased (P<0.001) DM yield. The increase in onion compost application rate resulted in a linear increase in DM yield, which indicates the positive crop response to onion compost application, but also that the crop did not reach its maximum yield.

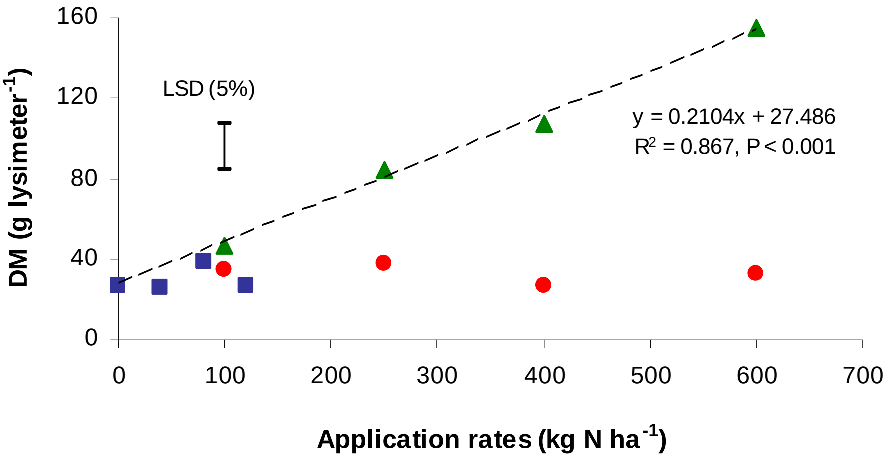

Figure 6.5 Forage maize DM yield at harvest resulted from biowaste, onion compost, and mineral fertilizer amended soil (■ : Mineral, ●: Biowaste, ▲: Onion).

The crop N uptake followed the same trends as the DM yield results. The onion compost (C: N ratio = 10.5) applied at rates equal or higher than 250 kg total N ha^{-1} significantly increased (P<0.001) crop N uptake. The increase of the rate of the onion compost application increased the N uptake linearly, suggesting that there was an increase in plant-available N mineralized with increased rates of onion compost.

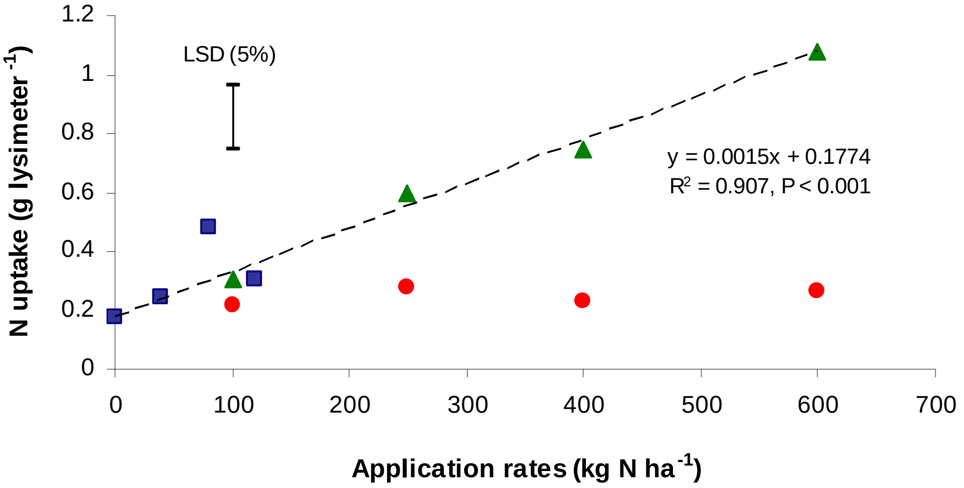

Figure 6.6 Forage maize N uptake at harvest resulted from biowaste, onion compost, and mineral fertilizer amended sand (■ : Mineral, ●: Biowaste, ▲: Onion).

The increase of crop yield with the increase of onion compost application rate is in accordance with the findings of Iglesias-Jimenez & Alvarez (1993), who used compost with C: N ratio lower than 12. The biowaste application resulted in similar N uptake to the control (P>0.05). The poor crop response to the biowaste application indicates immobilization of biowaste compost N. This result supports the findings of Sullivan et al. (2002) who found that composts with C: N ratio higher than 20 result in immobilization of compost N. The mineral N uptake trend is similar to that of the biowaste compost, despite an anomaly at the 80 kg N ha^{-1}.

The N recovery from the onion compost N was found equal to 6.4% (mean of all application rates), whereas from the biowaste compost N was equal to 1.1% (mean of all application rates). Table 6.3 shows the crop N recovery for the different compost application rates. The N recovery of the mineral fertilizer N was low for all rates of application. It was found equal to 8.4% at the rate of 40 kg N ha^{-1}, 17% at the rate of 80 kg N ha^{-1}, and 5.7% at the rate of 120 kg N ha^{-1}. The low values of crop N recovery found for the mineral fertilizer treatments should be mainly attributed to the high N leaching losses occurred, as shown in paragraph 6.3.2.

Table 6.3 Nitrogen recovery resulting from onion and biowaste composts at the different rates of application.

Application rates (kg total N ha^{-1})	**N recovery (%)** Biowaste	 Onion
100	1.8	5.6
250	1.8	7.3
400	0.4	6.2
600	0.5	6.6

6.3.1.3 Phosphorus and potassium uptake

Crop P and K uptake were significantly influenced by compost application. Figure 6.7 presents the forage maize P uptake resulted from the different treatments. The four reference treatments are summarized in one ‘mineral’ treatment, since they all received the same amount of P and K, and the differences between them were not significant ($P>0.05$).

Compost application significantly increased ($P<0.001$)) crop P uptake. Increasing the compost application rate also led to significantly increased ($P<0.001$) P uptake. Onion compost resulted in significantly higher ($P<0.001$) P uptake and total P concentration in the plant tissue (Figure 6.8), compared using biowaste compost.

According to maize appearance (see Figure A.3.3.6 in the Appendix), although a basic amount of fertilizer P was applied, the crop was deficient in P. P deficiency symptoms were observed as early as the 3rd leaf stage, mainly for the reference and the biowaste compost treatments, and the onion compost at the lower rates. This limited response to fertilizer P application may be attributed to P fixation or P leaching. P availability is discussed in paragraph 6.3.3.3.

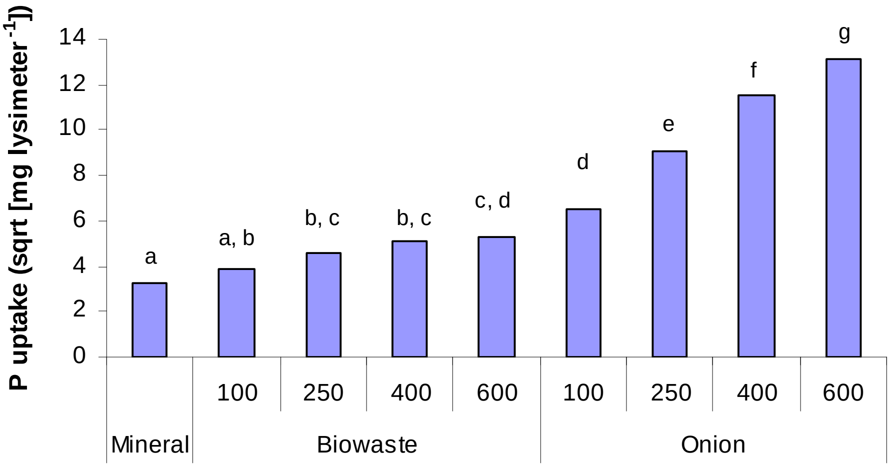

Figure 6.7 P uptake resulted from the biowaste and onion compost treatments, in comparison to the mean of all mineral fertilizer treatments (100, 250, 400, 600: application rates in kg total N ha^{-1}). The values shown are the square roots of the measured values. Columns labelled with the same letter are not significantly different (P>0.05).

Compost application also significantly increased (P<0.001) K uptake (Figure 6.9). Onion compost application resulted in significantly higher (P<0.001) K uptake compared to the biowaste compost. However, the biowaste compost application resulted in higher (P<0.05) plant K content (Figure 6.8). K deficiency symptoms were not observed for any treatment.

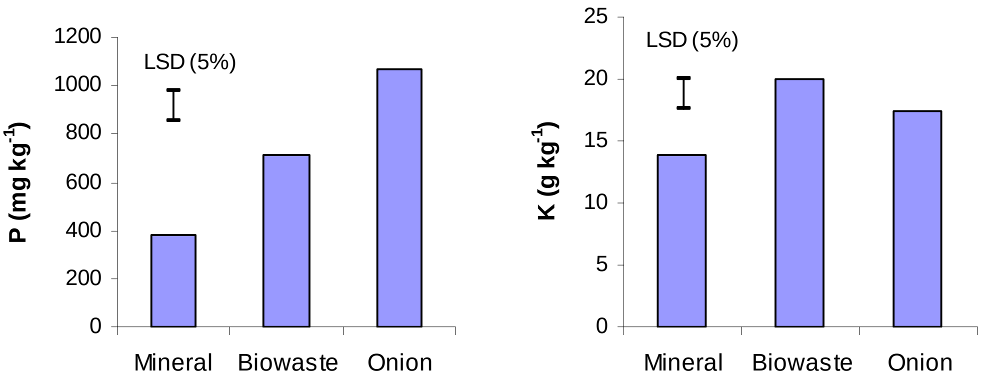

Figure 6.8 Forage maize total P (left) and total K (right) concentrations resulted from biowaste and onion compost amended sand, in comparison to the mean of all reference treatments.

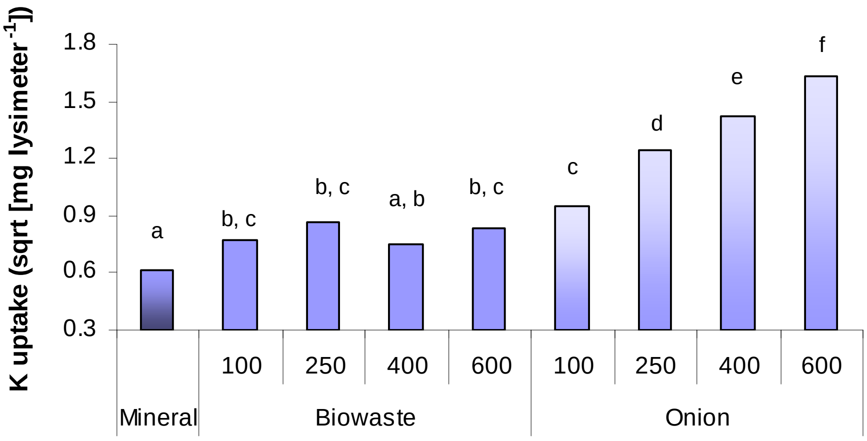

Figure 6.9 K uptake resulted from the biowaste and onion compost treatments, in comparison to the mean of all mineral fertilizer treatments (100, 250, 400, 600: application rates in kg total N ha^{-1}). The values shown are the square roots of the measured values. Columns labelled with the same letter are not significantly different (P>0.05).

Under the conditions of this study the application of mineral fertilization to poor quality sandy soils was not able to enhance forage maize production. The application of biowaste compost (C: N ratio > 20) resulted in low N availability and did not increase crop production. The application of onion compost (C: N ratio < 11) at rates > 250 kg of total N ha^{-1} was found to be the best practise to increase forage maize yield.

6.3.2 Nitrogen leaching

6.3.2.1 Mineral nitrogen

The mineral N leaching losses resulting from the different treatments are presented in Figure 6.10. During the first month of the experiment, both compost and mineral fertilizer treatments resulted in similar N leaching losses with the control. After this period, the mineral fertilizer treatments produced excessive levels of mineral N in the leachate. The increase of the mineral fertilizer application rate increased the mineral N in the leachate. The N leaching losses from the mineral fertilizer treatments occurred during the crop growing season, and therefore limited amount of mineral fertilizer N was available to the crop.

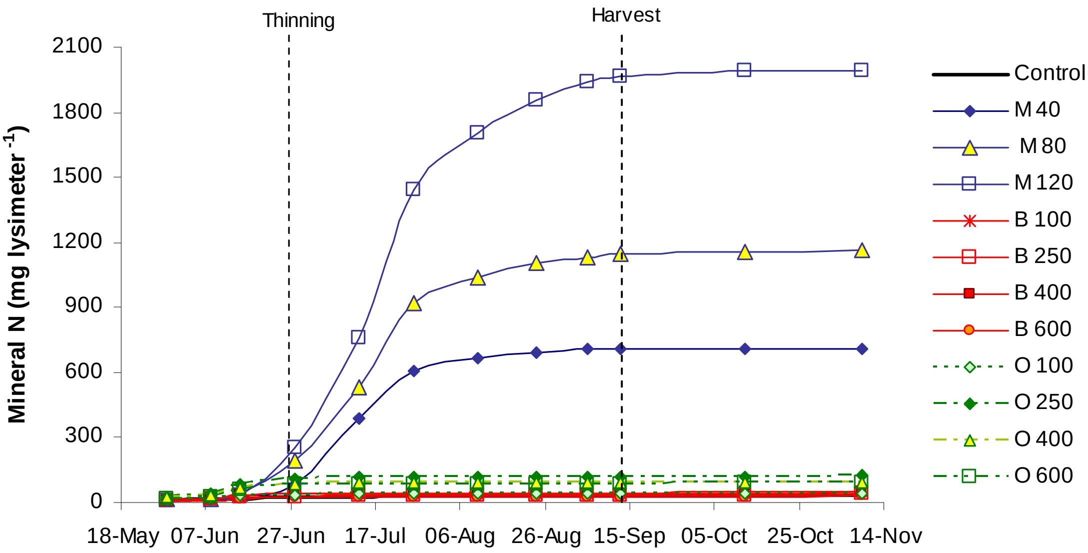

Figure 6.10 Cumulative mineral N leached from all 12 treatments (M: mineral fertilizer, B: biowaste compost, O: onion compost; 40, 80, 120: application rates in kg N ha^{-1}; 100, 250, 400, 600: application rates in kg total N ha^{-1}).

The amount of mineral N leached from the onion and the biowaste compost treatments was not significantly different (P>0.05) to that leached from the control treatment, throughout the measuring period (Figure 6.11).

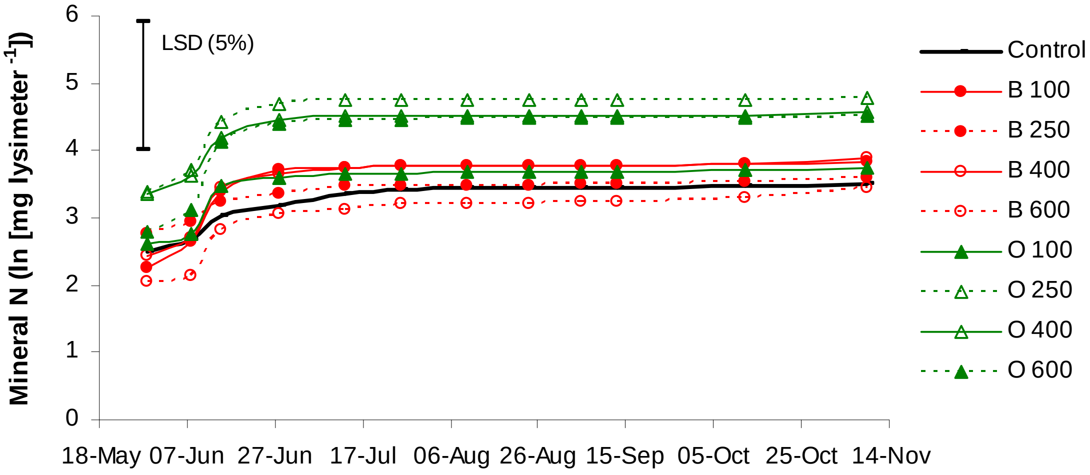

Figure 6.11 Cumulative mineral N leached from the compost treatments and the control. The values shown are the natural logarithms of the measured values (B: biowaste compost, O: onion compost; 100, 250, 400 and 600: application rates in kg total N ha^{-1}).

The finding that the mineral fertilizer resulted in significantly higher ($P<0.001$) amounts of N leached than the composts indicates environmental benefits from a water quality perspective following compost utilization in forage maize production on well drained, irrigated sandy soils, compared to mineral fertilization. Another important feature is that the effect of compost application rate on mineral N leached was observed not to be significant ($P>0.05$).

Figure 6.12 shows the effect of compost type on the total amount of mineral N leached during the experiment, in comparison to the reference treatments. Onion compost resulted in significantly higher ($P<0.05$) amount of mineral N leached in comparison to the biowaste compost. However, the mineral N leached from the biowaste or onion compost amended soil was not significantly different ($P>0.05$) from the control. The effect of compost application rate on the total amount of mineral N leached was found not significant ($P>0.05$). Of the total amount of mineral N leached, 96% was TON-N and the remaining 4% was NH_4-N.

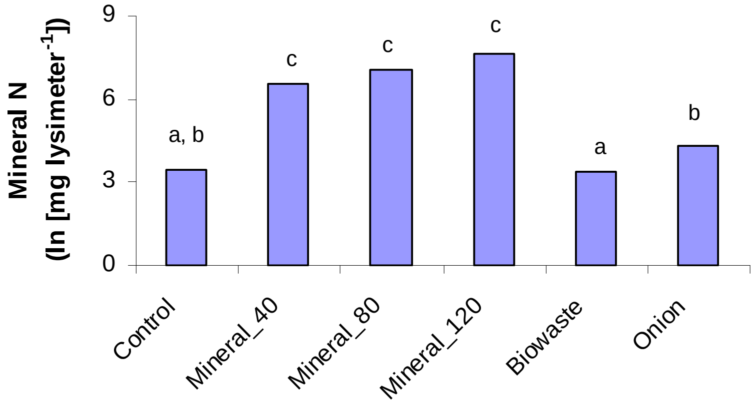

Figure 6.12 The effect of compost type on the total amount of mineral N leached in comparison to the reference treatments. The values shown are the natural logarithm of the measured values. Columns labelled with the same letter are not significantly different ($P>0.05$).

The amount of mineral N leached from the mineral fertilizer amended sand calculated as a percentage of the mineral fertilizer N applied was found 72 % at the rate of 40 kg N ha^{-1}, 61 % at the rate of 80 kg N ha^{-1}, and 71% at the rate of 120 kg N ha^{-1}. This finding explains the low crop yield response to the mineral fertilizer application, and it shows clearly that in such poor quality sandy soils the application of mineral fertilization constitutes a significant threat to groundwater quality. This excessive mineral fertilizer N leaching should be attributed to the low CEC of the sand, which impedes the available mineral N to be held in the soil, and subsequently it becomes susceptible to leaching. The amount of the mineral N leached from the compost amended sand calculated as a percentage of the total compost N applied was found 0.8% for the onion compost, and 0.3% for the biowaste compost.

6.3.2.2 Dissolved total and organic nitrogen

The amount of total dissolved N (TDN) was also measured in the leachate samples collected after June 29 (Figure 6.13), in order to determine whether there were any N leaching losses in organic forms from the compost treatments. The TDN in the leachate was found to follow the patterns of the mineral N presented in Figure 6.10, for the respective measuring period.

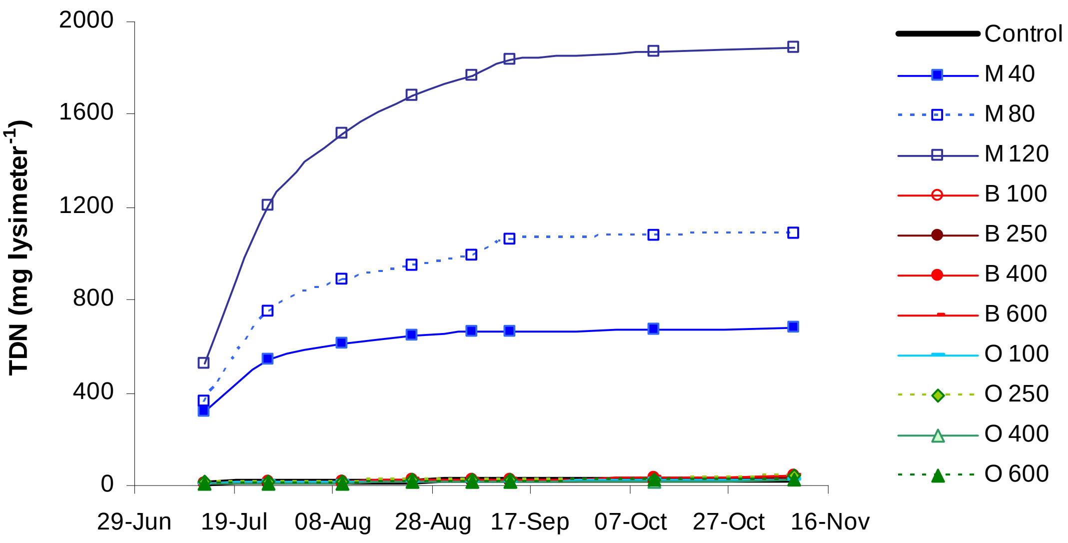

Figure 6.13 Cumulative TDN leached from all 12 treatments during the period of 29 June to 6 November 2006 (M: mineral fertilizer, B: biowaste compost, O: onion compost; 40, 80, 120: application rates in kg N ha^{-1}; 100, 250, 400, 600: application rates in kg total N ha^{-1}).

Mineral fertilizer resulted in significantly higher ($P<0.001$) amount of TDN in the leachate, compared to control and composts. The increase of mineral fertilizer application rate significantly increased (P<0.001) the TDN leached. On the contrary, both biowaste and onion composts resulted in similar amount of TDN with the control treatment (P>0.05). These findings indicate that the predominant form of nitrogen in the leachate is mineral N (in particular TON-N), irrespectively of the source of N applied: compost or mineral fertilizer N.

It should be noted that there were cases where TDN was measured lower than mineral N. This is not possible under the assumption shown by Equation 6.2 (TDN = DON + mineral N). These problems occurred for the mineral fertilizer treatments only on the dates of high mineral N leaching. Therefore, these problems should be attributed to the method of determination used, as the determination range was different for TDN and mineral N, and subsequently different dilutions were necessary to bring the values within limits. In the cases that mineral N was measured higher than TDN, DON was assumed to be zero. The estimated amount of dissolved soluble N (DON) in the leachate is presented in Figure 6.14. Compost application resulted in similar amount of DON leached with the control treatment (P>0.05), suggesting that compost application did not advance N leaching losses in organic forms.

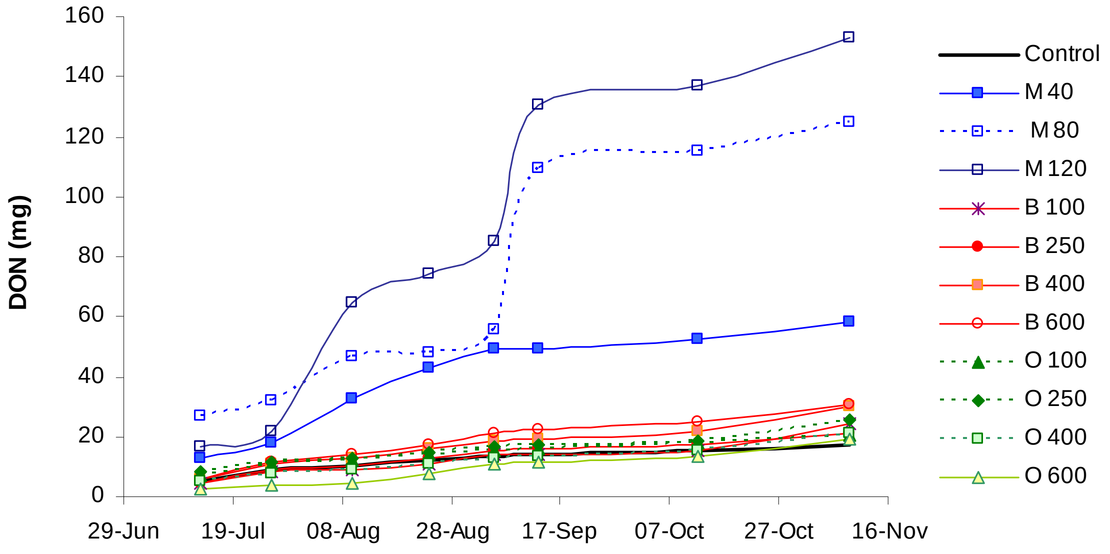

Figure 6.14 Cumulative DON leached from all 12 treatments during the period of 29 June to 6 November 2006. (M: mineral fertilizer, B: biowaste compost, O: onion compost; 40, 80, 120: application rates in kg N ha^{-1}; 100, 250, 400, 600: application rates in kg total N ha^{-1}).

The mineral and organic soluble N leaching losses results suggest that compost application to well drained sandy soils is advantageous over mineral fertilizers, and does not constitute a threat to water quality, at least in the short term.

6.3.2.3 Residual effect of compost application on potential nitrogen leaching

In summer 2007 leachate samples were collected to assess the residual effect of compost application on N leaching. The measurements were taken during this particular period, when compost N mineralization was expected to be higher due to higher temperatures. It was shown in Figure 6.10 that mineral N leaching from all treatments was close to zero from mid September 2006 (following crop harvest) until mid November when the measurements stopped under the assumption that no further N mineralization would occur during the winter months.

Nitrogen leaching (mineral N and TDN) results are shown in Figure 6.15. It can be seen that N leaching losses from the compost amended sand were at similar levels (P>0.05) as the control treatment, suggesting that increased N leaching losses at the second year after compost application are not expected.

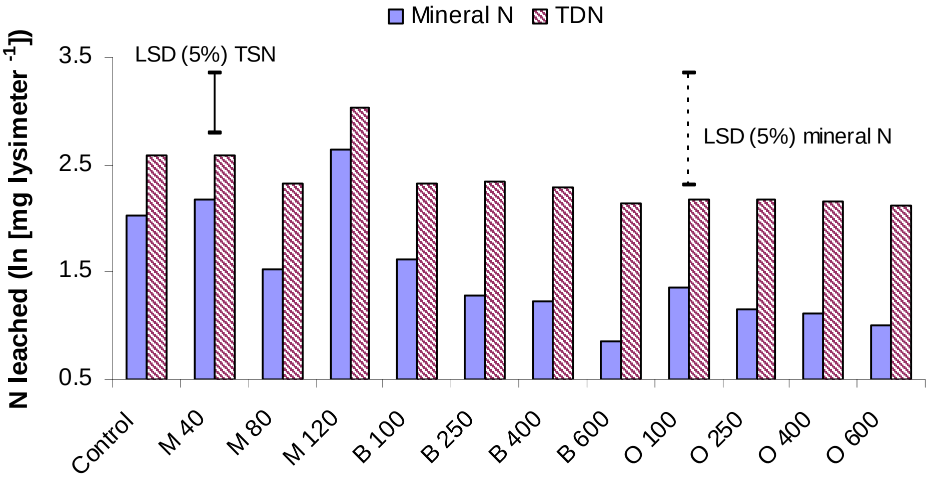

Figure 6.15 The residual effect of compost application on the total amount of mineral N and TDN leached, in comparison to the reference treatments for the period of 1 May to 19 August 2007. The values shown are the natural logarithms of the measured values.

Compost application at 100 kg total N ha^{-1} resulted in significantly lower ($P<0.05$) DON leached than the higher rates. However, the DON leached from the control was not significantly different ($P>0.05$) from all the other treatments (Figure 6.16), indicating that increased N leaching losses in organic forms at the second year following compost application are not expected.

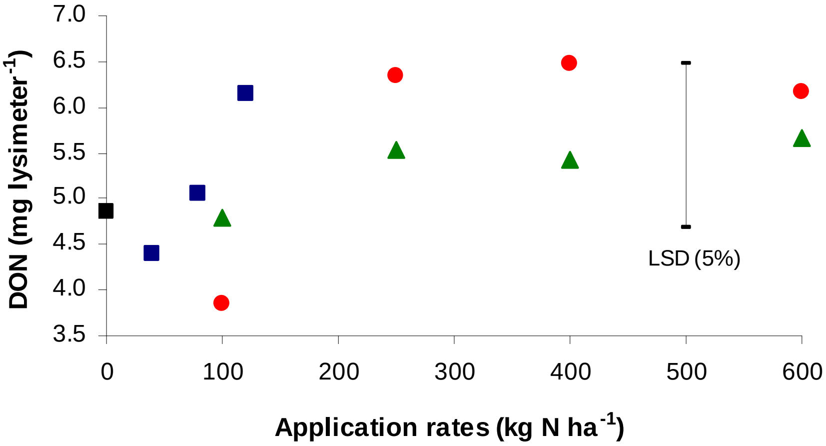

Figure 6.16 The residual effect of compost application on the total amount of DON leached, in comparison to the reference treatments for the period of 1 May to 19 August 2007 (■: Control, ■ : Mineral, ●: Biowaste, ▲: Onion).

6.3.2.4 Phosphate-phosphorus leached

Phosphate P concentration in the leachate was also measured for the period June 29 to November 6, 2006. The amount of phosphate P leached was very small for all treatments and sampling dates. The mean value, of all treatments, of the total amount of phosphate P leached over the measuring period was 0.47 mg per lysimeter (corresponding to about 0.02 kg P ha^{-1}). As shown in Figure 6.17, the mineral fertilizer treatments resulted in significantly lower amount of P in the leachate, compared with the compost treatments (The four reference treatments are summarized in one ‘mineral’ treatment, since they all received the same amount of P, and the differences between them were not significant ($P>0.05$)). This finding indicates that phosphate P was more susceptible to leaching from the compost amended sand than the mineral fertilized one.

Other work (Eghball et al., 1996; Anderson & Magdoff, 2005) has also shown that the organic additions to the soil increase the downward movement and the leaching potential of P. According to McGechan & Lewis (2002), the main concept explaining these findings is related to the transport of inorganic P in association with organic colloids (which can possibly be derived from animal manure, slurry, plant residues, or soil organic matter) through the soil by macropore flow.

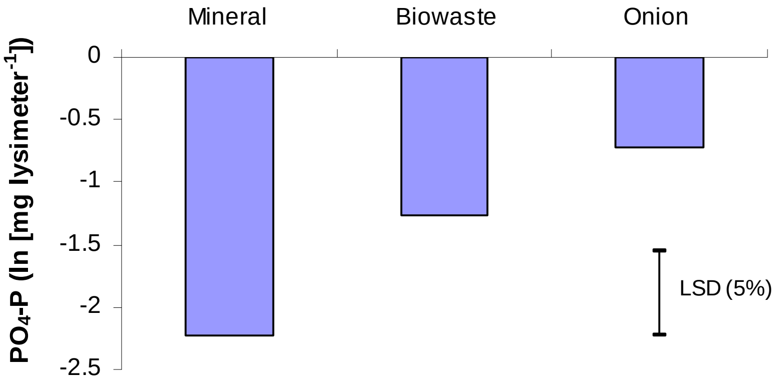

Figure 6.17 The amount of phosphate P in the leachate during the period of June 29 to November 6. The values shown are the natural logarithms of the measured values.

6.3.3 Sand properties

6.3.3.1 Organic matter, total nitrogen, total carbon and pH

At the end of the growing season, sand OM content and pH were found not to be significantly influenced ($P>0.05$) by compost application or mineral fertilization. Sand total N content was below 0.03% for all treatments, which is the lowest detectable limit within the method used.

Compost application at rates higher than 100 kg total N ha^{-1} produced significantly higher ($P<0.001$) total C content, in comparison to the reference treatments. Control total C content was 0.08%, whereas the mean total C value for the compost amended sand was 0.19%, indicating an increase of sand total C content by 126% due to compost application. Biowaste and onion compost resulted in similar levels ($P>0.05$) of sand total C content. The increase of compost application rate significantly increased

(P<0.001) total C content. Figure 6.18 shows the linear relationship between the amount of total C applied by the composts and the sand total C content.

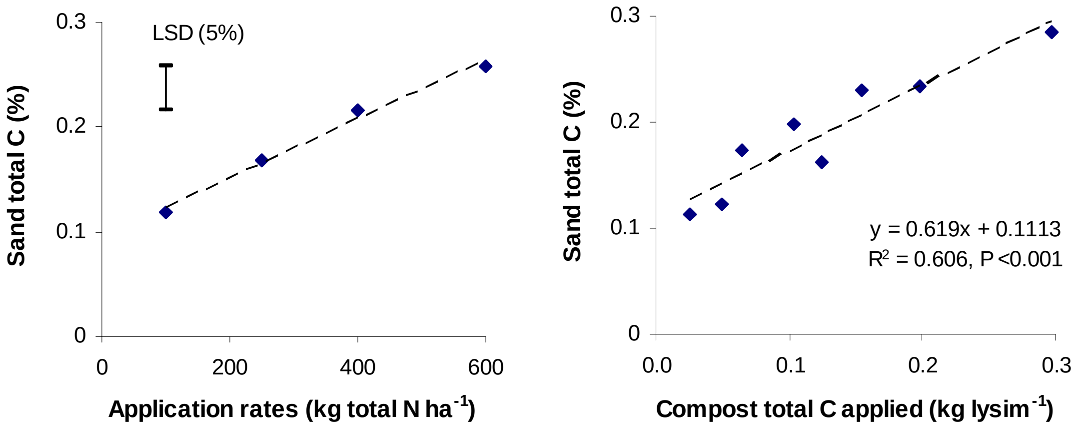

Figure 6.18 Sand total C content at the end of the crop growing season: the effect of compost application rate (left), and the relationship between compost total C applied and sand total C content (right).

6.3.3.2 Extractable potassium, sodium, electrical conductivity, exchangeable sodium percentage and mineral nitrogen

The sand extractable K and Na levels were significantly increased ($P<0.05$) due to compost application, in comparison to the reference treatments, by about 20% and 21%, respectively. Both biowaste and onion compost resulted in similar levels ($P>0.05$) of extractable K and Na (Figure 6.19). The four reference treatments are summarized in one ‘mineral’ treatment, since they all received the same amount of K, and the differences between them were not significant ($P>0.05$). The effect of compost application rate on sand K and Na levels was found not significant ($P>0.05$).

Despite the increase of sand Na levels, sodicity problems are not expected under these sandy soil conditions and especially under irrigation. The ESP of the composts amended sand at the highest rates of application was about 0.4%. Control ESP was also 0.4%, whereas the ESP of the mineral fertilizer amended sand at the highest rate of application was about 0.5%. ESP critical level for sodicity problems is 15%. EC_e for the control and the composts and mineral fertilizer amended sand at the highest reates of application was 0.5 dS m^{-1}, while the critical level for most crops is 4 dS m^{-1}. These findings indicate that salinity and sodicity problems are unlikely even at high rates of compost application to poor sandy soils.

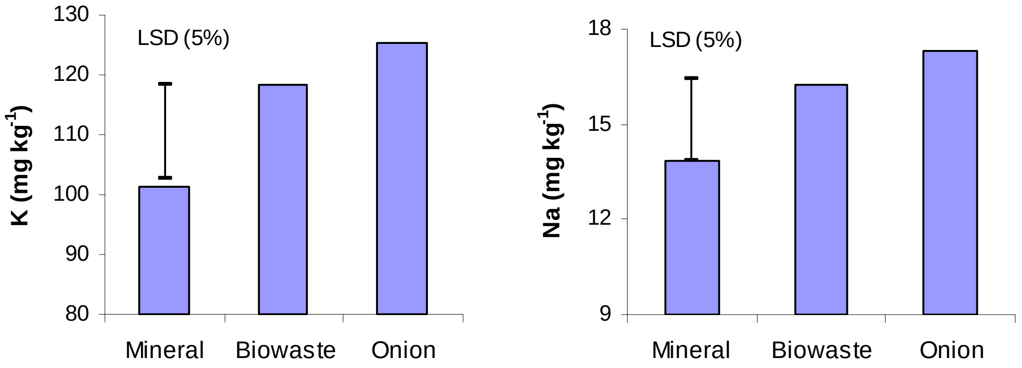

Figure 6.19 The effect of biowaste and onion compost application on sand available K (left) and Na (right) content in comparison to the mean of all reference treatments.

The sand mineral N content at harvest (residual mineral N) was found to be low for all treatments and depths. The effect of depth was found not to be significant (P>0.05). The total residual mineral N content (sum of all 3 depths) resulted from the mineral fertilizer applied at the rate of 120 kg N ha^{-1} was found significantly higher (P<0.05) than from the compost amended sand (Figure 6.20). The effect of compost application rate was found not significant (P>0.05).

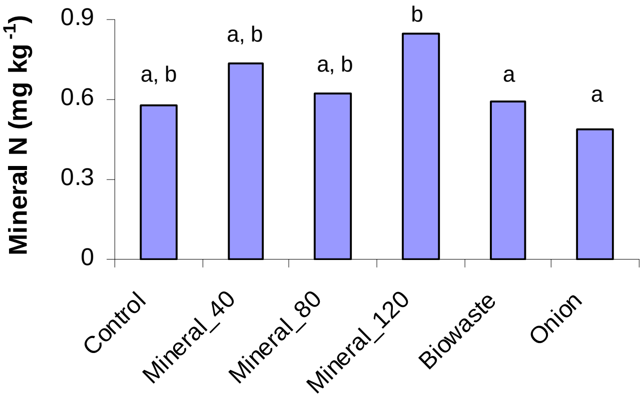

Figure 6.20 Sand mineral N content (0-0.65 m depths) at harvest resulted from compost and mineral fertilizer application. Columns labelled with the same letter are not significantly different (P>0.05).

The sand mineral N content within the top 0.25 m depth at thinning is shown in Figure 6.21. It can be seen that N availability from the biowaste compost amended sand was at similar levels to the control ($P>0.05$). The onion compost applied at rates higher than 100 kg total N ha^{-1} resulted in levels similar to the mineral N fertilized soil ($P>0.05$). The effect of mineral fertilizer rate was found not significant ($P>0.05$) for the rates higher than 40 kg N ha^{-1}. This finding indicates that N leaching below the top 0.25 m depths had already started before the thinning.

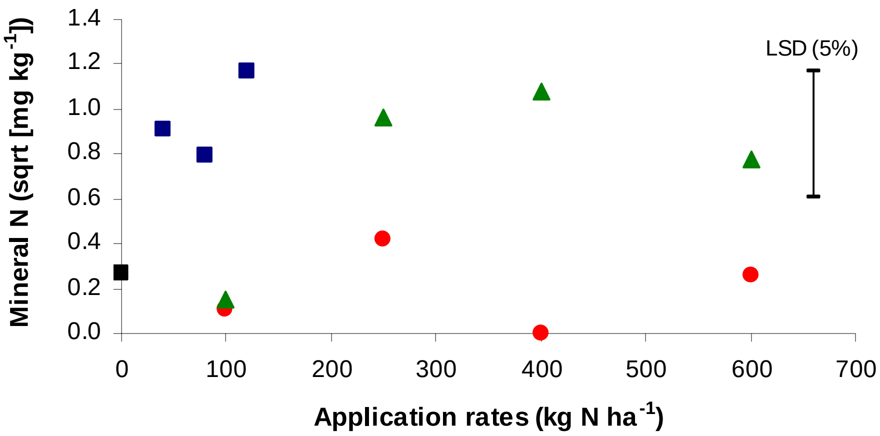

Figure 6.21 Sand mineral N content (0-0.25 m depths) at thinning resulted from compost and mineral fertilizer application at the different rates (■ : Control, 🟦 : Mineral, ●: Biowaste, ▲: Onion).

6.3.3.3 P availability

Compost application significantly increased the sand available P levels. The soil analysis at harvest showed that compost application resulted in significantly higher ($P<0.001$) sand available P levels, compared to the mineral fertilizer (Figure 6.22). The onion compost increased the sand available P levels by about 350%, and the biowaste by about 50%, compared to the mineral fertilizer.

Sand available P levels were also measured at thinning. At that stage, only the onion compost application resulted in significantly higher ($P<0.001$) sand P content. Table 6.4 presents the P availability levels measured at the beginning of the experiment (before any P addition), at the crop thinning (after initial P fertilization and compost application) and at harvest (after supplementary P fertilization). The 'check' treatment represents non-amended sand (no fertilizer P, K, N or Mg addition and no compost addition).

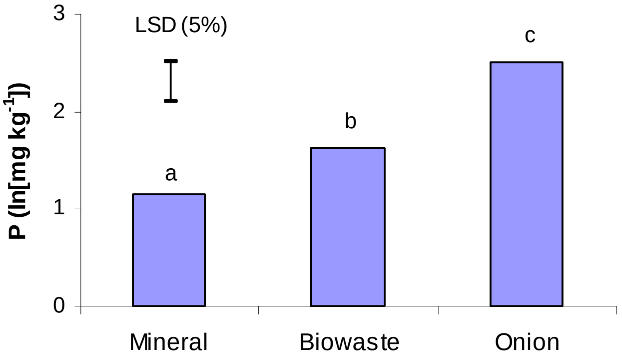

Figure 6.22 The effect of mineral fertilizer, biowaste and onion compost application on residual sand available P content (0-25 cm depth). The values shown are the natural logarithms of the measured values. Columns labelled with the same letter are not significantly different (P>0.05).

As shown in Table 6.4, from the beginning of the experiment to thinning the sand available P content increased in all treatments (mineral fertilizer, biowaste and onion compost), in response to the initial P fertilization (1.14 g fertilizer P and compost P). However, between thinning and harvest, sand available P levels kept increasing only in the onion compost amended sand, despite the fact that supplementary P fertilization (1.14 g) was added to all treatments.

Table 6.4 Sand P availability within the 0-0.25 m depths (mean values) at the beginning of the experiment, at crop thinning, and at harvest, resulted from the mineral fertilizer, biowaste and onion compost application, and the non-amended 'check' sand.

Treatment	**Available P (g lysimeter^{-1})**		
	Start	**Thinning**	**Harvest**
Check	0.05	0.10	0.03
Mineral	0.05	0.29	0.33
Biowaste	0.05	0.40	0.49
Onion	0.05	0.73	1.48

Table 6.5 presents the P budget from the beginning of the experiment to the crop harvest.

Table 6.5 Phosphorus budget (mean values) for the mineral fertilizer, biowaste and onion compost treatments (the values shown are per lysimeter).

Inputs	**(g)** (1)	**Outputs**	**(g)** (2)	**Un-accounted**	**(g)** (3)
Mineral Fertilizer					
available P (start) *	0.05	plant uptake	0.01		
fertilizer P	2.28	available P (harvest) *	0.33		
compost P	0				
Total	2.33	**Total**	0.34	**Total**	**1.99**
Biowaste Compost					
available P (start) *	0.05	plant uptake	0.02		
fertilizer P	2.28	available P (harvest) *	0.49		
compost P	0.16				
Total	2.49	**Total**	0.51	**Total**	**1.97**
Onion Compost					
available P (start) *	0.05	plant uptake	0.11		
fertilizer P	2.28	available P (harvest) *	1.48		
compost P	0.44				
Total	2.77	**Total**	1.59	**Total**	**1.18**

* Available P both at start and harvest refers to the top 0.25 m depths of sand.

There are a few important features to note regarding Table 6.5. Firstly, the un-accounted P (column 3) for all treatments was relatively high (85.5%, 79.4%, and 42.5% of the total P inputs for the mineral fertilizer, biowaste, and onion compost, respectively). Secondly, the amount of P present in the plant roots was not taken into consideration in the unaccounted losses. The crop P uptake by the aboveground biomass was relatively small, especially in comparison to the amount of fertilizer P added. The amount of P in the roots is not likely to change the amount of un-accounted P significantly, which might be supported by the observed P deficiency.

One possible explanation for the phenomenon could be P precipitation by Ca. The sand pH was slightly alkaline (7.5) favouring the formation of Ca phosphates with low solubility. According to Thompson & Troeh (1978), the most favourable pH for P availability is near neutral to slightly acid. Ca phosphates have such low solubility under alkaline conditions that P deficiencies commonly result at pH values near 8. Thus, it was hypothesized that the available P added as superphosphate may have been precipitated by Ca and therefore it was made unavailable to the crop.

Measurements of the sand total P and available Ca content at harvest were carried out in order to test the above mentioned hypothesis. The results are shown in Table 6.6. The amount of P precipitated was assumed to be equal to the difference between the net total P and available P at harvest. Net total P was calculated as the total P of each treatment minus the 'check' total P. Net total P and estimated precipitated P values are also shown in Table 6.6 (columns 4 and 5, respectively).

Table 6.6 Mean values of sand extractable Ca, total P, net total P and estimated precipitated P at harvest (0-0.25 m depths) resulted from the mineral fertilizer, biowaste and onion compost application, and the non-amended 'check' sand (the values shown are per lysimeter).

Treatment (1)	**Extr. Ca (g)** (2)	**Total P (g)** (3)	**Net total P (g)** (4)	**Precipitated P (g)** (5)
Check	28.0	7.14		
Mineral	25.5	9.07	1.93	1.60
Biowaste	30.9	7.69	0.55	0.05
Onion	33.6	8.64	1.50	0.02

As shown in Table 6.6, a large amount (80%) of the un-accounted P (Table 6.5 – column 3) of the mineral fertilized sand was found in unavailable forms (precipitated P/ Table 6.6 – column 5) within the top 0.25 m depths, supporting the hypothesis that mineral P was fixed and thus was unavailable to the crop. The Ca results do not show a clear relationship between precipitated P and extractable Ca levels. However, compost application significantly increased ($P<0.05$) the extractable Ca levels in comparison to the mineral fertilized sand. Hence, and taking into consideration the alkaline pH and the low clay and organic matter content of the sand, P precipitation by Ca seems to be the most possible explanation for P unavailability in the mineral fertilizer treatment.

The amount of P which was found to be precipitated for the compost treatments was very low (less than 3%). This finding demonstrates that P solubility/availability was significantly affected by compost application. The effect of compost application can be explained as a direct effect of adding compost plant available P and an indirect effect of adding organic matter. According to Bear (1964), and Thompson & Troeh (1978), the mineralization of organic phosphates by the decomposition of OM in itself is not considered a sizeable source of P for plants. However, decomposing OM releases acids which increase the solubility of Ca phosphates. The organic anions produced by microorganisms, complex the Ca and thereby solubilise the relatively insoluble soil phosphates.

So, after determining the amount of P precipitated within the top 25 cm, the new un-accounted P losses are 0.39 g, 1.92 g, and 1.16 g for the mineral fertilizer, the biowaste and the onion compost, respectively (calculated as the difference between the respective amounts of P in Table 6.5, column 3 minus the amounts of P in Table 6.6, column 5). These new un-accounted amounts of P may have been leached, and/or moved below the top 25 cm of sand. The amount of P measured in the leachate during the period from June 29 to November 6 was very low (less than 0.001 g) as shown in paragraph 6.3.2.4, indicating therefore that the P losses should be mainly attributed to movement of P downwards the soil profile (below the top 0.25 m depths) but with limited leaching.

Another feature to note is that the effect of compost type and application rate on P availability was significant (both at thinning and harvest). Onion compost application resulted in significantly higher ($P<0.001$) P availability than the biowaste compost. This finding could possibly be attributed to the different nature of organic matter added with each compost type (Bear, 1964), to the different amount of compost available P added, and/or to the different compost N availability.

Figure 6.23 presents the relationship between compost soluble P and crop P uptake. Although the increase of the application rate of both composts resulted in linear increase of P uptake, it can be seen that onion compost was more efficient to provide the P required.

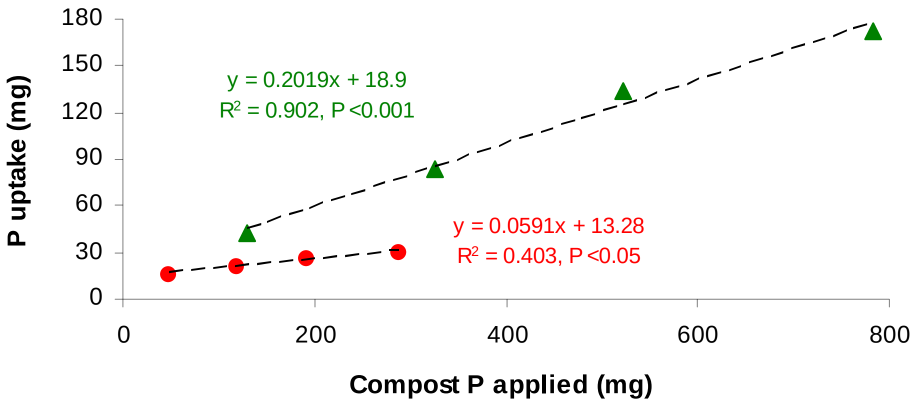

Figure 6.23 The amount of plant P uptake in relation to the amount of compost P applied, per lysimeter (●: Biowaste, ▲: Onion).

Higher compost application rates also resulted in higher amounts of organic matter applied. Figure 6.24 shows the relationship between maize P uptake and compost OM applied. Figure 6.24 supports the concept of the different nature of the organic matter added with each compost type. Crop P uptake was more responsive to the application of onion compost organic matter than of biowaste compost.

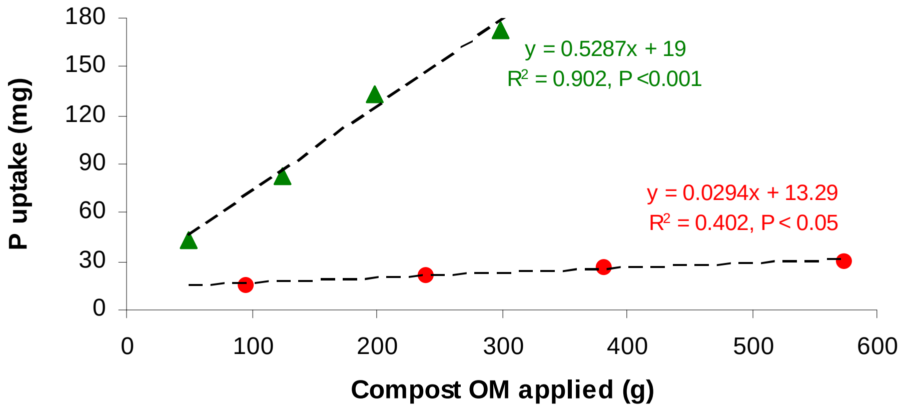

Figure 6.24 The amount of crop P uptake in relation to the amount of compost organic matter applied per lysimeter (●: Biowaste, ▲: Onion).

It can be seen from Figures 6.23 and 6.24 that forage maize P uptake response to compost OM applied follows the same pattern with the P uptake response to compost available P for each compost type. This finding suggests that both factors (compost available P and OM) may have influenced P uptake.

Biowaste compost application resulted in lower N availability, in comparison to the onion compost, and consequently in less plant and root growth (Figure 6.25). It should be expected therefore, that the plants which grew in the onion compost amended sand were more efficient to absorb P resulting in less P losses (Figure 6.17). However, the co-effect of P and N availability on plant and root development should be considered. Early season P availability is critical to produce a healthy root system (Grant et al., 2001).

Figure 6.25 Forage maize root development (at harvest) resulted from the mineral fertilizer (top), onion (middle) and biowaste compost (bottom) application at the highest rate.

In summary, the application of the mineral fertilizer P resulted in lower sand available P levels and also in lower maize P content and P uptake, compared to the compost treatments. These findings demonstrate the low use efficiency of mineral fertilizer P in the treatments where no compost was added. This low efficiency was mainly attributed to P fixation (precipitation by Ca). The application of composts increased P solubility and P use efficiency, but also it enhanced P transport downwards the soil profile.

The results of sand available P and N content in combination with the plant P and N uptake, throughout the study, indicate that forage maize production in the different treatments (mineral fertilizer, biowaste and onion compost) was co-affected by P and N availability. The application of the onion compost resulted in the highest crop production (Figure 6.5), in comparison to the mineral fertilizer and biowaste compost, as it resulted in relatively high availability of both N and P. Crop production in the mineral fertilized sand may have been predominantly restricted by the P deficiency, as indicated by the results at thinning, which showed that sand available P contents were low, whereas sand mineral N content and N uptake were high. Early season P availability is critical for optimum crop growth and development (Grant et al., 2001). Crop production in the biowaste compost amended sand seems to have been restricted by both P and N deficiency.

6.3.3.4 Water retention

Water retention was significantly increased ($P<0.05$) by compost application. The increase was significant up to soil water potential of -100 kPa, whereas closer to the permanent wilting point the volumetric water content was similar to the control (Figure 6.26).

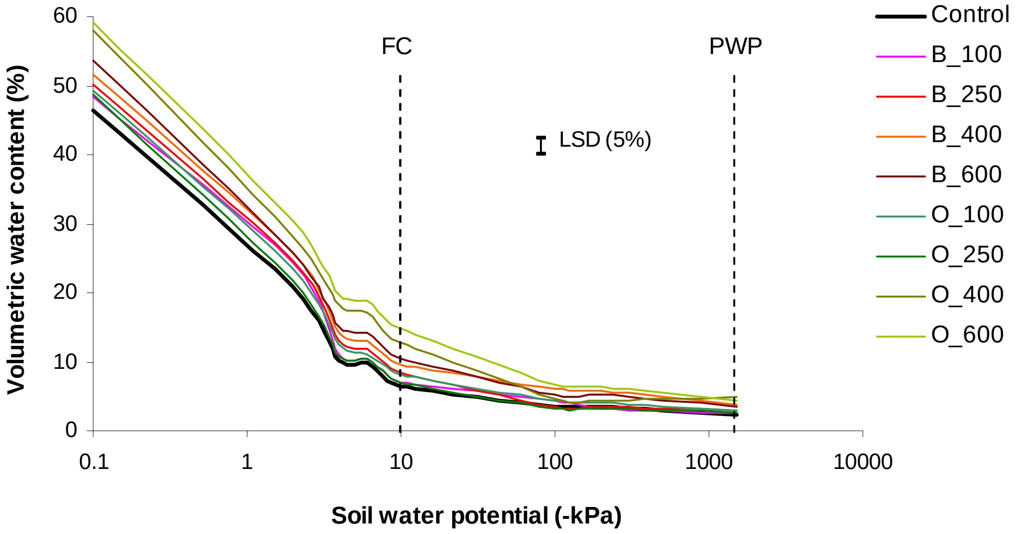

Figure 6.26 Water release curve as affected by compost application at different rates (B: biowaste compost, O: onion compost; 100, 250, 400, 600: application rates in kg total N ha^{-1}).

The available water to the crop (the water between field capacity (FC) and permanent wilting point (PWP)), was also found significantly higher ($P<0.001$) for the compost amended sand soil, when compost was applied at rates higher than 250 kg total N ha^{-1}. Onion compost application resulted in significantly higher ($P<0.05$) available water levels compared to the biowaste compost. Figure 6.27 presents the soil available water content as affected by biowaste and onion compost application at different rates.

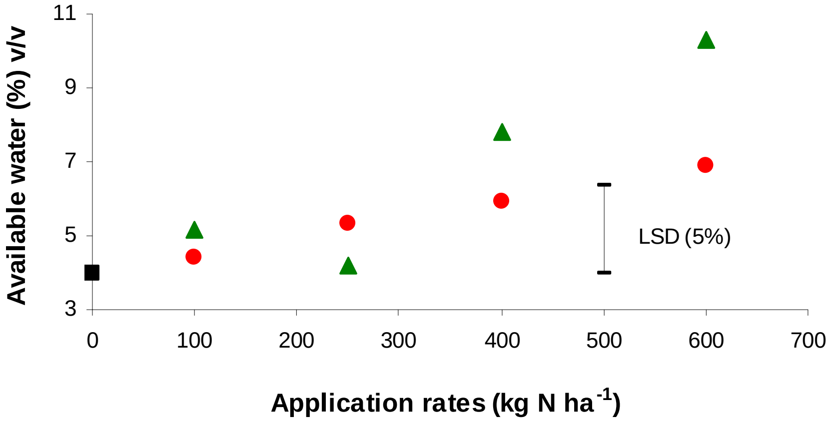

Figure 6.27 Soil available water content (v/v) as affected by compost application at different rates (■ : Control, ●: Biowaste, ▲: Onion).

6.4 Summary of major findings

The extreme soil conditions employed in the lysimeter trial, although they are not typical of agricultural soil, provided with interesting and useful results regarding the potential effects of using compost in agriculture.

The application of mineral fertilizer N, P and K only, with no compost addition, to the poor quality, well drained sandy soil used in the lysimeter trial was demonstrated not to be able to enhance forage maize production because of reasons related mainly to P and N unavailability. The majority (approximately 80%) of the fertilizer P applied was found fixed in the soil (0-0.25 m) and thus was unavailable to the crop (there is evidence to suggest that P fixation was due to P precipitation by Ca). Forage maize grown on mineral fertilized soil showed P deficiency symptoms from early growth stages and it did not respond to the supplementary fertilization with P. Hence, crop development was restricted.

Crop recovery of mineral fertilizer N was low (about 10%). The application of mineral fertilizer N resulted in excessive N leaching. About 70% of the mineral fertilizer N applied leached and thus it was made unavailable for crop production, whilst deteriorating water quality. The fertilizer N leaching losses occurred during the crop growing season, and therefore limited amount of mineral fertilizer N was available for crop uptake, thus restricting crop development.

Compost application to poor quality, well drained sandy soils was found capable of increasing forage maize production by improving soil quality and by supplying crop essential nutrients. The effects of compost application on forage maize production, N availability and soil properties were found to depend significantly on the compost type and application rate used.

The application of biowaste compost resulted in low N availability and did not increase crop production. The application of onion compost at rates higher than 250 kg total N ha^{-1} was found to be the best practise to increase forage maize yield under the conditions of the study.

Compost C: N ratio was found to be an important factor controlling N availability. The application of biowaste compost (C: N ratio of 20) to the sandy soil at rates higher than 250 kg total N ha^{-1} resulted in an initial immobilization of soil N. There was limited mineralization of the biowaste compost N later in the season and the mean crop N recovery was about 1.1% of the total N applied. Consequently, N was a factor restricting forage maize production in the biowaste compost amended soil. Onion compost with a C: N ratio of 10.5 resulted in mineralization of compost N. Mean crop recovery of onion compost N was about 6.4%. These findings support the case that compost C: N ratios between 9 and 15 exclude negative yield effects because of high N immobilization (Eriksen et al., 1999; Nevens & Raheul, 2003).

Compost application to the well drained sandy soil did not increase N leaching. The amount of the mineral N leached from the compost amended sand was 0.8% and 0.3% of the total compost N applied, for the onion and biowaste compost, respectively. The effect of compost application rate was not significant. Compost application did not result in increased N leaching losses in organic forms. There is evidence to suggest that compost application is not likely to result in excessive N leaching the following year after compost application, even if the soil is left in fallow.

Phosphorus availability was found to depend on compost application. Compost addition to soils increases P solubility, thus preventing P fixation. Depending on the compost type, and in particular the compost soluble P content and organic matter quality, compost application can result in the movement of inorganic P downward the soil profile (below the top 0.25 m), especially on sandy soils where the macropore flow is enhanced and inorganic P is transported in association with organic colloids, under the concept explained by McGehan and Lewis (2002). However, excessive P leaching is unlikely. Further work is necessary in order to determine the effects of compost OM on P availability.

Compost application to poor sandy soils improves soil quality by mainly increasing soil total C, extractable K, and available water content. About 60% of the total compost C applied was found to be added to the soil as total soil C content, irrespectively of the compost type. Soil K content was also found increased by about 20% in comparison to non-compost amended soil. The effects of compost type and application rate were not clear. Soil available water content was increased by compost application at rates higher than 250 kg total N ha^{-1}. This increase was mainly due to increased soil water retention capability close to FC. Soil water retention close to PWP was not enhanced.

6.5 Conclusions

The conclusions drawn from the lysimeter study are as follows:

a) In poor quality, well drained, sandy soils mineral fertilization is not likely to enhance crop production, and it constitutes a severe environmental threat due to excessive N leaching.
b) Compost application does not result in increased N leaching, thus it is

advantageous over mineral fertilizers from a water quality perspective.

c) Compost application increases the levels of total carbon, available phosphorus and potassium, and water retention of sandy soils, thus improving soil quality and consequently crop production.

d) There is evidence to suggest that compost application increases P solubility and hence adding compost to alkaline and calcareous soils is likely to prevent P fixation by Ca.

e) The application of biowaste compost to poor quality and well drained sandy soils may enhance the downward movement of available P and thus adversely affect crop production and, less likely, the water quality.

f) The land application of biowaste compost, characterized by a C: N ratio of 20, is likely to result in N immobilization and therefore have negative effects on crop production.

g) The application of onion compost, with a C: N ratio of 10.5, to forage maize production on sandy soil results in mineralization of compost N and crop N recovery of about 5.5-7.5% of the compost total N applied, thus enhancing crop yield.

h) The increase of compost application rate may increase crop yield without increasing N leaching. The application of onion compost (C: N ratio of 10.5) to a poor quality soil with low total and available N levels, in rates higher than 250 kg total N ha^{-1} increases forage maize production without increasing N leaching.

7 Incubation study: compost nitrogen mineralization

7.1 Introduction

Chapter 7 focuses on the study of the compost N mineralization under controlled laboratory conditions. The field, pot and lysimeter experiments studied the compost N mineralization potential on the soil-compost-plant-water system, under limited-controlled climatic conditions. The incubation trial aimed to assess the mineralization of compost N on the soil-compost-water system under controlled moisture and temperature conditions.

Furthermore, the incubation assay aimed to evaluate the effect of compost application on the soil organic soluble N pool (OSN) intending mainly to provide supportive evidence towards the N leaching in organic forms (DON – dissolved organic N) studied in the pot and lysimeter experiments.

The specific objectives of the incubation trial are presented below:

1. To evaluate the N mineralization potential of biowaste and vegetable waste compost application under defined conditions of temperature and moisture.
2. To determine the effect of compost application on the soil organic soluble N.

7.2 Materials and methods

7.2.1 Soil

The soil used for the incubation experiment was loamy sand (commercially available from Bailey’s, Norfolk). It was selected as an intermediate case (in terms of texture) of the sandy and the sandy loam soils used in the other experiments. Some properties of the soil at the beginning of the study are presented in Table 7.1. Soil was sieved to 2 mm and air-dried.

Table 7.1 Soil properties at the beginning of the experiment.

Sand	Silt	Clay	pH	OM	Mineral N	Total N	OSN	C: N ratio
(%)	(%)	(%)		(%)	(%)	(%)	(%)	
85.9	5.79	8.31	6.7	3.41	6.58	0.12	8.69	13.20

7.2.2 Composts

Biowaste and onion composts were used in the incubation study, taken from the same batches as those used for the field, pot and lysimeter experiments in 2006. The batches were sub-sampled in September 2006. The collected samples were left to air-dry and sieved (2 mm). Compost analysis for mineral N was undertaken on both the fresh sample (water extract, 1+5 v/v (BS EN 13652: 2001)) and the dried sample (KCl extract, 1+5 v/v). Compost OSN was determined on the dried sample (K_2SO_4 extract, 1+5 v/v). Compost analysis for mineral N and OSN on the air-dried sample was based on the methods used for measuring these properties in soils (see paragraph 7.2.4), adapting to 1+5 compost/ extractant volume ratio. The other compost parameters were determined according to the standard methods described in Chapter 3.

Table 7.2 presents some chemical properties of the two composts at the beginning of the experiment. Mean NH_4-N content measured in the fresh sample extract was 10.3 and 3.6 mg kg^{-1}, whereas TON-N was 108.5 and 792.2 mg kg^{-1}, for the biowaste and onion compost respectively.

Table 7.2 Biowaste and onion compost chemical properties.

	Mineral N ‡	Mineral N	OSN	Total N	Total C	C: N ratio	OM	pH ‡	EC ‡
	(%)						(%)		(dS m^{-1})
Biowaste	0.01	0.01	0.05	2.31	35.34	15.3	60.85	9.1	0.7
Onion	0.08	0.06	0.01	1.24	11.58	9.3	19.52	8.1	1.6

‡ Analysis undertaken on the fresh sample.

7.2.3 Incubation

For the incubation, 100g of the air-dried (2 mm sieve) soil was placed in the incubation pots (plastic pots of 5 cm diameter) (dry bulk density was 1.46 g ml^{-1}). The soil moisture was adjusted to 40% FC by adding deionised water and mixing. The soil was incubated under constant temperature of 20 °C to achieve stable soil conditions (Bowden, 1997), for 8 days. After the 8 days, the soil in the pots was amended with biowaste and onion compost, thoroughly mixed at three rates of 125, 250 and 500 kg total N ha^{-1}. The rates of compost applied per pot were calculated on a volumetric basis assuming a soil depth of 0.15 cm and a bulk density of 1.46 g ml^{-1}. Table 7.3 shows the amount of compost applied per pot at the different rates. Control (non-amended soil) was used as reference treatment.

Table 7.3 The amounts of biowaste and vegetable waste composts applied to 100 g of soil at the different rates.

Application rates (kg total N ha^{-1})	**Composts (g)** Onion	Biowaste
125	0.48	0.25
250	0.95	0.50
500	1.90	0.99

Deionised water was added to bring all soils and soil-compost mixtures to a moisture content of 60% FC, to ensure minimum N losses by denitrification (Rowell, 1994), and this was adjusted weekly. To ensure uniform moisture content the soil and soil-compost mixtures were mixed thoroughly after the water addition. The pots were covered with perforated foil to minimize water loss (Fig 7.1a). Soil FC was determined by the sand table method (Harst et al., 1975) at -10 kPa soil water potential equal to 11.7% gravimetric moisture.

The pots were randomized and incubated in growth chambers in the dark (Figure 7.1b), at a constant temperature of 28 °C. The duration of the trial was 16 weeks (112 days), approximating one growing season of forage maize. The temperature of 28 °C is higher than the average temperature during maize growing season, but it was to ensure that mineralization and immobilization would be more pronounced and allow more precise measurement. The sampling was destructive, by removing pots in triplicate for each treatment at each sampling date. Sampling was at days 0, 3, 7, 14, 21, 28, 42, 56, 84 and 112.

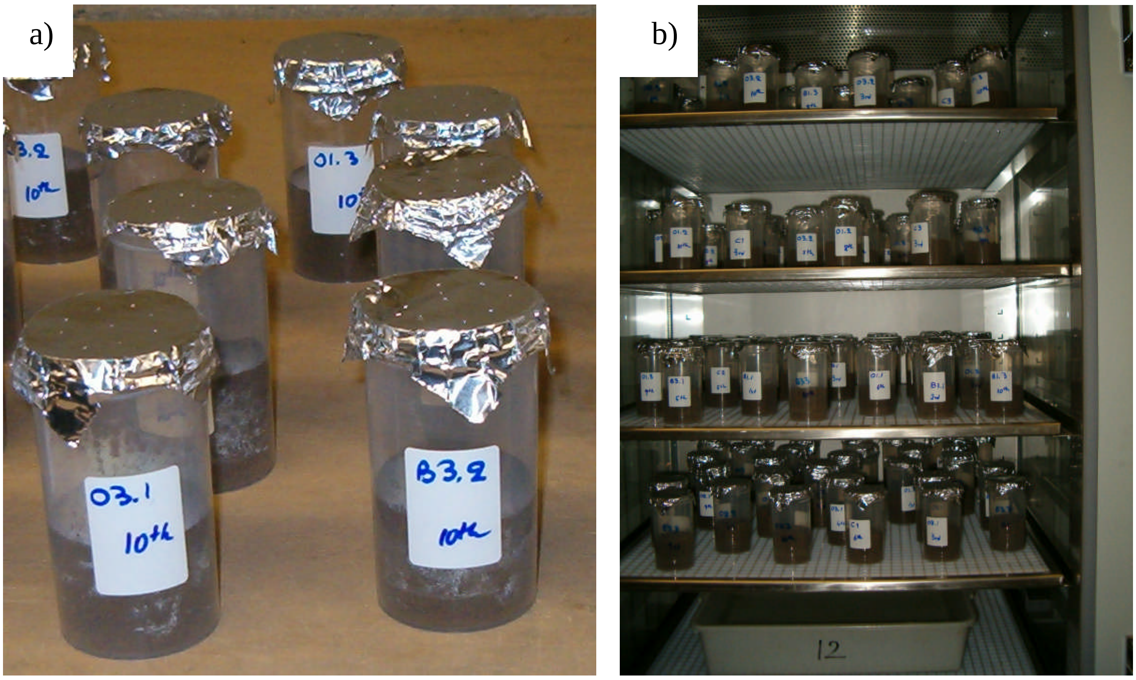

Figure 7.1 Incubation trial: The soil-compost-water system within the plastic pots on day 0. The pots are covered with perforated foil to minimize water losses by evaporation (a). The pots were placed in two growth chambers. On the bottom of each chamber there was a tray full of water for keeping the air humidity high within the cabinet (b).

7.2.4 Measurements and analysis

Soil ammonium N (NH_4-N) and total oxides of nitrogen (nitrate- and nitrite-N) (TON-N) content was measured in potassium chloride (KCl) extract by segmented flow analysis. Soil mineral N content was calculated as the sum of NH_4-N and TON-N.

The N mineralization rate (NMR) was estimated as a percentage of the compost total N applied (NMR_{total}) according to Equation 7.1 (Chaves et al., 2004), and as a percentage of the compost organic N applied (NMR_{org}) according to Equation 7.2 (Rogers et al., 2001).

$$NMR_{total}\ (\%) = 100 \times \frac{Mineral\ N\ amended\ soil\ - Mineral\ N\ control}{Total\ N\ applied} \tag{7.1}$$

$$NMR_{org}\ (\%) = 100 \times \frac{Mineral\ N\ amended\ soil\ - Mineral\ N\ control - Mineral\ N\ Compost}{Organic\ N\ applied} \tag{7.2}$$

The compost mineral N (Equation 7.2) was taken to be equal to the amount of mineral N determined in the fresh compost. Compost organic N (Equation 7.2) was calculated as the difference between the total compost N applied and the compost mineral N.

Rogers et al. (2001) calculates the compost organic N as the difference between the total and NH_4-N only. Such discrimination between the two forms of mineral N was considered more suitable for other types of waste (e.g. animal manures) that may contain up to 50% of the total N as NH_4-N and much lower percentage of TON-N. In our case, compost mineral N was low (<0.1%, as shown in Table 7.2) and also TON-N was higher than NH_4-N. Thus such discrimination was not considered appropriate.

The amount of soil organic soluble N (OSN) was estimated as the difference between the total soluble N (TSN) and the mineral N measured for each treatment and sampling date (Equation 7.3), as the OSN pools in soils cannot be measured directly by extraction (Murphy et al., 2000). Soil TSN was measured in potassium sulphate (K_2SO_4) extract (Jones & Willett, 2006) by segmented flow analysis.

$$OSN = TSN - Mineral\ N \tag{7.3}$$

Details regarding the methods involved in the soil analysis can be found in Appendix A.2.

7.2.5 Statistical analysis

The effects of each treatment and the influence of compost type and application rate on the measured variables were assessed by analysis of variance (at 5% probability level) and the relation between variables was established by regression analysis (at 5% probability level) using the statistical package GenStat (GenStat, 2006).

The accuracy of the statistical analysis was checked graphically by the residual plots. When the residuals did not follow the normal distribution then the data were normalized by applying a square root or a natural logarithm transformation. Also, in cases of an extreme value in the data set, this value was removed from the analysis when this data point caused the residuals to be not normal.

7.3 Results and discussion

7.3.1 Nitrogen mineralization

In general, trace amounts of NH_4-N were found during the incubation, indicating that the nitrification process was very rapid. Nitrogen mineralization is presented in terms of the sum of NH_4-N and TON-N.

The soil mineral N of the different treatments is presented as a function of the incubation time in Figure 7.2. Soil mineral N is shown to increase with the incubation time ($P<0.001$). Considering each sampling time, onion compost application resulted in significantly higher ($P<0.05$) amounts of soil mineral N than the control or the biowaste compost amended soil. Biowaste compost application resulted in similar ($P>0.05$) levels of soil mineral N to the control.

The effect of compost application rate on soil mineral N content was found not to be significant ($P>0.05$) for any given time, which is attributed to the fact that the increase in the amount of mineral N is not significantly different between the treatments. Considering, however, the mean mineral N over the whole duration of the incubation study, it was shown to increase significantly ($P<0.001$) with the increase of the onion compost application rate (Figure 7.3). Biowaste compost application at the highest rate significantly increased the mean soil mineral N content, in comparison to the control soil and the lower rates of biowaste compost application (Figure 7.3). The higher soil mineral N at the beginning of the study (day 0) resulted from the onion compost application is indicative of the higher compost mineral N content at the beginning of the study. This explains the higher soil mineral N levels at the higher rates of compost application, because, as it will be presented later, the effect of compost application rate did not significantly affect the compost N mineralization rates.

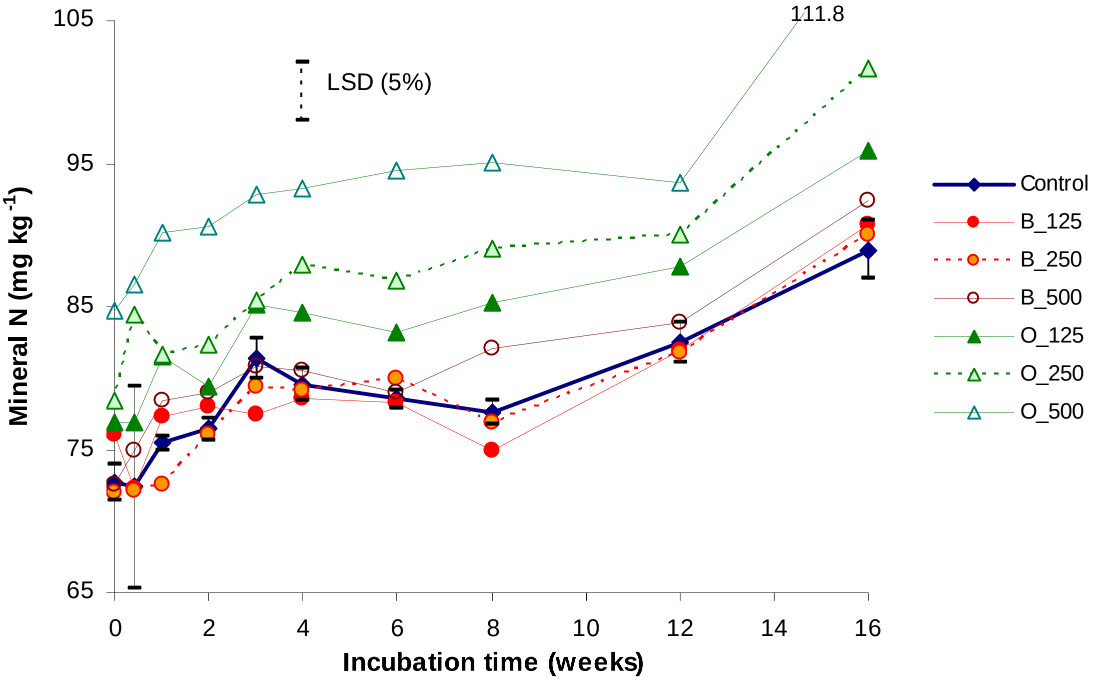

Figure 7.2 Mineral N content of the biowaste and onion compost amended soil and the control soil (B: Biowaste, O: Onion; 125, 250, 500: compost application rates in kg total N ha^{-1}). The dashed vertical bar shows the LSD (5%). The solid vertical bars represent the standard deviation of the control mineral N.

An important feature relating to Figure 7.2 is the N mineralization resulting from the control soil. Nitrogen mineralization from the control soil was found to increase significantly with the incubation time. This finding is supportive of the N mineralization results obtained for the control soil in the field experiment. Both the loamy sand soil used in the incubation trial and the sandy loam soil in the field experiment had similar OM, total N content and C: N ratio. A soil C: N ratio of 10.4 (field) or 13.2 (incubation) is indicative of N mineralization, as net mineralization occurs typically at soil C: N ratios below about 25:1 (Waksman & Tenney, 1927). Furthermore, a soil C: N ratio of

15 is suggested to be a critical limit separating soils in groups of higher or lower N release (Springob & Kirchmann, 2003), thus suggesting that high N mineralization is likely from both the loamy sand and sandy loam soils used in the incubation and field experiments, respectively, which was observed in both cases.

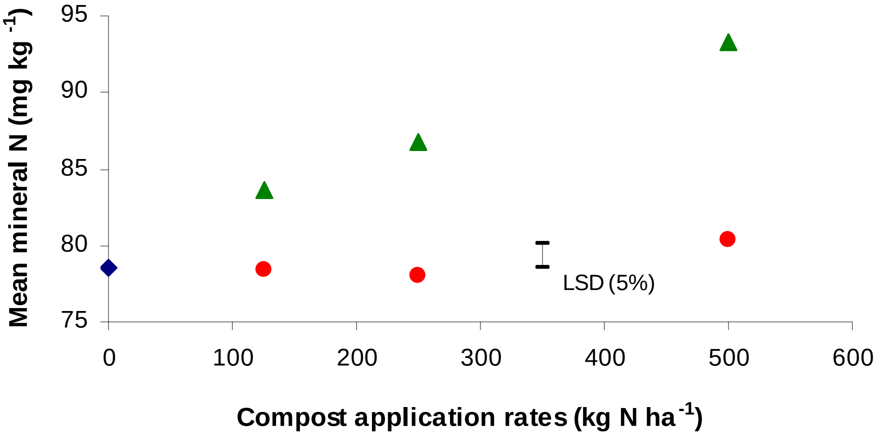

Figure 7.3 Mineral N content of the biowaste and onion compost amended soil and the control soil, calculated as the mean of the whole duration of the incubation study (◆: Control, ●: Biowaste, ▲: Onion).

As far as the mineralization rate of compost N is concerned, significant differences ($P<0.05$) in the NMR_{total} and NMR_{org} were found between the different compost types, whereas the effect of compost application rate was found not to be significant ($P>0.05$) in both cases (Figure 7.4 and Figure 7.5). Onion compost resulted in significantly higher ($P<0.05$) NMR_{total} than the biowaste compost during the whole incubation period. However, onion compost resulted in significantly higher ($P<0.05$) NMR_{org} than the biowaste compost only on week 8 and 16. The rest of the incubation period, the NMR_{org} of the onion compost was not significantly higher ($P>0.05$) than that of the biowaste compost.

The net N mineralization of the biowaste and onion compost during the incubation period (in % of total and organic N) is summarized in Table 7.4. The NMR_{total} of the onion compost ranged between 5.0% (week 3) and 11.1% (week 16), whereas the NMR_{total} of the biowaste compost between -3.0% (week 3) and 1.9% (week 16).

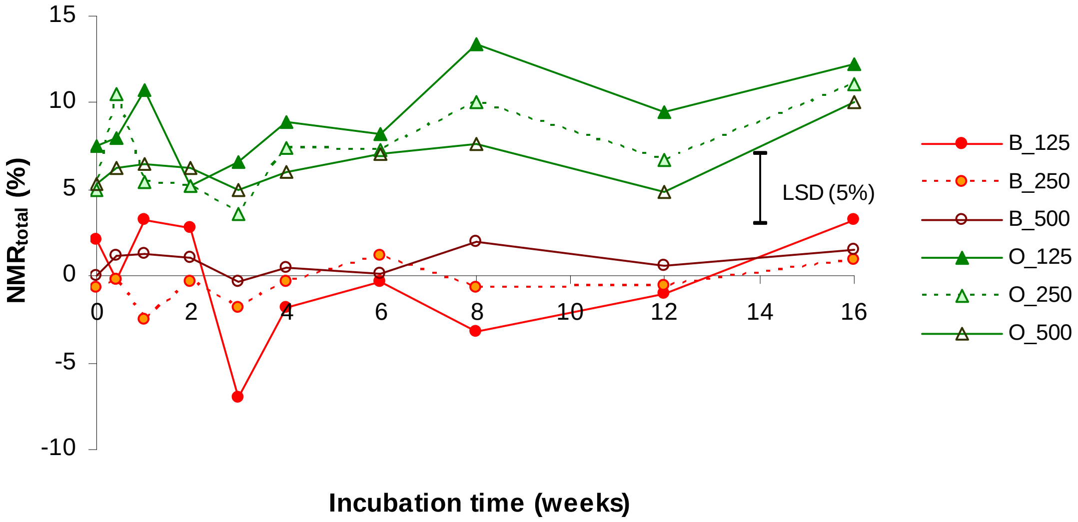

Figure 7.4 Percentage of total N mineralized (NMR_{total}) from biowaste and onion compost amended soil applied at different rates (B: Biowaste, O: Onion; 125, 250, 500: compost application rates in kg total N ha^{-1}).

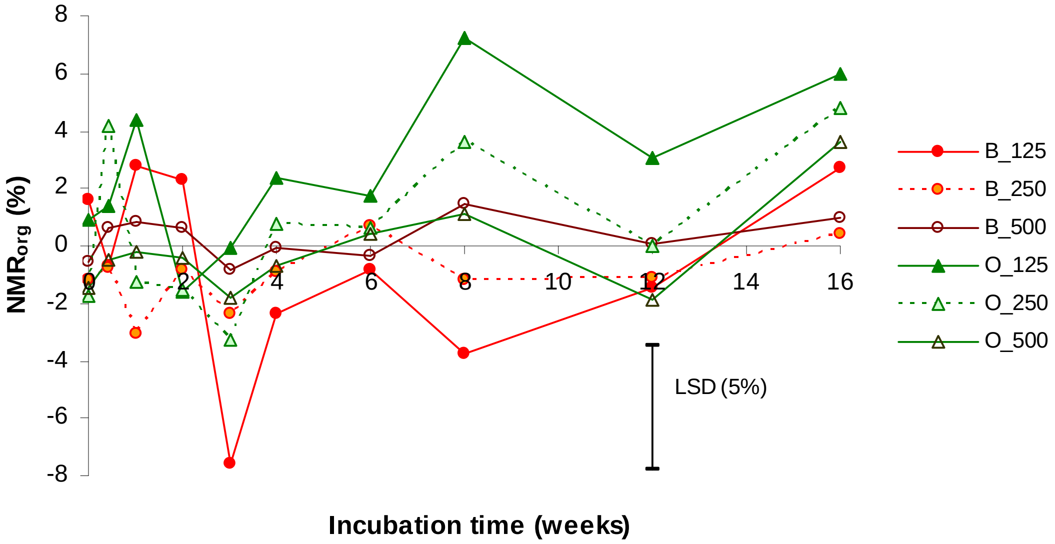

Figure 7.5 Percentage of organic N mineralized (NMR_{org}) from biowaste and onion compost amended soil applied at different rates (B: Biowaste, O: Onion; 125, 250, 500: compost application rates in kg total N ha^{-1}).

The NMR_{total} of the onion compost appears to decrease from week 2 to 3, then to increase till week 8, decrease in week 12 and finally increase (and peak) in week 16. The NMR_{total} of the biowaste compost remained fairly constant till week 2, then decreased in week 3, and increased in week 4 and from that point till week 12 was ranging close to zero. Biowaste compost mineralization of total N increased (and peaked) in week 16. Biowaste compost showed an immobilization period during weeks 3 to 12, and net N mineralization was at the beginning (weeks 0 to 2) and at the end of the incubation period (week 16). Onion compost resulted only in net N mineralization of the total N applied.

Table 7.4 Net mineralization of the biowaste and onion compost N during the incubation period (as a percentage of total and organic N). Means followed by different letters within each column are significantly different (P<0.05).

Time (weeks)	**NMR_{total} (%)**		**NMR_{org} (%)**	
	Biowaste	**Onion**	**Biowaste**	**Onion**
0	0.49 bc	5.93 ab	-0.02 bcd	-0.75 ab
0.4	0.23 bc	8.23 bc	-0.29 bcd	1.71 c
1	0.69 bc	7.55 b	0.18 cde	0.98 bc
2	1.20 bc	5.53 ab	0.68 de	-1.17 a
3	-3.04 a	5.04 a	-3.58 a	-1.70 a
4	-0.60 b	7.39 ab	-1.12 b	0.82 bc
6	0.37 bc	7.51 b	-0.15 bcd	0.95 bc
8	-0.63 b	10.37 cd	-1.15 b	4.00 d
12	-0.30 bc	7.01 ab	-0.82 bc	0.40 b
16	1.89 c	11.13 d	1.38 e	4.82 d

Considering the organic compost N, it was shown that the NMR_{org} of the onion compost ranged between -1.7% (week 3) and 4.8% (week 16), whereas the NMR_{org} of the biowaste compost between -3.6% (week 3) and 1.4% (week 16).

The NMR_{org} fluctuation with time follows similar pattern to the NMR_{total} for both compost types. Biowaste compost application resulted mainly in immobilization of organic N and net N mineralization was only during weeks 0 to 2, and at the end of the incubation period (week 16). Onion compost also resulted in immobilization of the organic N at the day of application (day 0) and also for the short period from week 2 to week 3, whereas during the rest incubation period net N mineralization of the organic N occurred. At the end of the incubation period both composts resulted in net N mineralization.

The N mineralization/immobilization curves of both composts did not level off with incubation time. This finding suggests that composts may be subjected to multiple phases of decomposition, as the compost constituents may differ in decomposition rates. Multiple-phase N mineralization was demonstrated by He et al. (2000), following compost application to a sandy soil under field conditions, and Rogers et al. (2001), following food processing waste application to sandy clay loam and sandy loam soils under incubation conditions. They concluded that N release was slower from composts with less readily available carbon sources.

7.3.2 Soil organic soluble nitrogen

The OSN of the different treatments is presented as a function of the incubation time in Figure 7.6. In general, biowaste or onion compost application at different rates resulted in similar ($P>0.05$) amounts of OSN with the control soil. Exception to this was only week 1, when compost application at 250 kg total N ha^{-1} significantly increased ($P<0.05$) the soil OSN compared to control. The above mentioned findings indicate that compost application did not significantly influence the OSN pool. Soil OSN was found to vary with time, even for the non-amended soil.

Comparing Figures 7.2 and 7.6, it can be seen that there was a trend to decrease OSN towards week 16, when soil mineral N levels were increasing. This finding may suggest that the OSN is likely to mineralize with time, rather than remain in soluble organic forms and leach as such. Another feature to note is that soil mineral N levels were higher than OSN levels during the whole duration of the experiment for both the control and compost amended soil, thus indicating that the dominant form of TSN was mineral N rather OSN. This result is in good agreement with the results obtained for the forms of TDN in the leachate in both the pot and lysimeter experiments.

Concentrations of OSN were found to range between 5.5-23.4% of the TSN pool. Other studies, however, have shown that in grassland soils the OSN pool can range between 20-90% of the TSN pool (Bhogal et al., 2000), or in arable soils between 40-50% TSN (Murphy et al., 2000). The soil used for the incubation experiment was sieved and air-dried and possibly this process resulted in increasing the mineralization of TSN (Sparling & Ross, 1988; Cabrera, 1993).

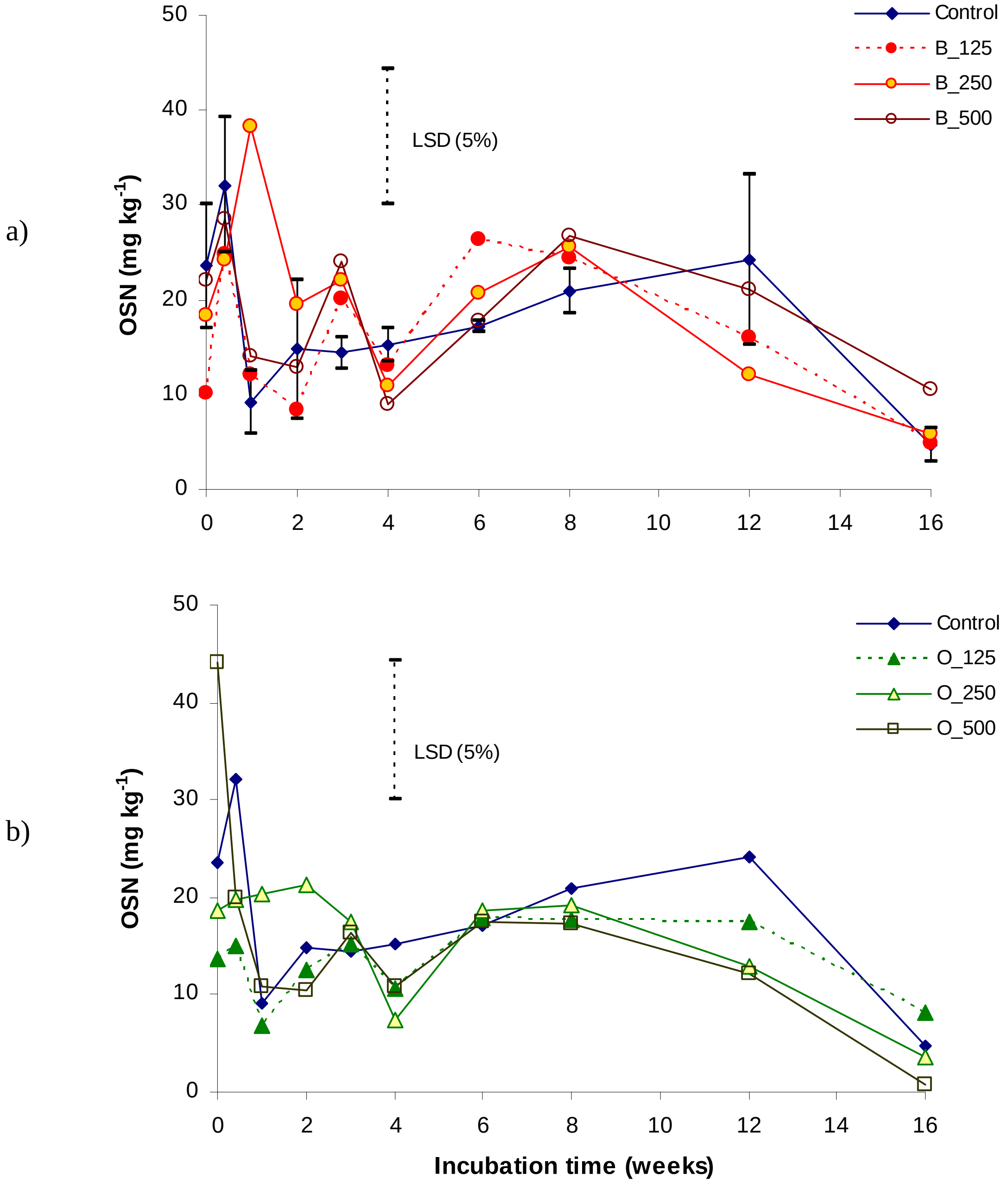

Figure 7.6 OSN content of the biowaste (a) and onion (b) compost amended soil, and the control soil (B: Biowaste, O: Onion; 125, 250, 500: compost application rates in kg total N ha^{-1}). The dashed vertical bar shows the LSD (5%). The solid vertical bars represent the standard deviation of the control OSN.

7.4 General discussion

The N mineralization results obtained from the incubation trial indicate that the net N mineralization resulting from compost application is generally low and it can range from -3% to 11% of the total N applied. Consequently compost application may result in both net mineralization and immobilization of N.

The composts used in the study were characterized by relatively low C: N ratios. The onion compost had a C: N of 9.3 and the biowaste of 15.3. Onion compost application resulted in significantly higher net N mineralization than the biowaste compost, thus suggesting that the lower the C: N ratio, the higher the N mineralization. Onion compost NMR_{total} was in the range of 5.0-11.1% during the incubation time, biowaste compost NMR_{total} was -3.0-1.9%.

Onion compost application increased the soil mineral N by both adding mineral N already existing in the compost and by mineralizing the organic compost N. The higher the application rate of the onion compost, the higher the soil mineral N levels. This was attributed to the higher amounts of compost mineral N added, as compost NMRs were found not to depend on the rate of compost application. The mineral N content of the biowaste compost was very low, therefore the amount of mineral N added to soil following compost application was due mainly to mineralization of organic N. Biowaste compost with higher C: N ratio had a significantly lower NMR_{total} and NMR_{org} than the onion compost.

At the end of the incubation both composts showed a net mineral N release. This finding is in good agreement with other incubation work using organic materials with low C: N ratios (Rogers et al., 2001; Chaves et al., 2004) and results of field studies which have shown that organic materials with C: N ratio between 9 and 15 prevent negative yield effects because of high N immobilization (Eriksen et al., 1999; Nevens & Reheul, 2003).

The monitoring of N mineralization with time showed a multiple-phase mineralization, possibly dependable on the compost carbon quality. This multiple-phase N mineralization may explain the slow release of compost N over time, which potentially results in decreased N losses and efficient N uptake by the crop.

The application of compost with C: N ratio of 9 to loamy sand soil was shown to result in an addition of mineral N of approximately 10% of the total N applied, within the first about 4 months following the application. The application of compost with C: N ratio of 15, resulted in an addition of mineral N of approximately 2% of the total N applied, although periods with N immobilization occurred. These mineralization rates of compost N indicate the amount of mineral N potentially supplied by the composts within a certain period. Such information is essential for proper recommendations regarding compost applications rates and timing, to match N availability and crop requirements better, whilst reducing the use of mineral fertilizers and the risk of water contamination.

Compost application generally did not affect the soil OSN pool during the incubation period. Murphy et al. (2000) concluded that significant amounts of DON can leach from agricultural soil but this amount is about 10% of the OSN extracted from the same soil. Following this conclusion, it was assumed that by monitoring the soil OSN fluctuations and how the OSN pool is affected by compost application, an estimate of the potential leaching of N in the form of DON could be obtained. As compost application generally did not increase the soil OSN during the incubation period, it was concluded therefore, that excessive leaching of DON following compost application is unlikely, at least the first 16 weeks after the application. This finding is in good agreement with the actual measurements of DON in the leachate collected from compost amended sandy and sandy loam soil (lysimeter and pot experiment, respectively).

The results of this study can be further used towards the development of a model to predict the N mineralization potential of composts based on certain compost characteristics (i.e. C: N ratio, C quality).

7.5 Conclusions

The following conclusions derived from the incubation study:

a) The results of the incubation trial are in good agreement with and support the results obtained from the field, pot and lysimeter experiments.

b) Compost N mineralization was found dependent on the C: N ratio and the degradability of carbon sources.

    - Soil application of onion compost with C: N ratio of 9 is likely to result in N mineralization of about 10% of the total N applied within the first about 4 months of compost application.
    - The application of biowaste compost with C: N ratio of 15, results in N mineralization of approximately 2% of the total N applied within the first about 4 months of compost application. However, periods with N immobilization are likely to occur.
    - Composts containing higher amounts of recalcitrant to decompose carbon substances are likely to result in lower N mineralization.
    - Onion compost is likely to result in higher N mineralization than the biowaste compost.
    - The rate of compost application was shown not to influence the N mineralization. However, the application of higher rates of composts with higher mineral N content results in higher soil mineral N content.

c) Composts may be subjected to multiple-phase N mineralization, thus resulting in slow N release over time and consequently in decreased risk of N losses.

d) The soil OSN pool is not likely to be significantly affected by compost application, at least the first 16 weeks following the application.

8 Economic evaluation

8.1 Introduction

Chapter 8 addresses the economic implications involved in compost utilization in agriculture, with particular regards to the agronomic perspective. An economic analysis of using compost in agriculture is required to study whether compost application could provide economic returns to the farmer, and thus whether farmers would be willing to use these materials on their land.

The analysis undertaken is based on the results of the field, pots and lysimeter experiments and therefore concerns short term compost application to agriculture. The objectives of the economic study were to identify the costs and benefits related to compost application in agriculture in the short term, from an agronomic perspective, and to discuss on the wider issues (e.g. environmental benefits) and economic implications of using compost in agriculture.

8.2 Methodology

8.2.1 Methods rational

The data used for the economic analysis was based on the results of the field, pots and lysimeter experiments. As discussed in previous chapters, the response to compost application varies with the compost type, rate and method of application, soil and crop used and the climatic conditions. For the economic analysis both types of compost and all rates of application were taken into consideration in order to show how the economics are affected by the different compost types and management practices.

Also, a simple cost-benefit analysis of using mineral fertilizers (MF) was undertaken for comparison. The analysis was based on the experiments, considering a single application of MF.

The lysimeter experiment (sandy soil, forage maize production) was undertaken for one growing period of forage maize. Because of the poor quality of the sandy soil used, it indicated clearly the potential effects of compost application on soil properties and crop production.

The field study (sandy loam soil, forage maize production) was undertaken for two years with the amount of compost application rate varying in each year. The soil in the field was fertile with relatively high organic matter content and the climatic conditions in both year of the experiment favoured forage maize production. Consequently, the crop response to compost and MF application was limited and not statistically different form the control in both years of the experiment. The effect of compost application was only obvious in the second year of the experiment. For the economic analysis it was assumed that in order to achieve the benefits shown in the second year of the experiment at least 2 years of compost application is necessary and so the amount of compost applied per year was calculated as the mean of both years.

The same sandy loam soil as the field study was also used in pots (sandy loam soil, forage maize and annual ryegrass production). Because forage maize response to the first year of compost application was limited, only the second period of the experiment (ryegrass) was analyzed, but again the amount of compost applied was taken as the mean of both years of application.

8.2.2 Calculations

8.2.2.1 Composts

The costs and benefits of compost use were calculated per ha and for the specific amounts of compost used in the three experiments as explained above. Application rate was based on compost total N content and therefore the amount of fresh compost applied was different for the two composts. Table 8.1 summarizes the amount of compost application in the different experiments in kg of fresh material per ha per year.

Table 8.1 Amounts of compost application in Mg fresh material per ha per year.

Application rates (kg total N ha^{-1} yr^{-1})	**Amount of fresh compost (Mg ha^{-1} yr^{-1})** Onion	Biowaste
Sandy soil, forage maize		
100	20.8	7.9
250	52.1	19.8
400	83.4	31.7
600	125.1	47.6
Sandy loam soil, forage maize		
90	20.4	8.7
208	46.0	19.4
325	72.9	30.7
Sandy loam soil, ryegrass		
90	24.5	8.3
208	55.8	19.0
325	87.0	29.6

The cost of using compost per ha (bought from a certified composting plant) was calculated as the total cost of buying the compost plus the haulage cost and spreading cost (Equation 8.1). In the case of using the compost produced and applied on the farm, the cost of using compost per ha was calculated as the total cost of producing the compost plus the spreading cost (Equation 8.2).

$$Total\ \cos t_{buy}\ (£\ ha^{-1}) = buy\ (£\ ha^{-1}) + haulage\ (£\ ha^{-1}) + spreading\ (£\ ha^{-1}) \qquad (8.1)$$

$$Total\ \cos t_{producing}\ (£\ ha^{-1}) = producing\ (£\ ha^{-1}) + spreading\ (£\ ha^{-1}) \qquad (8.2)$$

The benefit of using compost was calculated from the increase in yields, the mineral fertilizer N, P and K replacement costs, and the irrigation savings. The increase in yield was calculated in comparison to the control soil. The mineral fertilizer N replacement value (NFRV) was calculated based on the compost N recovery by each crop. P and K fertilizer replacement values were calculated based on the residual soil P and K Indices (Wallace, 2007). Irrigation savings were based on the increase of soil available water, as calculated from the moisture release curve produced for each soil type.

Other agronomic benefits of using compost include the increase in soil organic matter and consequently the improvement in structure and soil fertility. However, it is difficult to give a value to these benefits, especially in the short term, and therefore these long term benefits were excluded from the compost value determination.

The agronomic value of compost was calculated as the difference between the total benefit and the total cost of using compost. Environmental benefits resulting from using compost in agriculture were not taken into consideration for the determination of the agronomic value of compost.

8.2.2.2 Mineral fertilizer (MF)

The cost-benefit analysis of using MF was based on the amount of MF (N, P and K) required for optimum crop production in the three cases under consideration. Table 8.2 summarizes the required amount of MF per ha over the experimental periods.

Table 8.2 Amounts of mineral fertilizer N, P_2O_5 and K_2O applied per ha.

Conditions	**N**	**P_2O_5** (kg ha^{-1})	**K_2O**
Poor sandy soil, forage maize (1 yr)	120	110	230
Sandy loam, forage maize (2 yrs)	0	0	220
Sandy loam, forage maize & ryegrass (2 yrs)	180	0	200

It was considered a single application of MF undertaken whilst sowing (representing the MF application carried out during the experiments), and therefore the costs of applying the fertilizer were taken to be zero. Thus, the total costs of using the MF were considered to be equal to the costs of buying them.

In cases when MF is applied separately an application cost should be added to estimate the total cost. Fertilizer distributing costs (including loading and carting) depend on the amount of fertilizer to be applied per ha and the application method involved. The cost of broadcasting MF ranges from £ 6.2 ha^{-1} (125-375 kg ha^{-1}) to £ 14.75 ha^{-1} (1000-1250 kg ha^{-1}). Pneumatic distributing costs £ 14.00 ha^{-1} (125-375 kg ha^{-1}) (Nix, 2006).

The benefit of using MF was estimated from the increase in crop yield, in comparison to the control, according to the experimental results. The value of MF was calculated as the difference between the benefit and the total costs resulted from MF application.

8.3 Results

8.3.1 Costs of using compost

8.3.1.1 Cost of producing compost on farm

It was assumed that in the case where the compost is produced and applied on the farm, then the transport cost is zero. The cost of producing compost on the farm was determined according to the agricultural budgeting and costing book (ABC, 2007), taking into consideration that the production site was not requiring a EA licence. Consequently, the maximum amount of waste which could be composted is no more than 1000 m^3 at any one time (which will give about 500 m^3, or about 250 tonnes of compost). However, finished product or waste waiting to be composted does not count towards the total of 1000 m^3.

On farm sites, the machinery necessary for the composting process already exists, depending also on the method used. For static pile composting without forced aeration, as used for the production of the onion compost, the cost of producing the compost primarily consists of grading (screening prior to spreading) and shredding (the waste prior to composting). Because of the restriction of the amount of waste to be handled, it is more economic to hire the shredders and grader, at £ 10 and £ 2.5 Mg^{-1} compost, respectively (ABC, 2007), i.e. a total cost of £ 12.5 Mg^{-1} compost produced. Shredding, however, can be limited to the bulky materials rather the whole amount of waste to be composted. Assuming half of the waste needs shredding, this gives a shredding cost of £ 5 and a total cost of £ 7.5 Mg^{-1}.

Although only onion compost was produced on farm, the cost of producing both compost types was estimated in order to cover a wider range of cases. Biowaste compost cannot be produced on exempt sites on farm because of the kitchen waste included. However, it is shown in this research work that the composts are widely variable materials and therefore an on-farm produced compost (e.g. garden waste compost) could have characteristics similar to the biowaste compost and thus be applied at the same rates. Table 8.4 (columns 2 and 3) presents the potential cost of producing onion and “biowaste” composts on farm considering a total cost of £7.5 Mg^{-1}.

8.3.1.2 Cost of buying compost and haulage cost

The cost of buying compost was calculated by taking into consideration average typical costs of compost (£ Mg^{-1}, for certified compost according to BSI PAS 100). Typical cost range of compost of grade 0 - 10 mm is £ 0 to £ 5 Mg^{-1}, of 10 - 20 mm is £ 0 to £ 3 Mg^{-1}, and of 10 - 40 mm is £ 0 to £ 2 Mg^{-1} (considering agricultural use: large quantities of compost to be bought). A mean cost of buying compost of various grades ranging between 0 - 40 mm was assumed to be £ 2 Mg^{-1} (this price will be discussed later).

Typical costs of haulage in £ per Mg of compost are given in Table 8.3 (source: Nix, 2006). For composted biodegradable municipal waste with high nutrient content transport and application on farmland can be up to 20-40 km away (Crowe et al., 2002). A haulage cost of £ 4.77 Mg^{-1} compost was considered for transport up to 20 miles. The mean total cost to buy and transport the compost from the production site to the farm to be applied was taken as £6.77 Mg^{-1}. Table 8.4 (columns 4 and 5) presents the potential cost of buying and transport biowaste and “onion” (a compost produced in a certified plant with the same properties as the onion compost) composts considering a total cost of £6.77 Mg^{-1}.

Table 8.3 Haulage costs per Mg of compost in 2005 in the UK (source: Nix, 2006).

Miles	10	20	40	60	100	150
£ Mg^{-1}	4.29	4.77	5.74	6.71	8.65	11.08

8.3.1.3 Spreading cost

The cost of spreading was calculated considering that a FYM spreader was used. It was assumed that a spreader can hold 10 Mg of compost and needs about 3 min to load and 17 min for turnaround and spreading (Wallace, 2005). The cost of using a FYM tractor and loader is £ 26.50 per hour and the cost of using a FYM tractor and spreader is £ 30.00 per hour (Nix, 2006). Then the spreading cost of compost is £1.88 Mg^{-1}. Table 8.4 (columns 6 and 7) presents the potential cost of spreading the composts considering a total cost of £1.88 Mg^{-1}. It shows that the cost of spreading the compost is approximately four times less than producing or buying and transporting it to the field.

Table 8.4 Potential costs per year of producing, buying and haulage, and spreading compost for the different application rates used in the experiments.

Application rates (kg total N ha^{-1} yr^{-1}) (1)	**Costs (£ ha^{-1} yr^{-1})** **on-farm** **Onion** (2)	**on-farm** **"Biowaste"** (3)	**buy + haulage** **"Onion"** (4)	**buy + haulage** **Biowaste** (5)	**Spreading** **Onion** (6)	**Spreading** **Biowaste** (7)
Sandy soil, forage maize						
100	156.3	59.5	141.1	53.7	39.2	14.9
250	390.8	148.8	352.8	134.3	98.0	37.3
400	625.3	238.1	564.5	214.9	156.8	59.7
600	938.0	357.1	846.7	322.4	235.1	89.5
Sandy loam soil, forage maize						
90	153.2	65.2	138.3	58.8	38.4	16.3
208	344.8	145.5	311.3	131.3	86.4	36.5
325	546.7	230.0	493.5	207.7	137.0	57.7
Sandy loam soil, annual ryegrass						
90	184.0	62.0	166.1	56.0	46.1	15.6
208	418.3	142.2	377.6	128.4	104.9	35.7
325	652.4	222.2	588.9	200.6	163.5	55.7

8.3.1.4 Total cost of using compost

The total cost per ha per year of using composts produced on the farm site, where it is to be applied, (Equation 8.1) is shown in Table 8.5 (columns 2 and 3). The total cost of using composts produced on a certified plant (Equation 8.2) is shown in Table 8.5 (columns 4 and 5).

Table 8.5 Total cost per year of using compost, either produced on-farm or bought from a certified composting plant, for the different application rates used in the experiments, and the specific total cost of the onion and biowaste composts used in the experiments.

Application rates	**Total cost (£ ha^{-1} yr^{-1})**				**Specific total cost (£ ha^{-1})**	
	on-farm		**certified**		**on-farm**	**Certified**
(kg total N ha^{-1} yr^{-1})	**Onion**	**“Biowaste”**	**“Onion”**	**Biowaste**	**Onion**	**Biowaste**
(1)	(2)	(3)	(4)	(5)	(6)	(7)
Sandy soil, forage maize, 1 year of application						
100	195.5	74.4	180.3	68.7	196	69
250	488.8	186.1	450.8	171.6	489	172
400	782.1	297.8	721.2	274.6	782	275
600	1173.1	446.7	1081.8	411.9	1173	412
Sandy loam soil, forage maize, 2 years						
90	191.6	81.5	176.7	75.2	383	150
208	431.3	182.0	397.7	167.8	863	336
325	683.7	287.7	630.5	265.3	1367	531
Sandy loam soil, annual ryegrass, 2 years						
90	230.1	77.6	212.2	71.5	460	143
208	523.2	177.9	482.5	164.0	1046	328
325	816.0	277.9	752.5	256.3	1632	513

Columns 6 and 7 of Table 8.5 present the specific total cost of using onion and biowaste compost following the management system undertaken in this research project, taking in consideration the years of application. The specific total cost will be used later for the calculation of compost value.

Comparing the costs of producing the compost on the farm where it is to be applied with buying it from a certified plant, it can be seen that buying a compost can be cheaper, provided the transport distance between the farm and the composting plant is not further than 20 miles. Once the distance increases, then due the cost of haulage it saves money to the farmer to produce his own compost on the farm.

The cost of using onion compost (produced on-farm or on a certified composting facility) was higher than using biowaste compost. This was due to the fact that onion compost had lower total N content (fresh basis) than the biowaste compost, and consequently higher amount of onion compost was necessary for a given rate of total N ha^{-1}.

8.3.2 Benefits of compost application

8.3.2.1 Yield increase

The application of compost to forage maize production on a fertile sandy loam soil with high organic matter content did not influence crop yield in both years of compost application.

The application of compost to a fertile sandy loam soil with high organic matter content under irrigation conditions increased the yield of annual ryegrass in the second year of compost application, in comparison to control soil. The crop yield in the second year of compost application is attributed to the total amount of compost applied in both years/periods. Although the method of compost application was shown to have a significant effect on crop yield, the mean crop yield of both methods was considered.

The application of compost to forage maize production, on a poor sandy soil under irrigation conditions, increased crop yield in the first year of compost application.

The profit of using compost was calculated from the increase in crop yield in comparison to the control soil, assuming that the value of forage maize is £ 25 Mg^{-1} (at 30% DM) and of annual ryegrass is £ 30 Mg^{-1} (at 22.5% DM) (source: Nix, 2006).

Table 8.6 shows the profit (in £ ha^{-1}) of applying different rates of the two compost types in the three different cases mentioned above. It is shown that increasing the rate of compost and mineral fertilizer application results in increased profit in the case of the ryegrass production on the sandy loam soil. In the case of the forage maize production on the sandy loam soil, there was no profit made by any treatment, which was expected since there was no increase in yield.

In the case of the forage maize production on the sandy soil, increasing the rate of onion compost application resulted in increased profit. Biowaste compost application also increased profit, but the increase in the amount of biowaste applied did not increase the profit.

Table 8.6 Profit (£ ha^{-1}) resulted from increase in crop yield due to compost application to forage maize and ryegrass production.

Application rates (kg total N ha^{-1}) (1)	**Profit (£ ha^{-1})** **Onion** (2)	**Biowaste** (3)
Sandy soil, forage maize, 1 year		
100	69.8	26.3
250	203.9	38.2
400	284.2	1.4
600	448.4	22.1
Sandy loam soil, forage maize, 2 years		
180	0	0
415	0	0
650	0	0
Sandy loam soil, ryegrass, 2 years		
180	67.1	28.4
415	265.3	161.3
650	423.6	247.9

8.3.2.2 Fertilizer replacement value

Applying fertilizer N, P_2O_5 and K_2O involves two main costs: the cost of buying the fertilizer and the cost of applying it. In this study it was considered a single application of mineral fertilizers whilst sowing. Therefore the cost of applying the mineral fertilizer (MF) it is taken as zero, under the assumption that the fertilizer was applied while sowing.

Typical costs of buying MF N as ammonium nitrate, K_2O as muriate of potash and P_2O_5 as triple superphosphate are given in Table 8.7, assuming that fertilizers are delivered in 500 kg bags.

Table 8.7 Costs of mineral fertilizer N, P_2O_5 and K_2O (£ kg^{-1}) (source: Nix, 2006).

Fertilizer costs (£ kg^{-1})		
N	**P_2O_2**	**K_2O**
0.45	0.31	0.24

Table 8.8 presents the amount of MF N, P_2O_5 and K_2O applied to each treatment and the soil indices following the second year of compost application to sandy loam soil, or the first year of compost application to sandy soil. The effect of the MF N is assumed to last only during the first growing season following its application; therefore the amount of MF N presented in Table 8.8 corresponds only to the amount applied in 2006. The application of MF P_2O_5 and K_2O aimed to maintain soil P and K Index levels (MAFF, 2000), and hence the total amount of MF P_2O_5 and K_2O applied during the whole experimental period is presented for each treatment.

Table 8.8 Soil SNS, P and K Indices at the end of the growing season in 2006, as affected by the amount of mineral fertilizer N, P_2O_5 and K_2O applied to each treatment (for the estimation of the SNS Index, the potential mineralizable N was taken equal to zero).

Mineral fertilizer applied (kg ha^{-1})							**Soil Indices**								
MF treatments			**Compost treatments**				**MF**			**Onion**			**Biowaste**		
N	P_2O_5	K_2O	*Total N*	N	P_2O5	K_2O	SNS	P	K	SNS	P	K	SNS	P	K
Sandy soil, forage maize															
0	110	230	*100*	0	110	230	0	0	2-	0	1	2-	0	0	2-
40	110	230	*250*	0	110	230	0	0	2-	0	2	2+	0	0	2-
80	110	230	*400*	0	110	230	0	0	2-	0	2	2-	0	0	2-
120	110	230	*600*	0	110	230	0	0	2-	0	3	2+	0	1	2-
Sandy loam soil, forage maize															
0	0	110	*180*	0	0	0	6	6	3	6	6	3	6	6	3
40	0	110	*415*	0	0	0	6	6	3	6	6	4	6	6	3
80	0	220	*650*	0	0	0	6	6	3	6	6	4	6	6	4
Sandy loam soil, annual ryegrass															
0	0	90	*180*	0	0	0	1	5	2+	2	5	3	1	5	2+
180	0	90	*415*	0	0	0	2	5	2+	2	5	3	2	5	3
360	0	200	*650*	0	0	0	3	5	2+	3	5	3	2	5	3

In the case of sandy loam soil and both crops, it was shown that compost application can replace the addition of 220 or 200 kg ha^{-1} MF K_2O and even increase the soil content in extractable K, thus increasing the fertility of the soil. Two years of compost application to sandy loam soil with high P levels (P Index 5 or 6) did not increase the P index. As no addition of P was necessary for the crop production, the fertilizer P replacement value of the composts could not be calculated.

In the case of the poor sandy soil, K index was increased from 0 to 2 when mineral fertilizer K only, or fertilizer K and compost were added. P Index was increased with the combined application of onion compost and mineral fertilizer P or with biowaste compost at the highest rate combined with fertilizer P, indicating that with compost applications there will be savings in P fertilization in the following year. Because there were no treatments with no initial P and K mineral fertilizer, there is not enough evidence to suggest the direct effect of compost application on soil P and K. (Despite the fact that both P and K efficiency increased with compost application, which was also represented by the higher crop yield.) Consequently, the P and K fertilizer value of compost was taken equal to zero.

Two years of compost application to forage maize production on sandy loam soil with high SNS Index maintained the high potential nitrogen supply. As no addition of N was necessary for forage maize production in this soil, the fertilizer N replacement value of the composts cannot be calculated.

In the case of annual ryegrass production on sandy loam soil and forage maize on sandy soil the NFRV was calculated based on the N recovery values shown in Tables 5.3 and 6.3, respectively (Chapters 5 and 6). Table 8.9 presents the N, P and K fertilizer replacement value of the composts applied to forage maize and ryegrass production on sandy loam and sandy soil.

Table 8.9 Fertilizer N, P and K replacement value (FRV) (£ ha^{-1}) due to compost application to forage maize and ryegrass production on sandy loam and sandy soil.

Application rates	**FRV (£ ha^{-1})**							
	Onion				**Biowaste**			
(kg total N ha^{-1}) (1)	N (2)	P_2O_5 (3)	K_2O (4)	**Total** (5)	N (6)	P_2O_5 (7)	K_2O (8)	**Total** (9)
Sandy soil, forage maize								
100	2.5	0	0	2.5	0.8	0	0	0.8
250	8.3	0	0	8.3	2.0	0	0	2.0
400	11.1	0	0	11.1	0.8	0	0	0.8
600	17.8	0	0	17.8	1.5	0	0	1.5
Sandy loam soil, forage maize								
180	n.a.	n.a.	52.8	52.8	n.a.	n.a.	52.8	52.8
415	n.a.	n.a.	52.8	52.8	n.a.	n.a.	52.8	52.8
650	n.a.	n.a.	52.8	52.8	n.a.	n.a.	52.8	52.8
Sandy loam soil, annual ryegrass								
180	3.2	n.a.	48.0	51.2	0.9	n.a.	48.0	48.9
415	10.3	n.a.	48.0	58.3	6.5	n.a.	48.0	54.5
650	15.1	n.a.	48.0	63.1	8.7	n.a.	48.0	56.7

8.3.2.3 Irrigation savings

The water holding capacity of the sandy soil was found improved following one year of compost application. The water holding capacity of the sandy loam soil was not significantly affected by the two years of compost application.

The increase in the available water of the sandy soil due to compost application ranged between 5 to 61 mm, depending on the compost type and the rate of application. The irrigation savings of applying compost to sandy soil were calculated from the increase in the available water, assuming that this increase means the same amount of savings in irrigation water (mm). Water costs in the south-east Anglia region are typically considered to be about £ 75 per ha per 25 mm irrigation (Wallace, 07).

Table 8.10 Irrigation savings (£ ha^{-1}) resulted from compost application to sandy loam and sandy soil.

Application rates (kg total N ha^{-1})	**Irrigation savings (£ ha^{-1})** **Onion**	**Biowaste**
Sandy soil, forage maize		
100	24.1	14.3
250	46.8	38.8
400	115.3	58.4
600	183.5	87.9
Sandy loam soil, forage maize		
180	0	0
415	0	0
650	0	0
Sandy loam soil, annual ryegrass		
180	0	0
415	0	0
650	0	0

8.3.3 Compost agronomic value

The value of onion or biowaste compost was calculated by adding the profit resulting from the increase in yield (Table 8.5, columns 3 or 4), the mineral fertilizer replacement benefit (Table 8.9, columns 5 or 9) and the irrigation savings (Table 8.10, columns 2 or 3) and subtracting the costs of using the composts (Table 8.4, columns 6 or 7). The estimated compost value, therefore, represents the direct agronomic profit following compost application.

Table 8.11 presents the agronomic value of the biowaste and onion composts for the three cases under consideration. The values shown for the sandy loam soil are cumulative for two years, whereas for the sandy soil correspond to the first year of compost application. In the case of the sandy soil, a supplementary reduction in compost value of £ 90.4 ha^{-1} needs to be taken into account (for each compost type and rate used), as the calculated benefits also involved a fertilizer P and K application.

Table 8.11 shows that the application of both compost types resulted in negative values, which means that the costs of using composts were higher than the immediate agronomic benefits taken into consideration. The increase of compost application rates reduced the value per ha. The increase of compost application rates increases the costs of using the compost but also the benefits. Onion compost due to its lower nutrient content involved higher costs than the biowaste for a given rate, and therefore resulted in less profit per ha. The difference in the value of the two compost types decreases considering the value per ha per Mg.

Table 8.11 The agronomic value of applying different rates of onion and biowaste compost to forage maize and annual ryegrass production on sandy loam and sandy soil.

Application rates (kg total N ha^{-1})	**Fresh compost (Mg ha^{-1})**		**Value (£ ha^{-1})**		**Value (£ ha^{-1} Mg^{-1})**	
	Onion	**Biowaste**	**Onion**	**Biowaste**	**Onion**	**Biowaste**
Sandy soil, forage maize						
100	20.8	7.9	-99	-27	-4.76	-3.43
250	52.1	19.8	-230	-93	-4.41	-4.67
400	83.4	31.7	-371	-214	-4.45	-6.74
600	125.1	47.6	-523	-300	-4.18	-6.31
Sandy loam soil, forage maize						
180	40.8	17.4	-330	-98	-8.09	-5.61
415	92.0	38.8	-810	-283	-8.81	-7.29
650	145.8	61.3	-1315	-478	-9.02	-7.79
Sandy loam soil, annual ryegrass						
180	49.1	16.5	-342	-66	-6.97	-3.97
415	111.6	37.9	-723	-112	-6.48	-2.96
650	174.0	59.3	-1145	-208	-6.58	-3.51

8.3.4. Mineral fertilization

The cost-benefits analysis of applying the required amounts of mineral fertilizer N, P_2O_5 and K_2O for the production of forage maize and annual ryegrass, is summarized in Table 8.12. The amounts of the MF considered were shown in Table 8.2 and they represent the optimum amounts of N, P and K for the production of annual ryegrass and forage maize under the conditions of the experiments. According to these amounts the costs of buying the MF (presented in Table 8.7, assuming that fertilizers are delivered in 500 kg bags). The benefit of the MF application was based on the increase in crop yield.

The application of MF to forage maize production on a fertile sandy loam soil with high organic matter content did not influence crop yield in both years of compost application.

The application of mineral fertilizer to a fertile sandy loam soil with high organic matter content under irrigation conditions increased the yield of annual ryegrass, in comparison to control soil. The application of mineral fertilizer to forage maize production, on a poor sandy soil under irrigation conditions, only slightly increased crop yield.

Table 8.12 Costs, benefits and the value (£ ha^{-1}) of applying the required amounts of mineral fertilizer N, P_2O_5 and K_2O used in the three cases: forage maize production on sandy loam and sandy soil, ryegrass production on sandy loam soil.

	Cost	**Benefit**	**Value**
		(£ ha^{-1})	
Poor sandy soil, forage maize (1 yr)	144.4	0.7	-143.7
Sandy loam, forage maize (2 yrs)	52.8	0	-52.8
Sandy loam, forage maize & ryegrass (2yrs)	129.0	259.1	130.1

8.4 Discussion

The economic analysis demonstrated that compost application had a negative agronomic value in all cases under consideration, which means that the costs of using the composts were higher than the direct agronomic benefits (increase in crop yield, irrigation savings and replacement of MF) in the first one or two years of applying composts to light textured soils.

One feature to note, however, is that the sandy loam soil was capable of covering the N and P requirements of forage maize and also it was high in organic matter hence limiting the effects of compost application, which is a rare case in agricultural production. This was also suggested by the negative value obtained for the MF (Table 8.12). Thus, under typical light textured soil conditions compost application is expected to further increase crop yield and irrigation savings, hence increasing its value.

Table 8.11 shows that the agronomic value of the onion compost was approximately doubled when applied to the poor sandy soil. This was because of the increased agronomic benefits resulted from compost application. Also, if higher value crops are considered then the increase in yield is likely to further outweigh the costs.

Onion compost generally resulted in higher agronomic benefits than the biowaste compost; the costs of using the onion compost, however, were much higher and therefore the value of the onion compost (in £ ha^{-1}) was much lower than the biowaste. The costs of using onion compost were higher due to higher amounts of fresh compost necessary for the application of a given amount of total N, because of the lower N content per Mg of fresh material. This finding suggests that there is an economic advantage in using composts with high N content per fresh weight.

In the case that biowaste compost needed to be transported in distances longer than 20 miles, the value of the biowaste compost would be lower. Considering for example a haulage distance of 100 miles, then the value of biowaste compost, applied to ryegrass production on sandy soils, would be £ -6.84 ha^{-1} Mg^{-1}, whereas for 20 miles was £ -2.96 ha^{-1} Mg^{-1}. Haulage cost is an important factor controlling compost use in agriculture. Figure 8.1 shows the effect of haulage distance on biowaste compost value applied to ryegrass production on sandy loam soil.

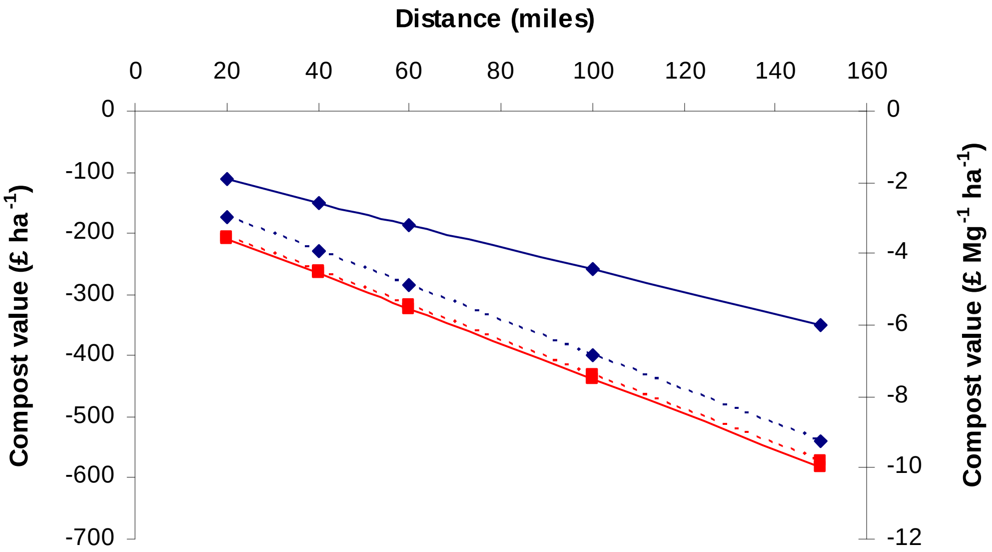

Figure 8.1 The effect of haulage distance on biowaste compost value applied to ryegrass production on sandy loam soil (🟥 : Biowaste compost applied at 59 Mg ha^{-1}, ◆ : Biowaste applied at 38 Mg ha^{-1}; the solid-line presents the compost value in £ ha^{-1}; the dashed line shows the compost value in £ Mg^{-1} ha^{-1})

An important feature to note, however, is that the value of biowaste compost applied to sandy loam soil at 650 kg total N ha^{-1} (59 Mg fresh material ha^{-1}) considering a haulage distance of 150 miles was higher than the onion compost applied at 415 kg total N ha^{-1} (112 Mg ha^{-1}) with no haulage cost involved. This finding demonstrates that the nutrient content of compost is more important than haulage distance in controlling the costs of using compost. Onion compost is advantageous over biowaste in value per Mg per ha only when the transport distance is 100 miles or more.

Using composts in agriculture generally resulted in less profit than using mineral fertilizer in the short term. However, in the extreme case of a poor sandy soil, using MF can cost as much as using composts in relatively low rates of application (up to 250 kg total N ha^{-1}). This is because high amounts of MF are necessary for crop production in such soils, but crop response is poor because of the low soil quality. These soils can be reclaimed with compost application mainly through the addition of OM.

Another important feature to note is that the prices of all fertilizers are increasing due to high costs of natural gases, decline of natural fertilizer reserves and continuous shouting down of fertilizing manufacturing companies. Table 8.13 shows the increase in fertilizer prices from September 2006 to February 2008 (source: Nix, 2006; Nix, 2007; FWI, 2008). Fertilizer prices have approximately doubled within the last two years, thus indicating that potentially in a few years time using mineral fertilizers is likely not to be more profitable than using compost. This increase in the fertilizer costs is in addition to the environmental impacts of depleting non-renewable resources (such as natural gas and earth deposits), requiring high energy, producing greenhouse gases and other waste substances, which result from the manufacturing of mineral fertilizers (Gedara, 2008).

Table 8.13 Fertilizer prices in the UK in 2006, 2007 and 2008.

Fertilizer	**Price (£ Mg^{-1})**		
	Sep 2006	Sep 2007	Feb 2008
N (34.5% N)	156	169	275
Urea granular (46% N)	160	195	320
Triple super phosphate (47% P_2O_5)	148	250	400
Muriate of potash (60% K_2O)	145	165	310

Other benefits of compost application to agricultural soil, such as the increase in soil organic matter content, the improvement in structure, the reduction of soil erosion or the liming effect of compost were not taken into account in this economic analysis. However, they need to be taken into consideration when deciding about using compost, especially within a longer term farm management scheme.

Other work on compost application to agriculture over longer periods of time has shown higher benefits of compost use. According to Wallace (2007) compost application at 30 Mg^{-1} ha^{-1} can result in benefit of £116 ha^{-1} yr^{-1}, in average over a 5 years application. Unpublished work by WRAP (2007) has shown that compost application at 34 Mg^{-1} ha^{-1} yr^{-1} can result in benefit of £ 204 ha^{-1} yr^{-1} as an average within the first six years of application to light soils. This value increases to £ 311 ha^{-1} yr^{-1} as an average within the following six years of application to light soils. Benefits from saving in liming, tillage and erosion decrease were also taken into account. The same work showed that the benefits to heavier soils are limited ranging from £ 78 and £ 89 ha^{-1} yr^{-1} as an average within the first six years of application for medium and heavy soils, respectively.

Improving soil quality is not only an economic benefit to farmers, but it also/primarily constitutes an important contribution to increase food production to meet the requirements of the rapidly rising world population.

In addition to the soil quality improvement, there are further environmental benefits that should also be considered for applying compost to agriculture. Compost application results in lower N leaching than using mineral fertilizer N because of the slow N release, as shown in this study. Furthermore, compost application to land increases the carbon sequestration to the soil and contributes towards the organic waste diversion from landfills.

The diversion of organic waste from landfills has also an economic value, as the amount of active waste, such is the organic waste, is charged for disposal in landfills (landfill tax). The landfill tax in 2007 in the UK was £ 24 Mg^{-1} active waste, but an increase of £ 8 Mg^{-1} yr^{-1} is applied from 2008/09 to at least 2010/11 (DEFRA, 2007). This means that the landfill tax would have doubled in 2010/11 reaching £ 48 Mg^{-1} organic waste disposed to landfills. Despite the fact that there is also a cost for disposing the waste to the composting facility, which may range from £ 22 to £ 30 Mg^{-1} green waste or £ 41 to £ 52 Mg^{-1} kitchen waste (letsrecycle, 2007), it can be seen that there is also potential economic benefit for diverting organic waste to composting.

Another fact that should be taken into consideration is that within the EU Common Agricultural Policy (CAP) reform 2003, the reinforced cross compliance includes respect of standards of good agricultural and environmental conditions, referring to soil protection from erosion, maintenance of soil organic matter and soil structure, and supporting the application of good quality compost (EC, 2003a). In the UK in particular, the application of composts is encouraged under the Single Payment Scheme (SPS) of cross compliance for the soil protection (DEFRA, 2006a). Since applying compost to the soil provides with the socio-environmental benefits mentioned above, it is on the interest of society to promote compost application to agricultural land, and consequently farmers could receive an environmental payment, in the form of e.g. free distribution and transport, in order to make it more appealing to farmers.

8.3 Conclusions

The economic analysis of the two years compost application to the production of forage maize and annual ryegrass on sandy and sandy loam soils led to the following conclusions:

a) The cost of using biowaste or vegetable waste compost in agriculture is relatively high, and it was found to outweigh the economic benefits from increased crop yield, mineral fertilizer replacement and irrigation savings, following a two year application to light textured fertile soils.
b) The cost of haulage is the most important cost associated with compost use in agriculture, when compost is bought from a centralized site. Collecting compost from a centralized facility is likely to cost less than producing the compost on the farm to be applied (exempt site) when the haulage distance between the farm

and the composting facility is less than 20 miles. The costs of shredding the waste for producing compost on-farm was the highest cost involved.

c) There is a significant economic advantage in using composts with higher N content per fresh weight because of the lower application costs involved.

d) The N fertilizer replacement value of the composts is low. The most important benefits of applying compost from an agronomic economic perspective are related to the increase in crop yield and the irrigation savings.

e) Compost agronomic value (in £ ha^{-1}) decreased with the increase of compost application rate, because the significant increase of the costs involved was not compensated with a similar increase in benefits.

f) Using compost was less profitable than mineral fertilization. However, using compost is a more sustainable practice due to the significantly increasing mineral fertilizer prices, the improvement of soil quality and the socio-environmental benefits of decreasing N leaching, reducing the production of greenhouse gases, decreasing the depleting of non-renewable resources, carbon sequestration to the soil and organic waste diversion from landfills.

9 Overall discussion

9.1 Introduction

This chapter integrates the experimental results of the four trials: field, pots, lysimeter and incubation, and provides further analysis on them. The aim of the analysis was to produce a holistic understanding of the effects of using biowaste and vegetable waste compost in agriculture.

The drivers for this chapter are the objectives of the research as stated in Chapter 1 (paragraph 1.3). Consequently, it focuses on the evaluation of the effect of compost type, application rate, and method on a) crop production, b) soil properties, and c) environmental risk from nitrate leaching. Also, Chapter 9 engages in the correlation of compost properties with crop and soil parameters. The economic implications of compost application to agricultural land were discussed in Chapter 8.

9.2 Effects on nitrogen dynamics

The investigation of the effects of applying biowaste and onion compost on N dynamics was of particular importance in order to meet the research objectives, as N is the main determinant for beneficial agronomic use and environmental risk from nitrate leaching. The N dynamics in the soil-compost-plant-water system are complicated (Amlinger et al., 2003; Gutser et al., 2005), and this was also demonstrated in the experimentation undertaken.

The focus of the study on N dynamics was on the mineralization of compost N and the fate of the N mineralized. Soil N and N losses through crop recovery and leaching were determined, as they constitute the major losses of N in agricultural soils in humid climates and under irrigated cropping systems (Tisdale et al., 1999). Gaseous losses and ammonium fixation were not considered. Figure 9.1 provides a diagram of the main routes of the compost mineralized N, along with the main factors affecting compost N mineralization and N availability, as were demonstrated in this research.

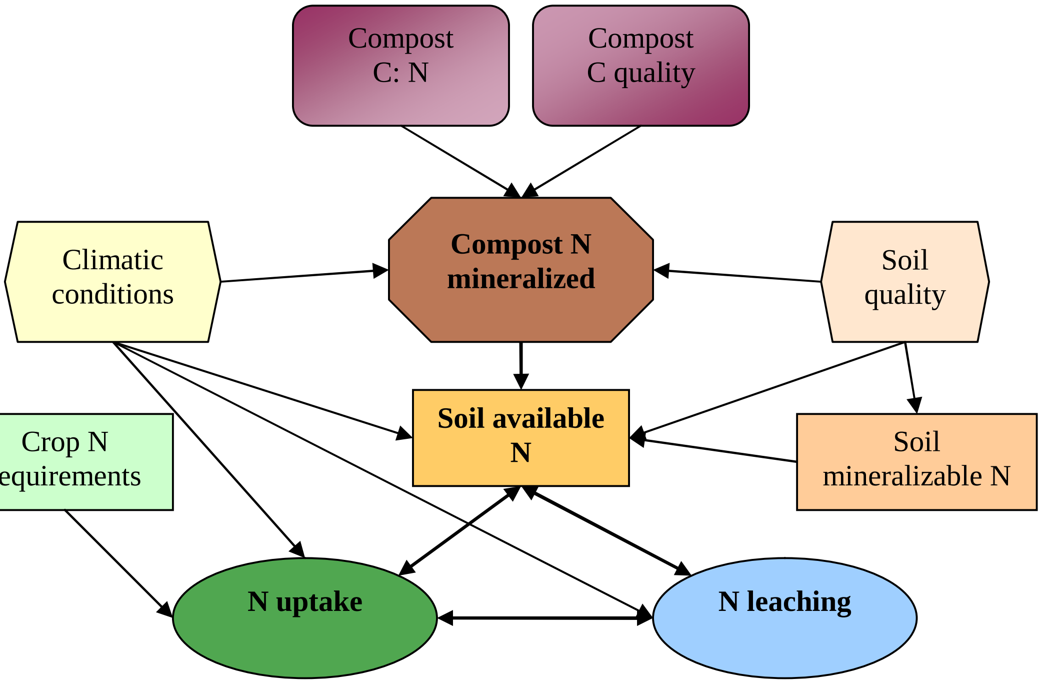

Figure 9.1 The main factors influencing compost N mineralization and the fate of compost N mineralized, as demonstrated in the research.

9.2.1 Compost nitrogen mineralization

The contribution of compost organic N to the N mineralized following the application of different types of composts with different C: N ratios to poor acidic, clay, and sandy loam soils, was shown in other work to range from negative to lower than 12% (Beloso, 1993; Gagnon et al., 1998; Gagnon and Simard, 1999; Mamo et al., 1999).

Compost C: N ratio is a commonly used indicator of the compost N mineralization potential (Sullivan et al., 2002; Wolkowski, 2003; Flavel & Murphy, 2006). Research on N mineralization has shown that there is a strong relationship between C: N ratios of various types of organic materials and the resulting N mineralization (Chadwick et al., 2000; Chaves et al., 2004).

The analysis of the data produced during this research project also suggested that there is a strong relationship between the compost C: N ratio and the compost N mineralized. The compost N mineralization rate (NMR) during the experiments was calculated a) as a percentage of the total compost N applied (NMR_{total} – Equation 9.1), and b) as a percentage of the organic compost N applied (NMR_{org} – Equation 9.2).

$$NMR_{total}\ (\%) = 100 \times \frac{Compost\ N\ Mineralized}{Compost\ Total\ N\ applied} \tag{9.1}$$

$$NMR_{org}\ (\%) = 100 \times \frac{Compost\ N\ Mineralized\ - Compost\ Mineral\ N\ applied}{Compost\ Organic\ N\ applied} \tag{9.2}$$

The mineral N content of compost is low and usually represents about 0 to 2% of the total N content (Beloso et al., 1993; He et al., 2000; Hadas et al., 2004). This range is in good agreement with the mineral N content measured for the biowaste and onion composts used in the experiments. However, it is important to provide farmers with reliable guidance on the supply from both the organic and the mineral N content, and therefore both the NMR_{total} and NMR_{org} were calculated.

The amount of compost N mineralized was taken to be equal to the sum of the amount of the net N recovered by the crop (when applicable), the amount of the net N leached (when applicable), and the amount of the net residual soil mineral N. The field experiment results were not taken into consideration as the extensive mineralization of the soil inherent N prevented the evaluation of the compost N availability.

In general, the compost NMR was found to be negatively related to the compost C: N ratio (NMR_{total}: $r = -0.767$, $P < 0.001$; NMR_{org}: $r = -0.673$, $P < 0.001$). Figure 9.2 presents the compost N mineralization in relation to compost C: N ratio.

The NMR (both organic and total) was best correlated with the compost C: N ratio. However, it was found that NMR was also significantly affected by the type of compost used, and the type of trial undertaken. The effect of compost application rate and method was not found to be significant (P>0.05) on both the NMR_{total} and NMR_{org}. Figure 9.3 presents the predicted compost N mineralization in relation to the C: N ratio of the biowaste and onion compost under the conditions of the different trials (NMR_{total}: r = -0.804, P <0.001; NMR_{org}: r = -0.733, P <0.001).

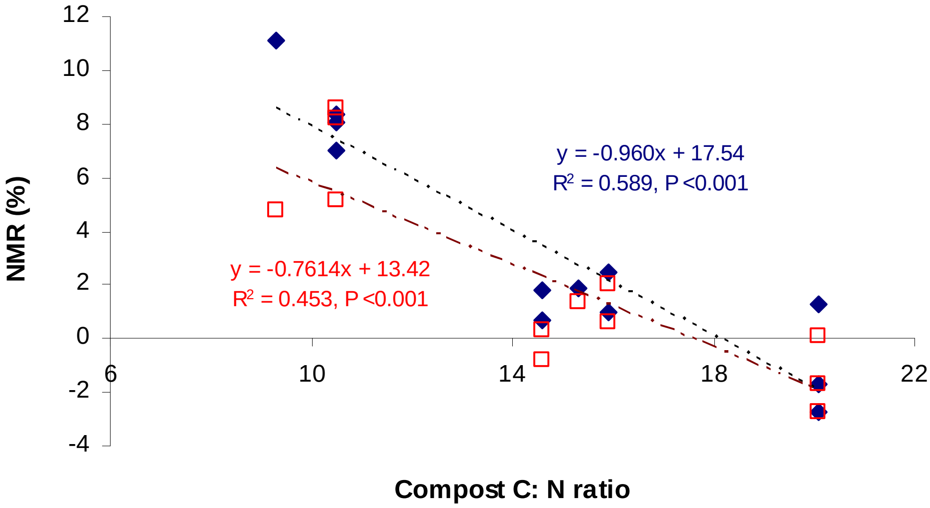

Figure 9.2 Compost N mineralization in relation to compost C: N ratio (◆: as a percentage of the total N applied, and ☐ : as a percentage of the organic N applied).

As it is shown in Figure 9.3, NMR_{total} was found to range from -3.7% to 1.1% for a compost with C: N ratio of 21 and from 5.1% to 9.9% for C: N of 9, depending on the compost type and environmental and soil conditions. NMR_{org} ranged from -3.9% to 0.8% for a compost with C: N ratio of 21 and from 3.2% to 7.9% for C: N of 9.

Studies on N mineralization from various organic materials (food waste and crop residues) have shown that the decomposability of the carbon sources of the materials also influences N mineralization. The more recalcitrant the carbon source to decomposition the slower the N mineralization (Janssen, 1996; Rogers et al. 2001). The relative concentrations of lignin and polyphenols are shown to be important carbon quality parameters that along with the N concentration of the material influence N mineralization (Janssen, 1996; Kumar & Goh, 2003; Chaves et al., 2004; Chaves et al., 2005a).

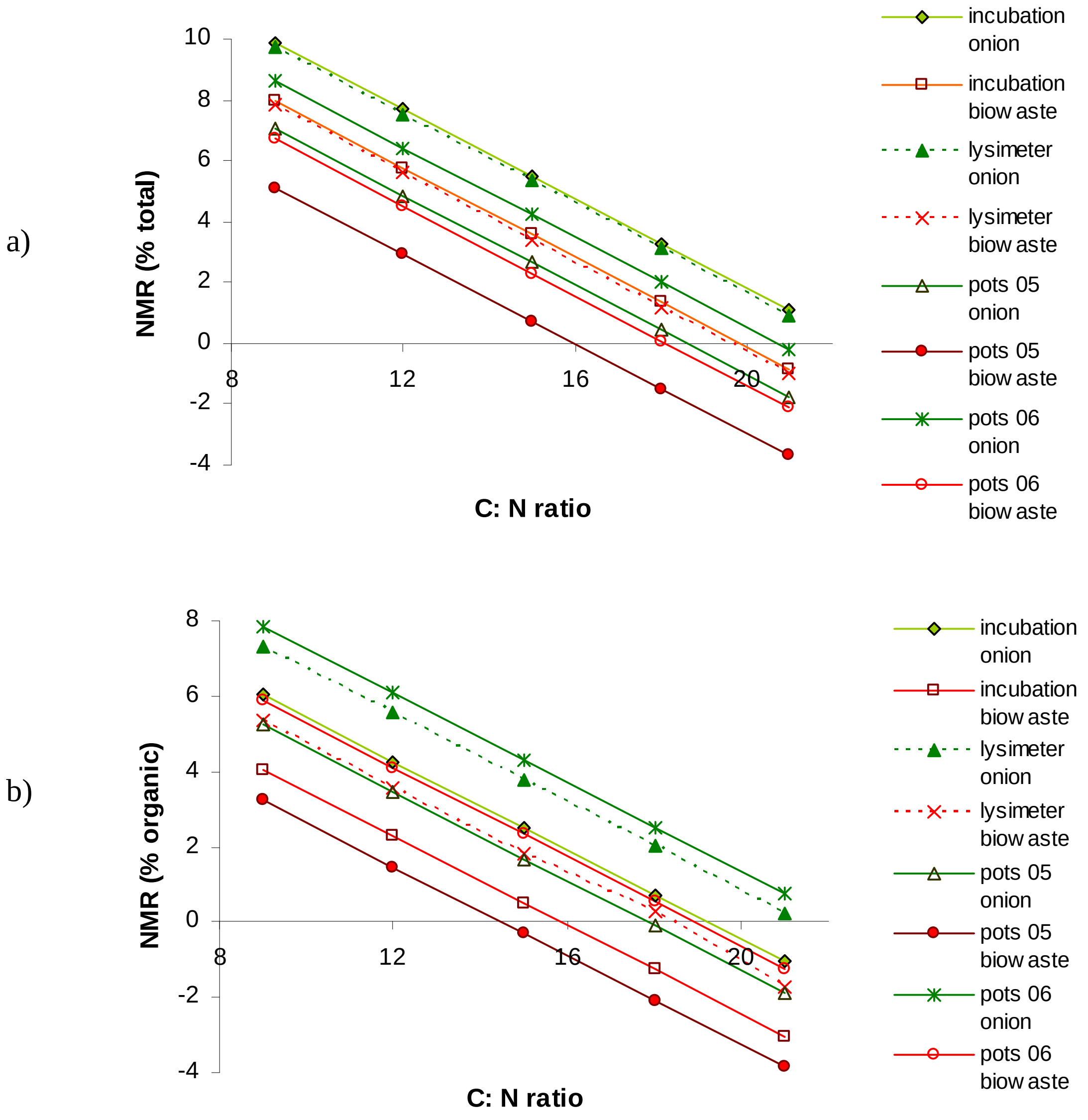

Figure 9.3 Compost N mineralization in relation to biowaste and onion compost C: N ratios, under different soil and climatic conditions (a) as a percentage of the total N applied, and (b) as a percentage of the organic N applied.

Tannins, which are polyphenolic compounds, may limit litter decomposition by: (1) themselves being resistant to decomposition, (2) binding proteins in protein-tannin complexes which are resistant to decomposition, (3) coating other compounds, such as cellulose, and protecting them from microbial attack, (4) direct toxicity to microorganisms involved in the process of N mineralization, such as bacteria, fungi and micro fauna, and (5) complexing or deactivating microbial exoenzymes (Kraus et al., 2003). The polyphenol effect on N availability therefore is not an immobilization of mineralized N, but an inhibition of N mineralization (Chaves et al., 2005a). Although the lignin or polyphenols content of the biowaste and onion compost used in the experiments were not determined (as biochemical compost properties were not considered), the results obtained from the experiments also suggest that N mineralization is dependent on carbon quality.

In particular, the biowaste compost contained garden waste (including significant amounts of wood waste), thus is likely to have higher amount of lignin, and/or other carbon sources that are more recalcitrant to decompose than the onion compost, which contained vegetable waste and straw. The biowaste compost used in 2005 had a lower C: N ratio than the onion compost in the same year (14.6 versus 15.9, respectively), but still N mineralization was significantly higher ($P<0.05$) from the onion than the biowaste compost, under the same soil and climatic conditions. Also, Figure 9.3 shows clearly that the onion compost has a higher N mineralization potential than the biowaste compost for a given C: N ratio, under the conditions of all the different trials. Thus suggesting that carbon quality, along with the C: N ratio, significantly influences the compost N mineralization.

Another important feature to note is that the results shown in Figure 9.3 indicate that compost with C: N ratio less than 15 is likely to result in net mineralization of organic N (rather than immobilization). This finding supports results from other work involving different types of organic materials (Jimenez & Alvarez, 1993; Chadwick et al., 2000; Gutser et al., 2005). Compost with C: N ratio higher than 15 is likely to result in limited N mineralization or immobilization of organic N, depending on compost C quality and soil and climatic conditions. The increase in compost C: N ratio favouring N immobilization. N immobilization may result because of inhibition of N mineralization due to the recalcitrance of compost organic compounds to decompose and/or because of the N mineralized made unavailable by the soil microorganisms.

Furthermore, the monitoring of N mineralization with time provided evidence of a multiple-phase mineralization following compost application to the soil, thus suggesting a non-steady N release, even if a compost is expected to result in net N mineralization

(C: N <15). He et al. (2000) and Rogers et al. (2001) also reported a multiple-phase N mineralization, which they concluded it was due to the different decomposability of the carbon sources. This finding is important because it implies a non-continuous or increasing N availability in the soil solution which would potentially increase the N leaching losses, but a slow release of N with time which would potentially increase the N use efficiency by the crop.

The significant effect of the trial type on the N mineralized is likely to be due to differences in the soil quality and the climatic conditions during the experiments. Janssen (1996) also reported differences in NMR between different experiments which were attributed to factors such as the soil pH and moisture. Rogers et al. (2001) reported that soil quality can affect compost N mineralization through its initial potential to better support the soil microbial community. According to Amlinger et al. (2003) the soil clay proportion is important as it can complex humus and result in decreased N mineralization. Soil moisture content is shown not to influence compost N mineralization (Benitez et al., 1998), with significant declines only under extreme wet or dry conditions (Amlinger et al., 2003). Chodak et al. (2001) showed that the increase of temperature increases N release.

9.2.2 Crop nitrogen recovery

Biowaste and onion compost application was found to influence forage maize and annual ryegrass N uptake. The amount of influence depended predominantly upon the compost C: N ratio, crop and/or soil conditions. In most cases the influence was positive.

Forage maize N uptake resulting from onion or biowaste compost amended sandy loam soil was not significantly different ($P>0.05$) from the non-amended soil, for both the field and pot experiments. This was attributed to the extensive mineralization of the soil inherent N, which was capable of meeting the forage maize N requirements, thus no further N addition was necessary for optimum crop production.

Onion compost (C: N = 10.5) amended sandy soil significantly increased ($P<0.05$) forage maize N uptake, when applied at rates equal or higher than 250 kg total N ha^{-1} (52 Mg^{-1} ha^{-1} of fresh material). The increase in application rate of the onion compost increased N uptake linearly, suggesting that there was an increase in plant-available N mineralized with increasing rates of onion compost. Biowaste compost (C: N = 20.1) amended sandy soil resulted in similar levels ($P>0.05$) of forage maize N uptake with the control soil. The low N recovery of the biowaste compost N was mainly attributed to the high compost C: N ratio, which resulted in limited net N mineralization/ immobilization.

Annual ryegrass N uptake was significantly increased ($P<0.05$) following the application of both onion (C: N = 10.5) and biowaste (C: N = 20.1) compost to sandy loam soil, at rates of 250 kg total N ha^{-1} (59 and 22 Mg^{-1} ha^{-1} fresh material of onion and biowaste compost, respectively) or higher. Annual ryegrass N uptake increased linearly with increasing compost application rate.

One feature to note is that biowaste compost (C: N = 20.1) increased annual ryegrass N uptake from the sandy loam soil, but not the forage maize N uptake from the sandy soil. Taking into consideration the N leaching losses that occurred (paragraph 9.2.3 – negative net N leaching from the biowaste compost amended sandy loam soil) it is concluded that the higher N uptake from the biowaste compost amended sandy loam soil was primarily a result of higher use efficiency of the soil and compost N by annual ryegrass, rather than a result of higher biowaste compost N mineralization. This finding is supported by the compost N mineralization potential presented for each trial (Figure 9.3); and it can be explained as a function of the crop, soil N availability and climatic conditions.

Table 9.1 presents the N recovery (as a percentage of the compost total N applied) as reported for each compost type, application rate and method, for the different crops and soils used. It was not possible to calculate forage maize recovery of compost N from the sandy loam soil, both in the field and pot experiments, as the N uptake from the compost amended soil was at similar levels with the control.

Table 9.1 Forage maize and annual ryegrass recovery (mean values) of biowaste and onion compost total N at the different rates and methods of application used.

Application rates (kg total N ha^{-1})	N recovery (% total N applied)					
	Sandy soil - forage maize		**Sandy loam - annual ryegrass**			
	Onion	Biowaste	Onion		Biowaste	
	Incorporated	Incorporated	Incorporated	Surface	Incorporated	Surface
100	5.6	1.8	4.9	8.3	0.6	3.3
250	7.3	1.8	5.3	12.0	6.5	4.3
400	6.2	0.4	5.2	10.7	4.5	4.7
600	6.6	0.5	-	-	-	-

As shown in Table 9.1 crop recovery of compost N in light textured soils may vary from 0.4% to 12%. This range is in good agreement with other research, where the compost N recovery by crops was found to range between 2% and 15% of the total compost N applied (Mamo et al., 1999; Nevens & Reheul, 2003; Wolkowski, 2003; Amlinger et al., 2003; Hartl & Erhart, 2005).

Table 9.1 also demonstrates that the application of onion compost resulted in significantly higher ($P<0.05$) N recovery than biowaste compost. The main reason for this is the lower C: N ratio of the onion compost, which results in higher mineralization of the compost N (as discussed in paragraph 9.2.1) and consequently in higher N availability for plant uptake.

The effect of compost application rate on N recovery was not significant ($P>0.05$). However, the application of biowaste compost to sandy soil at rates of 400 kg total N ha^{-1} or higher was observed to result in lower N uptake during the early stages of forage maize growth, than the non-amended soil. This finding is mainly attributed to early season immobilization of biowaste and soil N. Eriksen et al. (1999) also reported an early season immobilization of soil N following the application of about 310 kg total N ha^{-1} of MSW compost with a C: N ratio of 40. Rogers et al. (2001) found decreased NMR following the application of 224 kg total N ha^{-1} of food processing waste with a C:N of 31 in comparison to the NMR of half the rate. They suggested that the higher application rate may have overloaded the soil ecosystem, exceeding the maximum rate at which the soil microbial community could mineralize N, whilst possibly other factors, such as excess salts, were negatively affecting the soil microbes.

Onion compost surface application increased ($P<0.10$) ryegrass N recovery in comparison to incorporation. A possible explanation for this result is related to: a) faster mineralization of compost N when surface applied. Smith et al. (1997) suggested that surface applied organic materials tend not to be incorporated within the soil OM and consequently they are likely to mineralize faster, and/or b) better ryegrass establishment and development due to the higher soil moisture probably resulting from decreased evaporation losses with the surface applied compost.

The method of application of the biowaste compost, however, did not significantly affect the N recovery. This suggests that faster N mineralization is unlikely to occur following the surface application of composts with high C: N ratios (C: N >15), and/or the lower amount of biowaste compost surface applied (in terms of fresh material) resulted in less soil moisture retention than the onion compost.

The annual ryegrass recovery of the mineral fertilizer N (NH_4NO_3) applied to sandy loam soil was 7.8% at the rate of 180 kg N ha^{-1}, and 9.5% at 360 kg N ha^{-1}. On the sandy soil, the forage maize recovery of the mineral fertilizer N was 8.4% at the rate of 40 kg N ha^{-1}, 17% at 80 kg N ha^{-1}, and 5.7% at 120 kg N ha^{-1}. In both cases, the N recovery of the mineral fertilizer N applied is low and it indicates the high N losses occurred from the mineral fertilized soil.

Worldwide, fertilizer N recovery for cereal production is approximately 33% (Raun & Johnson, 1999). The low recovery of the NH_4NO_3 N applied to forage maize production in the lysimeter study is attributed to enhanced N leaching under the coarse sandy soil conditions (increased macropore flow and the low sand CEC resulted in increased nitrate leaching).

Perennial ryegrass recovery of NH_4NO_3 N is reported to range between 61-80% (Iglesias-Jimenez & Alvarez, 1993; Wilkins et al., 2000). Lower N recovery (34-49%)

has been documented by Cookson et al. (2000) under silt loam soil and irrigation adequate to keep soil between 70% and 90% FC. The low ryegrass recovery of the NH_4NO_3 N applied in the pot trial should be attributed to increased N losses by leaching, denitrification and/or NH_3 volatilization. Although measurements of gaseous losses of N did not take place, increased NH_3 loss from the NH_4NO_3 fertilized soil is likely to have occurred. This is because the fertilizer was applied at high rates on the soil surface, the soil was kept at or near field capacity, and slow drying conditions existed for several days thus increasing water evaporation from the soil surface and consequently encouraging NH_3 volatilization (Wild, 1988; Tisdale et al., 1999).

Comparing the crop N uptake between the compost amended and mineral fertilized soil, it can be seen that the range is similar. This indicates that under the conditions of this research (particularly referring to applying single rates of mineral fertilizer N under irrigation), mineral fertilizer N is not advantageous over composts for increasing crop N uptake and subsequently crop yield.

9.2.3 Nitrogen leaching

In general, compost application was shown not to increase N leaching losses on light textured soils, even under irrigation and high drainage rate (coarse sandy soil) conditions. This finding was related to the slow compost N mineralization, which leads to more efficient uptake by the crop. This was suggested by the fact that the increase of compost application rate did not increase N leaching, although it could increase N uptake.

There were cases, however, where the soil inherent N mineralized at rates that could meet the N requirements of the crop. In these cases, the application of compost was capable of increasing the N leaching, as the compost N mineralized was in excess of crop N requirements and thus susceptible to leaching.

In particular, compost application to forage maize and annual ryegrass production did not significantly increase ($P>0.05$) the amount of mineral N leached from sandy and sandy loam soil, respectively. However, in the case of forage maize production on the sandy loam soil the surface applied onion compost resulted in a significant increase ($P<0.05$) in N leaching. All the other treatments (onion incorporated and biowaste both incorporated and surface applied did not increase ($P>0.05$) N leaching compared to the control soil). Although direct measurements of N leaching were not carried out in the field experiment, the monitoring of soil mineral N levels of compost amended sandy loam soil provided evidence to suggest increased nitrate leaching following onion compost application to soils with high inherent N mineralization potential, capable of meeting crop N requirements.

Table 9.2 presents the amount of net mineral N leached from the biowaste and onion compost amended sandy or sandy loam soil as a percentage of the total N applied. Onion compost application generally resulted in higher mineral N leaching than the biowaste compost, corresponding to the higher N mineralization from the onion compost.

Table 9.2 Net mineral N leaching losses (mean values) resulted from the biowaste and onion compost application at different rates and methods on light soils under irrigation (as a percentage of the compost total N applied).

Application rates (kg total N ha^{-1})	**Net mineral N leached (% total N applied)**					
	Sandy soil		**Sandy loam soil**			
	Onion	Biowaste	Onion		Biowaste	
	Incorporated	Incorporated	Incorporated	Surface	Incorporated	Surface
2005			***forage maize***			
80	-	-	-0.5	3.0	1.6	1.0
165	-	-	0.3	1.8	0.9	0.5
250	-	-	0.7	2.4	0.0	0.2
2006	***forage maize***		***annual ryegrass***			
100	0.4	0.8	5.9	-7.5	-18.7	-20.5
250	1.5	0.1	3.2	-2.6	-8.4	-3.1
400	0.7	0.2	2.1	-2.3	-5.1	-4.0
600	0.4	0.0	-	-	-	-

The negative values shown in Table 9.2 mean that the N leaching from the compost amended soil was lower than the control soil. However, as shown in Table 9.1, N recovery was always positive. This finding indicates early season N immobilization due to biowaste compost application and increased N use efficiency of annual ryegrass from the compost amended soil, thus decreasing leaching losses in comparison to control.

The highest amount of net mineral N leached from the compost amended sandy soil was 3.8 kg ha^{-1}, corresponding to onion compost application at 250 kg total N ha^{-1}; this increase was found to be not significant ($P>0.05$) compared to the N leaching from the control soil (1.4 kg mineral N ha^{-1}). The highest amount of net mineral N leached from the compost amended sandy loam soil in 2006 was recorded to be 8.4 kg N ha^{-1}, corresponding to onion compost application by incorporation at 400 kg total N ha^{-1} to annual ryegrass (not significant increase ($P>0.05$) compared to the 67.5 kg mineral N ha^{-1} leached from the control soil). The maximum amount of net mineral N leached from the sandy loam soil in 2005 under forage maize production was 6 kg N ha^{-1}, when onion compost was surface applied (significant increase ($P<0.05$) compared to the 3.2 kg mineral N ha^{-1} leached from the control soil).

The amount of N leaching depends on the crop N uptake. In the case of the forage maize production on the sandy loam soil, the control soil was capable of covering the crop N requirements. Consequently, the compost N mineralized was in excess of the crop N requirements, and thus susceptible to leaching. In the cases when compost is not properly used (i.e. the compost N mineralized is in excess of the crop N requirements), the maximum potential of N leaching from the compost amended soil is not likely to be higher than 6% of the total compost N applied, as suggested by the results shown in Table 9.2.

The limited mineral N leaching results from the compost amended soil found in this work is in good agreement with findings of Hartl and Erhart (2005) who applied various rates of biowaste compost (C: N ranging between 9 and 49) over a 10 year field experiment on silty clay loam soil. They showed that on average of the 10 years the soil nitrate-N content (0-0.9 m depths) increased from 6 to 14 kg ha^{-1}, in comparison to the non-amended soil (65 kg NO_3-N ha^{-1}), corresponding to the application of about 81 to 205 kg total N ha^{-1} yr^{-1}. Nevens & Reheul (2003) found an increase of 3 kg N ha^{-1} in the residual soil NO_3-N, over a period of 4 yrs of vegetable, fruit and garden waste compost application (C: N of 10.2 at about 335 kg total N ha^{-1} yr^{-1}), in comparison to the control sandy loam soil (43 kg N ha^{-1}).

Work involving the application of high rates of compost N has shown higher potential for mineral N leaching following compost application. Eriksen et al. (1999), demonstrated, through the monitoring of soil NO_3-N within the 0-0.9 m depths, that only the highest rate (189 Mg ha^{-1}, corresponding to about 926 kg total N ha^{-1}) of a MSW compost (C: N of 40) resulted in significantly higher residual soil NO_3-N compared to the control sandy soil, the second year after application (increase of 13.6 kg NO_3-N ha^{-1}). Mamo et al. (1999) showed that the application of MSW compost, with C: N ratio ranging from 15 to 27, to loamy sand soil at high rates (about 900 kg total N ha^{-1} annually) increases the NO_3-N leaching 1.4 to 2.6 times higher than the non-amended soil over a 3 year period. Li et al. (1997) found that the amount of N leached from compost amended soil (at 100 Mg ha^{-1}) was between 3.3% and 15.8% of total compost N applied. However, they used different compost types (including biosolids and mixtures of biosolids with MSW) to sandy soil in leaching columns without a growing crop.

The application of 250 kg total N ha^{-1} per year or 500 kg total N ha^{-1} per two years constitutes the maximum amount of N that can be applied to the soil by organic materials according to the Water Code (MAFF, 1998b) for the protection of the water quality from excessive nitrate leaching. However, evidence from the short term application of biowaste and vegetable waste compost to light textured soils suggests that these limits can be questioned in the case of compost. Higher rates of compost application were shown to potentially increase crop yield without increasing N leaching, when properly managed (taking into consideration the crop N requirements, soil N availability and compost N mineralization potential).

The application of mineral fertilizer N (ammonium nitrate) to light textured soils was observed to significantly increase ($P<0.05$) the mineral N leaching in comparison to the control soil in all cases under consideration. The amount of net mineral N leached, calculated as a percentage of the mineral fertilizer N applied to forage maize production on sandy loam soil, was about 10% of the 30 kg N ha^{-1} applied. The mineral N leaching losses calculated as a percentage of the mineral fertilizer N applied (180 and 360 kg N ha^{-1}) was about 21% on the sandy loam soil under ryegrass production. Approximately 70% of the mineral fertilizer N (40, 80 and 120 kg N ha^{-1}) applied to forage maize production on sandy soil leached. The fertilizer N leaching losses occurred during the crop growing season, reducing the amount of mineral fertilizer N available for crop uptake, and thus restricting crop development, whilst deteriorating water quality. A feature to note, however, is that the observed excessive N leaching was enhanced by the fact that it was a single application of fertilizer N following the crop sowing.

As far as N leaching in organic forms from compost amended light textured soil is concerned, it was shown by measurements of the dissolved organic N in the leachate and the soil organic soluble N pool that it is low and not significantly different from the control or the mineral fertilized soil. Therefore, it should not be considered as an important source of N leaching from the compost amended soil at least in the short term.

To sum up, the above analysis demonstrates that N leaching depends on the interaction of factors such as compost N mineralization, plant N uptake, and soil parameters. There is evidence to suggest that properly managed compost application to light soils, is not likely to increase N leaching in comparison to non-amended soil, at least in the short term. Because of the slow release of compost N with time, compost application to light textured soils is likely to result in less N leaching than the single application of mineral fertilizer N, at least in the short term; thus compost application is advantageous over single mineral fertilization from a water quality perspective. Applying the compost based on its total N content while taking into consideration the compost C: N ratio, the crop N requirements and the soil N was shown to be an efficient management driver, in terms of sustainability, as it is likely to increase the agronomic benefit without resulting in excessive N leaching nor in the accumulation of other nutrients as it is often the case with P when applying sewage sludge or animal manure (Shober & Sims, 2003; Eghball et al., 2004).

9.2.4 Residual soil nitrogen

Soil residual mineral N (TON-N and NH_4-N), measured at the end of the trials, was generally low and not significantly (P>0.05) influenced by compost application. However, the residual soil organic N content was found to increase with compost application.

The highest increase (P>0.05) in the amount of residual TON-N (0-0.6 m depths) following compost application to forage maize production on sandy loam soil was approximately 14 kg TON-N ha^{-1}, resulting from the onion compost amended soil. The TON-N of the non-amended soil was 12.6 kg N ha^{-1}. The NH_4-N content was close to zero for all treatments. The two years of compost application to forage maize and ryegrass production on sandy loam soil (0-0.15 m depths) resulted in zero amounts of residual TON-N in all treatments. However, an increase (P<0.05) of 2.3 kg NH_4-N ha^{-1} was observed from the onion compost amended soil (control was 1.1 kg NH_4-N ha^{-1}). One year of compost application to the poor sandy soil resulted in similar TON-N content to the control soil (6.5 kg mineral N ha^{-1}, 0-0.65 m depths). NH_4-N content was zero for all treatments. These findings support the low N leaching potential of the compost amended soil, as discussed in paragraph 9.2.3. The residual mineral N from mineral fertilized soil was significantly higher (P<0.05) than the control soil in the case of the forage maize production on sandy loam soil.

The total N content (0-0.3 m depths) of the sandy loam soil in the field was not significantly affected (P>0.05) by compost application, but this could be explained by the high total N content of the field site at the beginning of the experiment. Hartl et al. (2003), who applied biowaste compost on a fertile soil (total N content of 2%) over a 5 year field experiment, observed limited statistically significant response to compost application. On the other hand, the sandy soil was very poor in total N (less than 0.03% which is the lowest detectable limit within the method used) and therefore the effect of compost application on soil total N levels could not be determined.

However, the residual total N content (0-0.15 m depths) of sandy loam soil in the pot experiment was found to be significantly increased (P<0.05) following the two years of compost application. As the residual soil mineral N content was very low, the increase of total N represented an increase in the soil organic N content. Mineral fertilizer N addition did not significantly increase (P>0.05) the soil organic N content.

Figure 9.4 illustrates the effect of compost application to the soil organic N levels. It can be seen that increasing the compost application rate results in significant increase (P<0.001) in the soil organic N content thus indicating the soil response to the amount of organic compost N applied. Soil storage of organic N in the compost treatments may explain why the low N recovery did not result in excessive N leaching during the experiments. Increased soil organic N (0-0.3m depths) resulting from biowaste compost application has been reported by Hartl & Erhart (2005) on a 10-year field experiment.

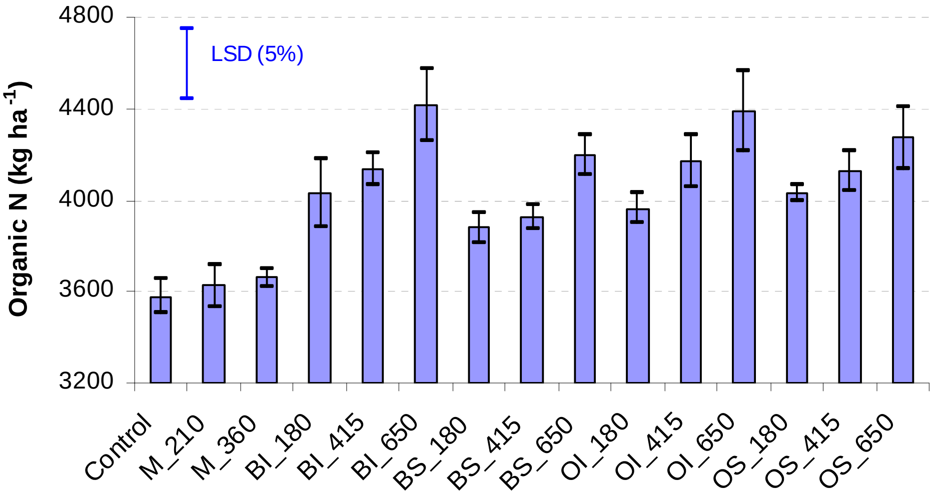

Figure 9.4 Soil residual organic N (mean values ± standard deviation) resulting from 2 years of compost and mineral fertilizer application to sandy loam soil (forage maize, year 1; annual ryegrass, year 2) (BI: biowaste incorporated, BS: biowaste surface applied, OI: onion incorporated, OS: onion surface applied, M: mineral fertilizer; 180-650: total amount of N applied within the 2 years in kg N ha^{-1}).

9.3 Soil properties (other than nitrogen)

9.3.1 Total carbon and organic matter content

The increase in organic N within the upper 0.15 m of compost amended sandy loam soil was complemented by significant increase in carbon, which indicates that the organic N was ‘tied up’ in the soil organic matter. The total C content of the sandy loam soil (0-0.15m depths) and the sandy soil (0-0.25 m depths) was found to increase linearly with increased rates of both onion and biowaste compost (Figure 9.5). However, the increase of the total C content of the sandy loam soil was not clear within the top 0.3 m under field conditions, indicating the slower increase of soil total C content of the lower layers of the soil profile following compost application.

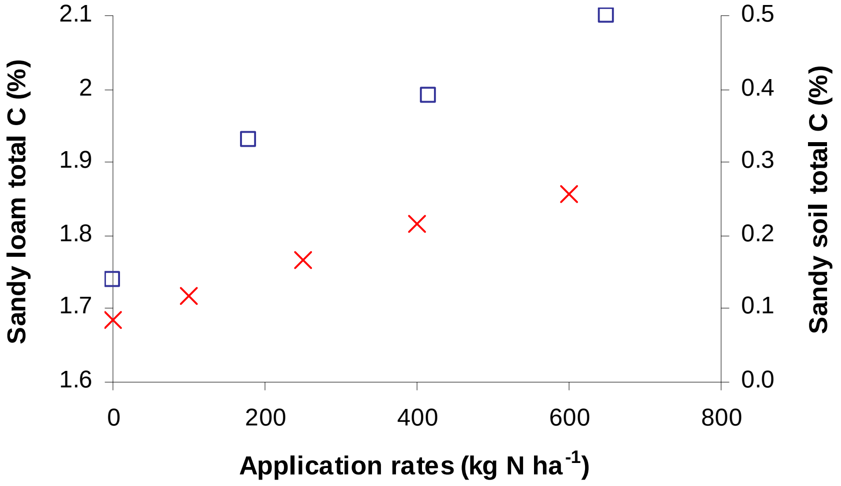

Figure 9.5 Soil residual total C content resulting from 2 years of compost application to sandy loam soil (0-0.15 m depths) – pot experiment (□), and 1 year of compost application to sandy soil (0-0.25 m depths) – lysimeter experiment (x). The application rates correspond to the total amount of compost total N applied within the experimental period.

There was a linear relationship between resulting soil total C content and compost total C applied, only in the case of the sandy soil. The effect of compost total C applied on the total C content of the sandy loam soil was not clear, despite the significant increase in soil total C content with increased rates of compost. This should be attributed to the addition of significant amounts of crop residues, which especially under the limited soil volume conditions of the pot experiment, increased the soil total C pool, thus the effect of compost C applied was not clear. The influence of the additions of crop residues to the soil total C pool may also explain why the effect of compost type on soil total C content was found not to be significant.

The effect of compost application on soil organic matter content was less obvious than the effect on soil total C content. The OM content of sandy loam and sandy soils (top 0.3 m or top 0.25 m, respectively) was not significantly influenced by compost application. However, the OM content (top 0.15 m) of the sandy loam soil was increased significantly following two years of compost application at rates higher than 90 kg total N ha^{-1} yr^{-1}. The increase in compost application rate significantly increased the soil OM, up to 12.4% compared to the non-amended soil, with the addition of an average rate of 325 kg total N ha^{-1} yr^{-1}. The effect of compost type on soil OM content was not significant. This is attributed to the fact that similar amounts of OM were added to the soil (in total within the two years of the study) following the application of biowaste and onion compost.

These results demonstrate the potential of increasing soil OM with compost application and that OM builds up slowly in the soil and therefore the effect of compost application is more obvious within the top 0.15 m of light textured soils. Hartl & Erhart (2005) found increased soil OM following 10 years of application of biowaste compost. Compost incorporation resulted in higher soil OM than surface application. This finding supports the faster mineralization of surface applied organic materials, as suggested by Smith et al. (1997). Also, it indicates that compost application by incorporation is likely to result in faster increase of soil organic matter content, and consequently sooner amelioration of soil quality.

9.3.2 Extractable potassium

Soil residual extractable K content was demonstrated to increase significantly following compost application, thus increasing K availability for crop production. The K content (top 0.15 m and top 0.3 m) of the sandy loam soil increased linearly with compost application, in response to the increase in the amount of compost soluble K applied (r =

0.893, P < 0.001). Figure 9.6 presents predicted values of soil residual extractable K (in 1 ha) resulting from two years of compost application to sandy loam soil.

For a given rate, onion compost resulted in significantly higher soil K content than the biowaste, because of its higher soluble K content. Biowaste and onion compost K content ranged from about 0.7% to1.1% in DM basis.

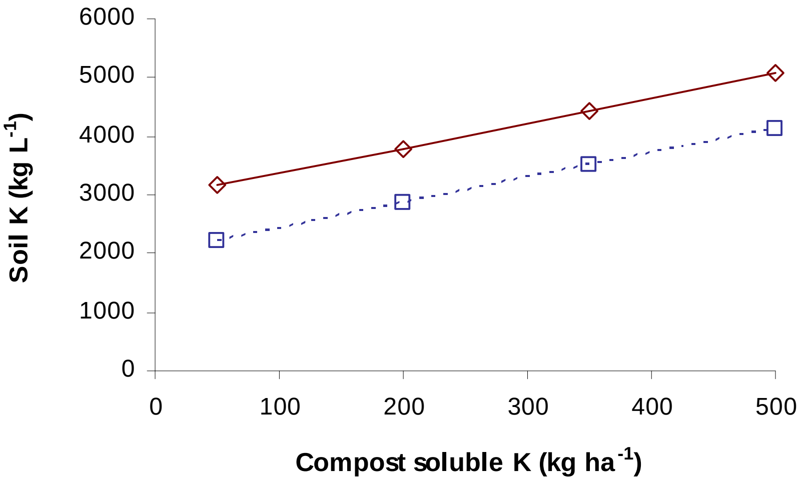

Figure 9.6 Predicted soil residual extractable K (in 1 ha) resulting from 2 years of compost application to sandy loam soil under field (◇) and pot (□) conditions. The highest rate of compost soluble K approximately corresponds to onion compost application at 325 kg total N ha^{-1} yr^{-1}, and the lowest to biowaste compost application at 90 kg total N ha^{-1} yr^{-1}.

The application of compost to sandy soil increased the soil K content by about 20%. The effect of rate was not significant, reflecting the differences in crop K uptake and/or suggesting potential K losses through leaching. Residual soil K increase following biowaste and green waste compost application has also been reported on silt loam and silty clay loam soils (Parkinson et al., 1999; Hartl et al., 2003).

9.3.3 Olsen phosphorus

Phosphorus availability was significantly influenced following compost application. The effect of compost application on soil available P was found to be dependent on soil texture and compost type.

The application of composts was observed to increase P solubility. Biowaste compost application to coarse sandy soils was found capable of resulting in the movement of available inorganic P downwards through the soil profile, below the root zone, thus resulting in limited P availability for plant uptake and suggesting potential risks from P leaching. Inorganic P is likely to be transported in association with organic colloids from compost, as explained by McGehan and Lewis (2002), especially on coarse textured sandy soils due to the increased macropore flow. This finding warns for potential reduced crop yields due to P deficiencies and increased water risk from eutrophication following compost application to coarse textured light soils.

The application of onion compost resulted in higher amounts of available P than the biowaste compost. This finding was attributed to both the higher soluble P applied with the onion compost and the lower capability of the onion compost to transport inorganic P (probably due to differences in compost OM quality/organic colloids between the two compost types). Further work is necessary in order to determine the effects of compost, and in particular compost OM, on P transport through the soil profile.

The increase of P solubility due to compost addition, however, is of advantage in alkaline and calcareous soils, as it is likely to prevent P fixation by Ca, thus increasing P availability to the crop. In these soils, the application of mineral fertilizer P did not increase P availability to the crop, as the inorganic P added was precipitated. The combined application of mineral fertilizer P and onion compost increased the residual sand available P content (0-0.25 m depths) by about 350%, whereas the addition of biowaste compost by 50%, compared to the only mineral fertilized sandy soil.

Compost application to sandy loam soil increased P availability. Onion compost resulted in significantly higher residual soil P than the biowaste compost. The application of onion compost significantly increased ($P<0.05$) the soil available P content (0-0.15 m depths) even at the lower rates of application. The biowaste compost resulted in a significant increase only when applied at the highest rate (on average 325 kg total N ha^{-1} yr^{-1}). The mean increase of available P content (0-0.15 m depths) of the sandy loam soil was 16% and 58% due to biowaste and onion compost application, respectively. Residual soil available P content (0-0.3 m depths) was not significantly affected by compost application. Other work has also shown no significant increase in soil P (0-0.3m depths) following the application of different types of compost (Madejón et al., 2001; Hartl et al., 2003).

9.3.4 Extractable sodium and electrical conductivity

Soil residual extractable Na content was found to increase with compost application. Under the field conditions, where no irrigation was applied, Na content (0-0.3 m depths) increased up to about 200%, in comparison to the non-amended soil, with a two year biowaste compost application by incorporation, at an average rate of 325 kg total N ha^{-1} yr^{-1}. There was a linear relationship between compost application rate and compost soluble Na, when the composts were incorporated (see Figures 4.19 & 4.20, Chapter 4). The surface application of composts resulted in significantly less soil Na than incorporation.

Under irrigation, soil Na content increased significantly following compost application by incorporation, by about 21% in the sandy soil (0-0.25 m depths) and 11% in the sandy loam soil (0-0.15 m depths), compared to the control soil.

However, in all cases, the soil ESP was very low (less than 1%) indicating that sodicity problems are not likely at these soil types, especially under irrigation. Care, however, needs to be taken when considering consecutive compost application to heavy soils (limited drainage), in Na sensitive areas (e.g. coastal areas), or in the production of sensitive crops.

The EC_e of the sandy soil under irrigation was not influenced by compost application, even at the rate of 600 kg total N ha^{-1}, suggesting that salinity problems are not likely following the application of composts to poor sandy soils under irrigation. However, the EC_e of the sandy loam soil was found to increase significantly with the application of the onion compost to non-irrigated soil. The two years application of onion compost at an average rate of 325 kg total N ha^{-1} yr^{-1} resulted in soil EC_e of 1.4 dS m^{-1}, while the control EC_e was 0.6 dS m^{-1}. An EC_e value of 1.4 dS m^{-1} is below the limit of 4 dS m^{-1} which is critical for most crops. However, this result may indicate potential salinity problems following year-after-year compost application at high rates.

Increased levels of soil EC due to compost application have been reported by Stamatiadis et al. (1999) and Madejón et al. (2001). They also concluded that although the increase of EC_e was not found capable of causing a sodium hazard to the soil, it indicated potential problems following the repeated application of compost to agricultural soil.

9.3.5 Soil water retention

Compost application was shown to increase the soil water retention of light soils. The available water content (v/v) of the sandy soil was found to increase from 4% (control) to 7% and 10% following biowaste and onion compost application, respectively, at 600 kg total N ha^{-1}. The one year application of composts to sandy soil significantly increased the soil volumetric water content when applied at rates equal or higher than 400 kg total N ha^{-1}. This increase was attributed to increased soil water retention capability close to FC. Soil water retention close to PWP was not enhanced.

Two years of compost application to sandy loam soil with high organic matter content did not result in significant increase of soil water retention, under field conditions. However, there was a trend for increasing the soil available water with compost application (from 15.4% (control) to 16.5% (compost amended soil) on a volumetric basis) indicating that long term compost application is likely to increase soil water retention, even on soils with relatively high OM content, thus improving crop growth and contributing to water conservation.

The method of compost application significantly influenced the water holding capacity of the sandy loam soil. Soil water content (0-0.15 m depths) appears to increase when composts are surface applied, as it was suggested by measurements of the gravimetric soil moisture content. This was attributed to the lower evaporation losses resulting from the surface application (mulching effect), and it was less obvious under the production of ryegrass. These findings are in good agreement with other work which has demonstrated that soil water holding capacity is likely to improve when composts are surface applied to row crops because of lower evaporation losses as demonstrated by Pinamonti (1998) and Agassi et al. (2004). Compost incorporation may increase soil water holding capacity mainly through the improvement of soil structure, as shown by Aggelides & Londra (2000).

9.4 Effects on crop production

Forage maize and annual ryegrass production on compost amended light textured soils was generally shown to increase mainly as a result of the amelioration of soil quality (i.e. improved fertility and water retention).

However, a two years application of biowaste and vegetable waste compost to a fertile sandy loam soil did not affect forage maize production. Crop DM yield from the compost amended soil were similar to the non-amended soil, as the soil itself was capable of supplying the necessary nutrients for crop production. Forage maize production was not affected by mineral fertilization either. The field was rich in organic matter, phosphorus and potassium and the total nitrogen content was high and mineralized at rates that were found capable of meeting the crop N requirements. The climatic conditions were positively influencing forage maize growth. The lack of forage maize response to compost application to the fertile sandy loam used in the study is in agreement with findings of Wolkowski (2003), who showed that maize yield is less responsive to compost application at sites with higher soil OM content.

Forage maize DM yield increased significantly following onion compost application to poor sandy soil. The application of mineral fertilizer N, P and K only, and the application of biowaste compost resulted in lower yields because of reasons related mainly to P and N unavailability. (The majority of the mineral fertilizer N leached during the crop growing season, whereas biowaste compost N was immobilized. Mineral fertilizer P was fixed by Ca, whereas biowaste compost resulted in downward transport of P. Thus in both cases N and P were unavailable for crop uptake). The application of onion compost at rates higher than 250 kg total N ha^{-1} was found to be the best practise to increase forage maize yield in poor sandy soils.

However, forage maize yield was much lower than the optimum of 40 Mg ha^{-1} fresh yield (MAFF, 2000) even on the onion compost amended soil (forage maize fresh yield was 22 Mg ha^{-1} following onion compost application at 600 kg total N ha^{-1}). This finding suggests that supplementary fertilization with mineral fertilizer N to the compost amended sandy soil during the period of maize highest N demand would increase crop yield, although optimum yields are unlikely under the poor soil conditions of the study. Low amounts of fertilizer N could be split-applied so as to avoid excessive nitrate leaching.

Annual ryegrass production on sandy loam soil was significantly enhanced by both onion and biowaste compost application at rates of 250 kg total N ha^{-1} (59 and 22 Mg^{-1} ha^{-1} fresh material of onion and biowaste compost, respectively) or higher. The increase in DM yield was at levels comparable to the mineral fertilized soil at 180 kg N ha^{-1}, although the highest yield resulted from an application of mineral fertilizer at 360 kg N ha^{-1}. Surface application of composts resulted in higher DM yield than incorporation, as a result of the higher soil moisture retention which was critical because of the late establishment of the crop. Onion compost surface applied at 400 kg total N ha^{-1} resulted in similar yield to mineral fertilization with 360 kg N ha^{-1}. Douglas et al. (2003) reported that the application of vegetable waste compost at 250 kg total N ha^{-1} yr^{-1} resulted in similar annual ryegrass DM yield with the application of ammonium nitrate at 250 kg N ha^{-1} yr^{-1}.

The ryegrass yield obtained from the mineral fertilized soil is in good agreement with other studies (Whitehead, 1995; Wilkins, 2000). However, the crop N concentration was low in all treatments (mean N concentration ranged from 10 to 11.6 mg N kg^{-1} DM, whilst the minimum level of N required by productive grazing animals is 20 g N kg^{-1} DM (ARC, 1980)). This finding, therefore, suggests that both mineral fertilizer and compost application did not meet the crop N requirements for optimum production, although differentiation in crop production is likely under field and pot studies.

9.5 Management considerations

Compost type was shown to be an important factor affecting both crop production and soil properties. Onion compost was advantageous over biowaste compost in increasing soil available N, P, K and water content for crop production. Neither of the two compost types under consideration resulted in increased nitrate leaching in the short term.

In general, the application of a biowaste or vegetable waste compost with C: N ratio lower than 15, is likely to result in net mineralization of compost N. The maximum amount of compost N mineralized was shown to be approximately 10% of the total compost N applied. Using a compost with C: N ratio higher than 15 may result in N immobilization thus potentially negatively affecting crop production. There is an advantage in using compost with C: N ratio lower than 15, as it can enrich both the soil available N for crop production, but also the soil organic N and C pool. This is because the rate of mineralization is relatively low hence improving the shorter- and longer-term soil fertility.

Applying the compost based on its total N content whilst taking into consideration the compost C: N ratio (to predict compost N mineralization), the crop N requirements and the soil N was shown to be an efficient management driver, in terms of sustainability, as it is likely to increase the agronomic benefit without resulting in increased N leaching. However, this management system may result in the application of high rates of compost, especially in N poor soils, under the production of crops with high N requirements.

Using high rates of compost may not result in increased nitrate leaching when they comply with the above mentioned management strategy, but they may introduce risks of salinity and increase soil Na, especially on a year after year application and in certain soil types, areas and climate (e.g. heavy soils, coastal, or low rainfall and non-irrigated areas). Crop sensitivity to salinity should also be considered. Using lower rates of application (not higher than 250 kg total N ha^{-1} yr^{-1}) supplemented by the application of mineral fertilizer N might be, therefore, the best management option for medium or heavy soils, under no or limited irrigation. On the other hand, increased rates of application are likely to result in increased organic matter content and water retention, hence improving soil quality and water conservation, which is of great importance especially in arid areas and for light textured soils. Therefore, parameters as the climatic conditions of the area and the soil type need also to be taken into consideration for calculating the optimum rate of compost application.

As far as the method of application is concerned, surface applied biowaste or vegetable waste compost is shown to result in higher soil moisture, under enhanced evaporation conditions, and in lower soil Na content. However, compost surface application may enhance the mineralization of the OM applied, NH_3 volatilization, denitrification and potential odour problems may rise (e.g. with composts with low DM content). Consequently, soil, climatic and compost parameters need also to be taken into consideration for determining the optimum method for compost application.

This study has shown that applying the compost while you prepare the seed bed for summer crops, or whilst sowing, can increase crop yield without increasing N leaching. Also, it can meet the P and K requirements of forage maize and annual ryegrass production. Supplementary fertilization with mineral fertilizer N later in the crop growing season might be necessary for optimum crop production.

The economic implications of applying biowaste and vegetable waste compost to agriculture are greatly influenced by the compost N content of the fresh material and the haulage distance. In the short term using compost is likely to be less profitable than mineral fertilization. However, using compost is a more sustainable practise due to the significantly increasing mineral fertilizer prices, the improvement of soil quality and the socio-environmental benefits of decreasing N leaching, reducing the production of greenhouse gases, decreasing the depleting of non-renewable resources, carbon sequestration to the soil and organic waste diversion from landfills.

10 Conclusions and recommendations

This thesis aimed to contribute towards the identification of a system for managing the utilization of vegetable waste and biowaste compost for sustainable agriculture, with particular regards to nitrogen availability and leaching.

The experimental research undertaken included controlled laboratory, lysimeter, pot and field studies. It focused on the characterization of the biowaste (source-separated biodegradable MSW) and vegetable waste (agricultural plant-tissue waste) compost, the correlation of compost properties with soil and crop parameters, the evaluation of compost N mineralization and N leaching, the changes in soil quality and the crop production following the short term compost application to light soils. The economic implications involved were also considered.

The conclusions and the areas for further research drawn from the experimental results and analysis are presented below. The detailed conclusions of each experiment and analysis can be found in Chapters 3 to 8.

10.1 Conclusions

This comprehensive study, in lab, lysimeter, pot and field conditions, allowed for the comparison of biowaste and vegetable waste composts, in agronomic, environmental and economic terms. These two compost types are of agricultural and municipal interest and they have been recently considered for agricultural use in the UK, following the implementation of the Landfill Directive. The research work undertaken added to the current knowledge regarding the availability of nutrients following compost amendment. It demonstrated that properly managed biowaste and vegetable waste composts, whilst relatively low in nutrient status, can be utilized in agriculture leading to both agronomic and environmental benefits. Following a two year application, these benefits can be mainly summarized as improvement of soil quality, and consequently improved crop production, diversion of organic waste from landfills and reduced impacts from nitrate leaching.

In particular, the compost analysis and the short term application of biowaste and vegetable waste compost to agriculture have shown that:

1. Biowaste and vegetable waste composts are variable materials.
    a. Compost chemical and physical properties vary significantly between compost types and years of production. In order therefore to properly manage compost application, agricultural practise should adjust to the specific characteristics of each material.
    b. With regards to the compost properties examined over the two years:
        i. The heavy metal content of both composts was within the limits established by the BSI PAS 100: 2005, so therefore there was no restriction, concerning heavy metals, for the management scenario for application to land.
        ii. The nutrient content of both composts was low suggesting a low fertilizer value of both types. On average (DM basis) the onion compost contained: 1.1% total N, 0.03% soluble mineral N, 0.9% soluble K, 0.03% soluble P and 0.03% soluble Na, whereas the biowaste: 1.8% total N, 0.02% soluble mineral N, 0.8% soluble K, 0.03% soluble P and 0.13% soluble Na.
        iii. The organic matter content, however, was high: 58% and 29% DM for the biowaste and onion compost, respectively, indicating potential increase in the soil OM content following compost land application.
        iv. The DM content was 59% and 42% for the biowaste and onion compost, respectively. Because of the lower DM content of the onion compost, in combination with the lower N content, larger amount of onion compost was necessary in order to apply a given quantity of N, thus increasing the costs of using the material in comparison to biowaste.
2. Compost N mineralization depends significantly on the compost C: N ratio.

   a. Biowaste and vegetable waste composts with C: N ratio lower than 15 are likely to result in net mineralization of compost organic N, thus increasing soil available N and excluding negative effects on crop production from N immobilization.
   b. N mineralization following the application of biowaste and vegetable waste composts with C: N ratios varying between 9 and 21, ranged from -3.7% to 9.9% of the total N applied, or from -3.9% to 7.9% of the organic N applied, depending mainly on the compost C quality and climatic and soil conditions.
   c. N mineralization was higher from vegetable waste compost than biowaste compost, mainly because of the lower C: N ratio.
   d. Compost N mineralization was not significantly influenced by the rate and method of compost application.

3. Forage maize and annual ryegrass N recovery from biowaste and vegetable waste compost amended light textured soils varied from 0% to 12% of the compost total N applied.
   a. N uptake from the onion compost amended soil was higher than from the biowaste compost amended one, corresponding to the higher N mineralization from the onion compost, thus resulting in better quality of the forage crop.
   b. The application of high rates (≥400 kg total N ha^{-1}) of biowaste compost with C: N of 20 to poor sandy soil was shown to decrease N uptake at the early stages of forage maize growth, in comparison to the non-amended soil, due to early season immobilization of compost and soil N. Although forage maize production was compensated at the end of the growing season, care should be taken for shorter growing season crops.
   c. Onion compost surface application was shown to increase ryegrass N recovery, in comparison to incorporation, possibly because of faster mineralization of N and/or better crop establishment and development.
   d. The application of single rates of mineral fertilizer N under irrigation was not advantageous over composts for increasing forage maize or annual ryegrass production on light textured soils.

4. Forage maize dry matter yield was enhanced following biowaste and vegetable waste compost application to agriculture, by 0% to 477% compared to control soil. Annual ryegrass DM yield increased from 0% to 61% in comparison to control.
    a. Crop DM yield response to compost application depended significantly on soil quality. The poorer the soil, the higher the crop response.
    b. Higher DM yields were achieved with the use of higher rates of compost, and onion compost application.
    c. Supplementary N fertilization may be necessary for optimum crop production.
5. Properly managed biowaste and vegetable waste compost utilization (applying the compost based on its total N content while taking into consideration the compost C: N ratio, the crop N requirements and the soil N) is not likely to increase N leaching losses on light textured soils in the short term, in comparison to non-amended soil, even under irrigation and high drainage rate conditions.
    a. The maximum potential of mineral N leaching from compost amended soil was less than 6% of the total compost N applied.
    b. Onion compost application generally resulted in higher mineral N leaching than the biowaste compost, corresponding to the higher N mineralization from the onion compost. However, in almost all cases, the amount of mineral N leached from the onion compost amended soil was not significantly higher than the non-amended soil.
    c. N leaching in organic forms from compost amended light textured soil is not an important source of N leaching from the compost amended soil at least in the short term.
    d. Soil residual mineral N did not increase following compost application to light textured soils, but soil organic N did, hence explaining the limited N leaching despite the low crop N recovery.
    e. The application of mineral fertilizer N to light textured soils significantly increased the mineral N leaching in comparison to the control soil in all cases under consideration.

    f. Properly managed compost was shown advantageous over mineral fertilizers from a water quality perspective because of the slow release of the compost N mineralized.

6. Biowaste and vegetable waste compost application to light textured soil is likely to:
    a. Increase soil total C and OM content, therefore increasing C sequestration to the soil and improving soil quality.
    b. Increase soil extractable K content, linearly with compost application rates in response to compost soluble K content, thus reducing or replacing the need for the application of mineral fertilizer K.
    c. Influence soil available P content:
        i. Increase P solubility, hence potentially resulting in transport of soil available P downwards through the profile, below the root zone of coarse textured soils, and hence reduce P availability for plant uptake.
        ii. In calcareous alkaline soils the increase of P solubility prevents P fixation and consequently can potentially increase P availability for plant uptake.
        iii. Onion compost resulted in higher P availability for plant uptake compared to biowaste compost, for reasons related to the compost OM quality and the higher content of soluble P.
    d. Increase extractable Na levels and potentially introduce or enhance salinity and/or sodicity problems in heavy soils, in Na sensitive areas, or in the production of salinity sensitive crops (e.g. onions, carrots and strawberries), especially under a year-after-year application regime. In light textured soils, salinity or sodicity problems are not expected, especially under irrigation. Compost surface application resulted in lower soil Na levels than incorporation.
    e. Increase the water holding capacity and available water content of light soils, thus resulting in better crop production and water conservation.

   f. Onion compost was advantageous over biowaste compost in increasing soil available N, P, K and water content, hence resulting in better crop production.

7. Applying biowaste or vegetable waste compost to forage maize and annual ryegrass production was shown to be less profitable than mineral fertilization in the short term.
   a. Key factors influencing the costs of using compost in agriculture are the DM and N content of the compost and the haulage distance.
   b. Using compost, however, is a more sustainable practice due to the significantly increasing mineral fertilizer prices, the improvement of soil quality and the socio-environmental benefits of decreasing N leaching, reducing the production of greenhouse gases, decreasing the depleting of non-renewable resources, carbon sequestration to the soil and organic waste diversion from landfills.

10.2 Recommendations for further research

In this research work the effects of a short term compost application on soil properties, crop production and N leaching were studied. The longer term effects of compost application need to be assessed, especially in terms of N leaching potential as a large amount of organic N is being built up in the soil. Long term studies of the effects of using biowaste and vegetable waste compost in agriculture are necessary for the development of the optimum system for managing the utilization of compost for sustainable agriculture, which will ensure the environmental risk is minimized and the agronomic benefit is maximized.

This study provided evidence that the quality of compost carbon influences the availability of compost N. It was suggested that N mineralization is not driven only by compost C: N ratio, but the recalcitrance of carbon compounds is important, too. However, compost C quality was not taken into consideration within the compost characterization and therefore safe conclusions on the effects of compost C on N mineralization cannot be drawn directly. Compost N mineralization is suggested to be studied in relation to compost C quality so to better understand and more efficiently quantify the compost N mineralization potential. As compost N mineralization was found to depend on environmental conditions, controlled laboratory incubation trials involving different types of compost and soil, would allow for better determination of the relationship between carbon quality and N mineralization. However, field, pot and lysimeter experiments are also necessary to quantify compost N availability to crops.

A potential area of further work is related to the determination of P solubility and movement following compost application. This study produced evidence to suggest that compost application to coarse textured soils is likely to enhance the movement of inorganic P downwards through the soil profile, beyond the root zone. Work is necessary for the better understanding of the mechanisms and the potential environmental risks of water eutrophication due to potential P leaching. Lysimeter experiments and column leaching tests involving different types of soil and compost could contribute towards this direction. Composts would have to be additionally analyzed for their organic matter quality (e.g. humic substances), as there was evidence that it may interfere with inorganic P transport. Analyzing also for compost soluble and total P content is likely to provide with better understanding regarding P availability.

In this work the surface application of compost was shown to result in higher soil water retention and less soil Na concentration, which are important agricultural benefits, especially in certain areas. However, it is necessary to determine the potential gaseous N losses following the surface application of compost to ensure minimum environmental impacts.

Only light soils and a limited range of crops (forage maize and annual ryegrass) were considered within this study. Compost application to heavier soils and a wider range of crops, especially high value crops, is suggested for a better agronomic and economic evaluation.

Furthermore, the development of equipment able to apply compost at more precise rates to ensure that compost is spread more evenly on the soil, and the crops receive a better balanced supply of nutrients constitutes a potential area for future work, so that the efficiency of compost utilization is improved.

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A Appendices

A.1 Compost analysis

Tables A.1.1 and A.1.2 present the physicochemical properties of biowaste, onion, garden and MBT composts samples, collected in 2005. Table A.1.3 presents the physicochemical properties of biowaste and onion composts samples, collected in 2006.

All three tables include the number of samples, the arithmetic mean, the confidence interval (CI) of the mean at 95%, the median, minimum and maximum values, and the standard error of the mean (Std. Error) and they illustrate the variability of the compost materials.

Table A.1.1 Physical and chemical properties of the biowaste, onion, garden and MBT composts (samples collected in spring 2005).

Properties	**Sample No**	**Mean**	**Mean - 95% CI**	**Mean + 95% CI**	**Median**	**Min**	**Max**	**Std. Error**
Bulk density (g cm^{-3})								
Biowaste	11	0.41	0.39	0.43	0.41	0.35	0.45	0.01
Onion	6	0.43	0.30	0.56	0.46	0.27	0.57	0.05
Garden	3	0.55	0.39	0.71	0.57	0.48	0.60	0.04
MBT	3	0.42	0.29	0.55	0.40	0.38	0.48	0.03
DM (%)								
Biowaste	20	48.05	46.79	49.30	47.38	44.97	55.03	0.60
Onion	20	33.44	30.46	36.42	31.99	24.70	49.31	1.42
Garden	3	56.94	55.63	58.25	57.16	56.34	57.32	0.30
MBT	3	89.09	88.57	89.61	89.00	88.95	89.33	0.12
OM (%)								
Biowaste	10	45.66	44.50	46.82	45.76	42.81	48.68	0.51
Onion	10	37.07	30.45	43.69	37.97	18.87	48.63	2.93
Garden	3	37.37	33.09	41.66	38.17	35.39	38.55	1.00
MBT	3	27.24	20.99	33.50	27.47	24.62	29.64	1.45
pH								
Biowaste	10	8.9	8.8	9.0	8.8	8.7	9.1	0.05
Onion	10	7.1	6.1	8.1	7.1	4.5	8.5	0.43
Garden	3	8.9	8.8	9.0	8.9	8.9	9.0	0.03
MBT	3	8.2	8.0	8.3	8.2	8.1	8.2	0.04
EC_e (dS m^{-1}) – saturation paste								
Biowaste	3	5.8	5.6	6.0	5.8	5.7	5.9	0.05
Onion	3	4.3	3.8	4.8	4.3	4.1	4.5	0.11
Garden	3	3.5	3.2	3.8	3.4	3.4	3.6	0.07
MBT	3	3.8	3.7	3.9	3.8	3.8	3.9	0.02
Na (mg L^{-1}) – saturation paste								
Biowaste	3	543.2	525.4	561.1	547.4	535.0	547.4	4.14
Onion	3	140.7	123.4	158.0	139.1	134.7	148.3	4.02
Garden	3	193.1	173.4	212.7	193.0	185.2	201.0	4.57
MBT	3	1147.4	1092.6	1202.2	1147.3	1125.4	1169.5	12.73
EC_e (dS m^{-1}) BS EN 13038: 2000								
Biowaste	7	0.7	0.7	0.8	0.7	0.6	0.8	0.02
Onion	4	0.8	0.6	1.1	0.9	0.6	1.0	0.08
Na (mg kg^{-1}) BS EN 13652: 2001								
Biowaste	7	537.27	439.00	635.54	605.61	359.29	634.70	40.16
Onion	4	37.43	25.62	49.24	39.24	27.05	44.21	3.71

Table A.1.2 Nutrient status of the biowaste, onion, garden and MBT composts (samples collected in spring 2005): total N, total C, C: N ratio, and soluble K, N (NH_4-N and TON-N) and P.

Properties	**Sample No**	**Mean**	**Mean - 95% CI**	**Mean + 95% CI**	**Median**	**Min**	**Max**	**Std. Error**
Total N (%)								
Biowaste	36	1.76	1.64	1.88	1.58	1.25	2.27	0.06
Onion	36	1.20	1.13	1.27	1.20	0.64	1.54	0.03
Garden	9	1.54	1.34	1.74	1.60	1.01	1.83	0.09
MBT	9	0.90	0.88	0.93	0.91	0.86	0.96	0.01
Total C (%)								
Biowaste	36	27.51	26.23	28.79	26.89	23.43	41.32	0.63
Onion	36	19.12	17.67	20.56	19.66	9.61	28.40	0.71
Garden	9	22.85	20.68	25.03	23.53	16.76	25.96	0.94
MBT	9	15.36	14.93	15.79	15.18	14.71	16.15	0.19
C: N ratio								
Biowaste	36	16.22	14.84	17.61	16.48	11.72	33.10	0.68
Onion	36	16.51	14.92	18.10	16.78	10.17	25.44	0.78
Garden	9	14.94	14.34	15.55	14.72	14.13	16.55	0.26
MBT	9	16.99	16.45	17.53	17.13	15.92	17.87	0.23
K (g kg^{-1})								
Biowaste	10	7.24	6.81	7.68	7.07	6.41	7.97	0.19
Onion	10	10.53	8.20	12.85	10.03	5.84	16.98	1.03
Garden	3	5.07	4.63	5.51	5.13	4.87	5.20	0.10
MBT	3	2.35	1.92	2.78	2.45	2.15	2.45	0.10
NH_4-N (mg kg^{-1})								
Biowaste	10	221.03	104.12	337.93	269.47	0.00	370.24	51.68
Onion	10	227.70	-50.39	505.79	0.00	0.00	970.83	122.93
Garden	3	0.00	N.A.	N.A.	0.00	0.00	0.00	0.00
MBT	3	40.91	28.11	53.72	37.94	37.94	46.87	2.98
TON-N (mg kg^{-1})								
Biowaste	10	62.42	0.95	123.89	13.22	0.00	199.40	27.17
Onion	10	86.20	-5.52	177.93	17.30	0.00	325.48	40.55
Garden	3	0.00	N.A.	N.A.	0.00	0.00	0.00	0.00
MBT	3	0.00	N.A.	N.A.	0.00	0.00	0.00	0.00
PO_4-P (mg kg^{-1})								
Biowaste	10	268.29	211.25	325.34	304.13	145.02	357.02	25.22
Onion	10	202.49	57.29	347.69	141.15	14.15	564.44	64.19
Garden	3	211.81	170.27	253.35	211.81	195.09	228.53	9.65
MBT	3	5.21	-1.19	11.61	6.70	2.23	6.70	1.49

Table A.1.3 Physical and chemical properties of the biowaste and onion composts (samples collected in spring 2006). Mean values of each property labelled with the same letter are not significantly different (P>0.05).

Properties	**Sample No**	**Mean**	**Mean - 95% CI**	**Mean + 95% CI**	**Median**	**Min**	**Max**	**Std. Error**
Bulk density (g cm^{-3})								
Biowaste	6	0.24 a	0.23	0.26	0.24	0.22	0.27	0.01
Onion	6	0.67 b	0.64	0.69	0.67	0.64	0.70	0.01
DM (%)								
Biowaste	5	70.01 b	66.60	73.41	69.22	67.71	74.42	1.23
Onion	5	50.54 a	48.30	52.77	50.70	48.24	52.93	0.80
OM (%)								
Biowaste	9	72.63 b	71.27	73.98	72.17	70.21	75.81	0.59
Onion	9	19.90 a	17.60	22.20	18.44	17.80	26.54	1.00
pH								
Biowaste	5	8.5 b	8.3	8.7	8.5	8.3	8.7	0.06
Onion	5	8.2 a	8.1	8.3	8.2	8.1	8.3	0.04
EC (dS m^{-1}) BS EN								
Biowaste	5	1.0 a	N.A.	N.A.	1.0	1.0	1.0	0.00
Onion	5	1.4 b	1.37	1.51	1.4	1.4	1.5	0.02
Na (g kg^{-1}) BS EN								
Biowaste	5	1.97 b	1.67	2.27	1.96	1.65	2.29	0.11
Onion	5	0.53 a	0.45	0.62	0.58	0.45	0.59	0.03
Total N (%)								
Biowaste	9	1.89 b	1.74	2.04	1.89	1.62	2.22	0.07
Onion	9	0.98 a	0.93	1.04	0.99	0.88	1.06	0.02
Total C (%)								
Biowaste	9	37.53 b	36.16	38.90	37.73	34.00	39.64	0.59
Onion	9	10.31 a	9.61	11.02	10.46	8.95	11.57	0.31
C: N ratio								
Biowaste	9	20.09 b	18.09	22.08	20.26	16.79	24.51	0.87
Onion	9	10.47 a	10.06	10.88	10.24	9.89	11.27	0.18
K (g kg^{-1})								
Biowaste	5	8.57 b	7.69	9.45	8.61	7.58	9.57	0.32
Onion	5	6.70 a	6.46	6.95	6.65	6.45	6.95	0.09
NH_4-N (mg kg^{-1})								
Biowaste	5	202.52 b	170.39	234.66	198.69	176.40	237.63	11.57
Onion	5	9.27 a	6.98	11.57	9.42	6.54	11.74	0.83
TON-N (mg kg^{-1})								
Biowaste	5	0.00 a	N.A.	N.A.	0.00	0.00	0.00	0.00
Onion	5	180.80 b	114.42	247.18	160.21	120.79	243.23	23.91
PO_4-P (mg kg^{-1})								
Biowaste	5	362.81 a	82.53	643.09	301.26	80.73	600.34	100.95
Onion	5	521.89 a	361.93	681.86	572.17	304.06	636.62	57.62

A.2 Common methods

This section of the Appendices summarizes the common laboratory methods used for the analysis of soil, plant and leachate samples produced in the field, pot, lysimeter and incubation experiments, and the sampling technique involved.

A.2.1 Soil sampling and analysis

Soil analysis was undertaken in all 4 experiments. Table A.2.1 presents the chemical and physical soil properties determined in each experiment.

In the field, soil samples were taken from the following layers: 0-0.3 m and 0.3-0.6 m, using an auger. Three soil samples were collected from each plot and mixed together to produce one bulk sample. Undisturbed samples were also collected by inserting density rings into the soil. In the lysimeter experiment, soil samples were taken from the following soil layers: 0-0.25 m, 0.25-0.45 m and 0.45-0.65 m, using an auger. Two soil samples were collected from each lysimeter and mixed together to produce one bulk sample. Undisturbed samples were also collected.

Samples were collected form the soil in pots using a small auger (approximately 0.02 m diameter). At least three samples were collected from each pot and mixed together to produce one bulk sample. In the incubation trial, as the sampling technique was destructive, all the soil in the incubation pots was used as sample for analysis, after it was removed from the incubator.

Table A.2.2 presents information regarding the determinants, the method of analysis and the determination technique employed, along with reference for each treatment. The methods used are not described in detail as they constitute standard laboratory methods.

Table A.2.1 Summary of the soil analysis carried out in each experiment.

Measurements	**Experiments**			
	Field	Pots	Lysimeters	Incubation
DM	+	+	+	+
OM	+	+	+	-
pH	+	+	+	-
Total N	+	+	+	+
Total C	+	+	+	+
Total P	-	-	+	-
TSN	-	-	-	+
NH_4-N	+	+	+	+
TON-N	+	+	+	+
Extractable P	+	+	+	-
Extractable K	+	+	+	-
Extractable Na	+	+	+	-
Available Ca	-	-	+	-
EC_e	+	+	+	-
Ca + Mg in saturation	+	+	+	-
Na in saturation	+	+	+	-
WRC	+	-	+	-

Table A.2.2 Soil analysis: determinants and methods synopsis.

Determinant	Sample type	Method/Method synopsis	Determination	Reference/Lab reference
Total C	***	Catalytic tube combustion (30 mg soil)	Elemental Analyzer	NR-SAS/SOP 9
Total N	***	Catalytic tube combustion (30 mg soil)	Elemental Analyzer	NR-SAS/SOP 11
Total soluble N		10 g soil extracted with 50 ml K_2SO_4 (0.5 M)	Segmented Flow Analysis (SFA)	Jones & Willet, 2006
NH4-N	‡	20 g soil extracted with 100 ml KCl (2 M)	SFA	NSRI/AL/SOP 13
TON-N	‡	20 g soil extracted with 100 ml KCl (2 M)	SFA	NSRI/AL/SOP 13
Total P	*	Aqua Regia: 0.5 g soil extracted with HCl + HNO_3 with microwave digestion	Spectrophotometer	NR-SAS/SOP 17
Extractable P	*	Olsen: 2.5 g soil extracted with 50 ml $NaHCO_3$ (0.5 M)	Spectrophotometer	NSRI/AL/SOP 15
Extractable K	*	10 g soil extracted with 50 ml NH_4NO_3 (1 M)	Flame photometer	NSRI/AL/SOP 14
Extractable Na	*	10 g soil extracted with 50 ml NH_4NO_3 (1 M)	Flame photometer	NSRI/AL/SOP 14
Extractable Ca	*	10 g soil extracted with 50 ml NH_4NO_3 (1 M)	Flame photometer	NSRI/AL/SOP 14
Extractable Mg	*	10 g soil extracted with 50 ml NH_4NO_3 (1 M)	Spectrophotometer	MAFF, 1986
OM	*	Loss on ignition: 5 g soil placed in the furnace at 450 °C for minimum 3.5 hrs		NR-SAS/SOP 23
pH	*	Water suspension (1: 2.5 w/w): 30 g soil + 75 ml water	pH-meter	NR-SAS/SOP 6
EC_e	*	250 g soil + necessary water for saturation, extraction with suction	EC_e-meter	NR-SAS/SOP 14
Ca + Mg in saturation	*	Saturation paste extract (the sane used for EC_e)	Titration with 0.01N EDTA	Richards, 1954
Na in saturation	*	Saturation paste extract (the sane used for EC_e)	Flame photometer	Richards, 1954
Texture	*	Pipette method (10 g soil)		NR-SAS/SOP 5
WRC	#	Sand table & pressure membrane methods		NR-SAS/SOP 24
DM	‡	Oven drying: 30 g soil placed in the oven at 103 °C for 48 hr		NR-SAS/SOP 3

‡ Analysis in moist soil - no sample preparation involved
* Analysis in air-dried soil - sieved via 2 mm
*** Analysis in oven dried (103 °C for 2 hrs) soil - sieved via 0.5 mm (sub-sampled from the air-dried/ 2mm sieved soil sample)
# Undisturbed moist soil sample

A.2.2 Plant growth monitoring, sampling and analysis

Forage maize growth in the field was monitored by measuring the plant height of 8 plants in the centre of each plot. Plant height was defined as the distance from the back of the collar of the upper complete leaf to the soil surface. A leaf was deemed to be complete when its collar was completely formed (Easson & Fearnehough, 2000). Forage maize growth monitoring in lysimeters and pots was carried out in all plants present in the lysimeter or pot. Shoot diameter was also measured at a distance of 0.06 m from the soil surface using a vernier calliper.

At harvest, 3 of the 8 plants used for growth monitoring in the field were randomly selected and harvested by hand (approximately 0.05 m above the soil surface). All maize plants present in the lysimeter and pot experiments were harvested (approximately 0.02 m above the soil surface) and used for the analysis.

The harvested forage maize plants were separated to two parts: the cobs and the rest of the plant. Both parts of forage maize plant were chopped. The chopped samples were weighted fresh and then oven dried to determine the DM content of cobs and the rest of the plant separately. Maize cobs were dried at 60 ˚C until they reached constant weight and the rest of the plant at 105 ˚C for 26 hours. The dry weight of the sample was recorded. Then, the dried samples were grinded through a 2 mm sieve. The resulted samples were sub-sampled and grinded through a 1 mm sieve.

The annual ryegrass was harvested by cutting the plant tops 0.03 m above the soil surface. At each harvest time, DM content and yield were determined by drying the ryegrass tops at 60 ˚C until constant weight. The dried sample was then grinded through a 1 mm sieve.

The dried plant samples were analyzed for total N, total P and total K content, as summarized in Table A.2.3. Forage maize total N concentrations were determined separately for the cobs and the rest of the maize plant. However, the analysis for P and K was on the bulk sample (cobs + the rest of the plant), which was produced according to the percentage (w/w) of each constituent of the whole plant DM. Ryegrass was only analyzed for total N.

Table A.2.3 Plant analysis: determinants and methods synopsis.

Determinant	Method/Method synopsis	Determination	Lab reference
Total N	Catalytic tube combustion (30 mg plant material)	Elemental Analyzer	NR-SAS/SOP 11
Total P	Dry combustion (2 g plant material) and acid digestion (HCl (6 M))	Spectrophotometer	NSRI/AL/SOP 18 & NSRI/AL/SOP 19
Total K	Dry combustion (2 g plant material) and acid digestion (HCl (6 M))	Flame photometer	NSRI/AL/SOP 18

A.2.3 Leachate sampling and analysis

Leachate samples were taken on site, by sub-sampling from the total amount of leachate collected in each bucket, using 60 ml plastic bottles. Volume measurements were taken from the remaining amount of leachate. Leachate volume was determined using a volumetric cylinder. Then the buckets were emptied and put back in their place.

Samples after collection were filtered through Whatman No 4, to remove impurities, which could interfere with the measurements. The filtered samples were stored in the fridge (4 °C) and analyzed within one week from the sampling date. If necessary to store the samples for longer periods than one week, the filtered samples were placed in the freezer (-20 °C) and analyzed as soon as possible. Leachate samples analysis is summarized in Table A.2.4

Table A.2.4 Leachate analysis: determinants and methods synopsis.

Determinant	Experiment		Determination	Reference
	Pots	Lysimeters		
TDN	+	+	SFA	NR-SAS/SOP 28
NH_4-N	+	+	SFA	NR-SAS/SOP 31
TON-N	+	+	SFA	NR-SAS/SOP 32
PO_4-P	-	+	SFA	NR-SAS/SOP 33

For the determination of soil, plant, compost and leachate properties standard laboratory equipment was used. Below are presented a few details regarding the automated equipment used: the TCD Elemental Analyzer and the SFA.

CHNOS Elemental Analyser VARIO EL III

The elementary analyser Vario EL is fully automatic instrument for the quantitative determination of Carbon, Nitrogen and Sulphur. It works according to the principle of catalytic tube combustion in an oxygenated CO_2 atmosphere and high temperatures. The combustion gases are freed from foreign gases (i.e. volatile halogen). The desired measuring components are separated from each other with the help of specific adsorption columns and determined in succession with a thermal conductivity detector (TCD). Helium (He) serves as flushing and carrier gas. The automatic control of the analysis procedure is accomplished through the software.

BURKARD Series 2000 SFA (Segmented Flow Analyser)

The segmented continuous analysis technique involves mixing a liquid sample with reagents pumped in a continuously flowing stream and segmented with air bubbles. The sample and reagents are mixed by passing through glass coils and also through a temperature control coil if heat is required to speed development of the colour complex before detection using a colorimeter, fluorimeter, or other suitable detector in the UV and visible wavelength ranges. A cellophane dialysis membrane is used to remove the high molecular weight components contained in samples, as they were interfering with the chemical reactions. Within the SFA, NH_4-N was determined by the automated phenate method, TON-N (NO_3-N and NO_2-N) by the automated hydrazine reduction method and PO_4-P was determined by the automated ascorbic acid reduction method. Total soluble/dissolved N was first oxidised to TON-N and then was determined by the automated hydrazine reduction method. During the analysis the performance of the analyser is checked regularly (every 10 samples) against a known concentration (drift). Standards depend on the sample type: leachate sample (standards in water), soil or compost extracts (standards in the extractant).

A.3 Experiments

This section of the Appendices refers to Chapters 4, 5 and 6 providing supplementary information about the field, pot and lysimeter experiments, respectively.

A.3.1 Field experiment

A.3.1.1 Layout, tillage, monitoring and climatic conditions

Figure A.3.1.1 illustrates the layout of the field experiment in 2006. The circled area was added in 2006 in order for a mineral fertilizer rate of 40 kg N ha^{-1} (Mineral_40) to be included in the study. In 2005 these plots were spare plots which received the same treatment as the control. In order not to have all the Mineral_40 plots in the same area, the plot of block 2 used as control (0 kg N ha^{-1}) in 2005 was used as Mineral_40 in 2006. Compost application rates 1, 2 and 3 corresponded to the application of 80, 165 and 250 kg total N ha^{-1} in 2005, whereas in 2006 to 100, 250, 400 kg total N ha^{-1}, respectively.

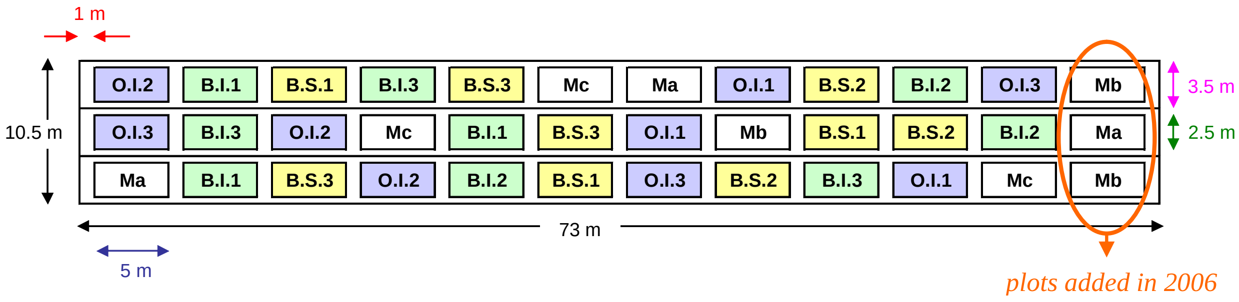


Figure A.3.1.1 Field experiment layout in 2006 (B: biowaste, O: onion, M: mineral fertilizer; I: incorporated, S: surface applied; a, b, c: application rates of 0, 40, 80 kg N ha^{-1} (mineral fertilizer); 1, 2, 3: application rates in kg total N ha^{-1} (compost). From top to bottom: blocks 1 to 3. The circled area was added in 2006.

Table A.3.1.1 summarizes the experimental establishment and monitoring showing the dates of the experiment establishment practices, along with the soil and crop monitoring dates.

Table A.3.1.1 Dates of experimental establishment and monitoring.

Date	Practices
18-May-05	Soil sampling START
25-May-05	Field ploughing
01-Jun-05	Compost incorporation (power harrow) & seedbed preparation (ring rolls)
09-Jun-05	Forage maize sowing
09-Jun-05	Biowaste compost and mineral fertilizer surface application
16-Jun-05	Spraying with 'Stomp' herbicide
17-Jun-05	Crop emergence started
21-Jun-05	Crop growth measurements
11-Jul-05	Crop growth measurements
31-Jul-05	Crop growth measurements
20-Aug-05	Crop growth measurements
19-Sep-05	Crop harvest
26-Sep-05	Soil sampling
16-Mar-06	Soil sampling
May 06	Field ploughing
08-Jun-06	Compost incorporation (power harrow)
16-Jun-06	Seedbed preparation (power harrow)
19-Jun-06	Forage maize sowing
19-Jun-06	Biowaste compost and mineral fertilizer surface application & tillage with ring rolls
26-Jun-06	Crop emergence started
27-Jun-06	Spraying with 'Stomp' herbicide
29-Jun-06	Irrigation with 20 mm water & sowing new seeds in the gaps (block 2 - first 3 plots)
25-Jul-06	Soil sampling
02-Oct-06	Crop harvest
09-Oct-06	Soil sampling

Prior to compost incorporation, the composts were spread onto the respective plots by hand, as evenly as possible. Then they were incorporated within the top approximately 0.15 m depth by a power harrow. The power harrow was driven over the whole field (not only over the respective plots). In 2006 due to the very dry conditions, the field was power harrowed twice in order to produce a better seedbed.

After the sowing was completed, biowaste compost and mineral fertilizers were surface applied to the respective plots.

Weed control was carried out by applying the herbicide ‘Stomp’ (Pendimethalin) about 1 week after maize sowing each year. Stomp was applied at the rate of 3.75 litres ha^{-1}, in

200 litres of water ha^{-1}. In 2005 no irrigation was applied, whereas in 2006, the field was irrigated with 20 mm water 10 days after crop emergence, due to the warm and dry weather conditions.

Table A.3.1.2 presents information regarding the climatic conditions during the experimental period.

Table A.3.1.2 Weather data in a monthly basis during the field experiment – mean values.

Date	**Tmax °C**	**Tmin °C**	**RH %**	**Sun hr**	**Solar MJ m^{-2}**	**Net MJ m^{-2}**	**Wind km d^{-1}**	**ET mm d^{-1}**	**Rain mm d^{-1}**
May-05	16.0	6.3	80.9	4.5	14.7	9.2	159.2	2.3	1.5
Jun-05	20.5	10.2	82.8	4.4	15.6	10.1	136.1	2.7	2.2
Jul-05	21.1	12.7	84.6	4.0	14.2	9.2	129.4	2.6	0.9
Aug-05	21.8	11.1	80.4	5.1	13.7	8.2	111.9	2.4	1.7
Sep-05	20.2	10.7	88.1	3.9	9.6	5.3	122.8	1.6	1.8
Oct-05	16.7	10.1	94.1	2.1	5.1	2.5	137.6	0.7	2.0
Nov-05	9.3	2.3	94.8	1.7	3.0	0.8	166.1	0.2	1.3
Dec-05	7.1	1.9	96.1	0.8	1.8	0.4	165.2	0.1	1.1
Jan-06	6.5	2.1	96.8	0.5	1.9	1.1	154.1	0.3	0.7
Feb-06	6.0	1.5	90.6	1.3	3.8	2.0	180.0	0.5	1.3
Mar-06	8.2	2.4	84.8	2.0	6.9	3.8	215.2	1.0	1.3
Apr-06	12.8	5.3	84.0	3.2	11.2	6.7	173.0	1.7	1.2
May-06	16.7	8.7	88.8	3.2	12.9	8.3	164.4	2.0	2.8
Jun-06	21.7	11.3	78.8	6.0	17.9	11.2	92.3	3.1	0.5
Jul-06	26.6	14.4	77.0	6.0	17.2	10.7	120.9	3.5	0.8
Aug-06	20.5	12.6	85.6	2.5	10.8	6.9	135.7	2.1	3.2
Sep-06	21.8	13.4	91.0	3.7	9.5	5.4	140.1	1.7	1.5
Oct-06	16.8	10.2	97.5	1.9	5.0	2.5	147.6	0.7	1.7

A.3.1.2 Calculation of mineral fertilizer application rates

A.3.1.2.1 Nitrogen

Mineral fertilizer nitrogen (ammonium nitrate) was applied to separate plots to determine the N availability from the composts. In the first year of the experiment (2005), one rate of mineral fertilizer nitrogen (80 kg N ha^{-1}) was used. This rate corresponded to the recommended amount of N for the production of forage maize in a soil with soil nitrogen supply (SNS) index equal to 1 (MAFF, 2000). The SNS index was estimated by the SNS analysis method (MAFF, 2000), according to the Equation A.3.1.1.

SNS = SMN (0-0.9 m) + estimate of total crop N content at the time of sampling for SMN + estimate of organic N mineralised during the growing season (A.3.1.1)

The soil mineral nitrogen (SMN) was estimated for the top 0.6 m depths, because of the sandstone present at 0.6 m depths. The sampling for SMN was carried out in May 2005. SMN was 73.3 kg ha^{-1}. At the time of sampling there was no crop in the field and therefore the crop N uptake at the time was considered as zero. As there was no crop in the field, the estimation of the potentially mineralised organic N during the growing season was not directly possible. It was assumed that the soil mineralizable N was low, and thus it was hypothesized that the SNS index was 1 or 2. In order to ensure the mineral fertilizer treatment would provide the necessary N for the maximum yield, the SNS index was taken to be equal to 1, and consequently the recommended amount of N for the production of forage maize was 80 kg N ha^{-1}.

In 2006, the estimation of SNS was carried out in September, after the forage maize harvest. The SMN (0-0.6 m) and crop N uptake were measured and the N mineralized was estimated. The results indicated that the SNS index was 6. This finding, along with the crop yield results of both years of the study, indicate that the SNS Index in the first year was underestimated, as the extended mineralization of soil inherent N was not predicted. At an SNS index of 6 no fertilizer N addition is necessary for optimum forage maize production. Despite that, two rates of mineral fertilizer N were used (40 and 80 kg N ha^{-1}) in order to provide for an estimate of the compost N mineralization via the measurements of soil mineral N during the second year of the field trial.

A.3.1.2.1 Phosphorus and potassium

Mineral fertilization with P and K corresponded to the recommended amounts of P and K for the optimum production of forage maize in the sandy loam soil of the field site. P and K requirements were based on the soil content of extractable P and K. Soil P and K index were estimated by the soil classification indices method (MAFF, 2000).

The soil was rich in phosphorus (soil P index was 5-6) and therefore no fertilizer P was necessary to be applied for optimum forage maize production in both years of the study.

Soil K index was 3 in both years. Hence, the recommended amount of K for optimum forage maize production was 91.3 kg ha^{-1} yr^{-1}. Fertilizer K (sulphate of potash) was applied in 2005 only to the mineral fertilizer plots (also amended with 80 kg ha^{-1} of mineral fertilizer N). In 2006, fertilizer K (sulphate of potash) was applied to all reference treatments (mineral N fertilizer rates of 0, 40 and 80 kg ha^{-1}) in order to ensure that K would not be a limiting factor for crop growth, so as to better study the effects of soil N mineralization on crop production.

A.3.1.3 Forage maize production in 2005 and 2006 - pictures

Forage maize growth was vigorous and the plants were well developed in both years of the experiment. No deficiency symptoms were observed during both years of the study. Figure A.3.1.2 illustrates the field trial in 2005 at different stages of crop growth.

In general, no differences could be observed by eye within the different plots of the trial during the whole experimental period, supporting the results presented in the main document.

Figure A.3.1.3 illustrates the field trial in 2006, two weeks before the crop harvest. It can be seen that forage maize plants were well-developed. No difference in plant height or other morphological characteristics can be observed.

A minimum of two cobs was present in each plant in both years of the study. Figure A.3.1.4 shows a sample of maize cobs collected from the control plots in September 2005. Figures A.3.1.2 to A.3.1.4 are indicative of the crop development in both years of the study and can support the high DM yield and N uptake results reported in both years of the experiment.

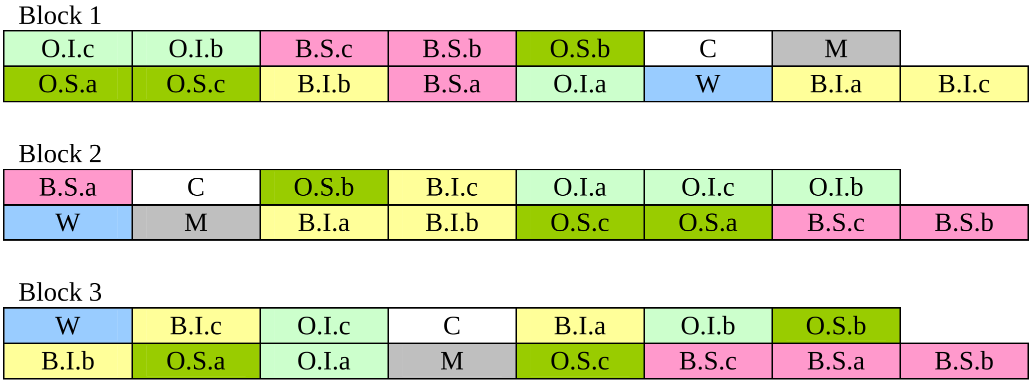
Figure A.3.1.2 View of the field trial in 2005 at different stages of forage maize growth: 5th-6th leaf stage, 3.5 weeks after crop emergence (top); 9th-10th leaf stage, 6 weeks after crop emergence (middle); complete development, a few days before harvest (bottom).

Figure A.3.1.3 View of the field trial 2 weeks before the crop harvest in 2006.

Figure A.3.1.4 Forage maize cobs collected for analysis from the control plot in 2005.

A.3.2 Pot experiment

Figure A.3.2.1 illustrates the pot experiment layout in 2005. The water-control pots were used to monitor soil moisture in order to ensure that water was not a limiting factor for crop growth. A theta-probe and an equi-tensiometer were placed in the water-control pots (Figure A.3.2.2), which contained non-amended soil, to monitor soil moisture content. Temperature and moisture data were collected using a data logger.



Block 1

O.I.c	O.I.b	B.S.c	B.S.b	O.S.b	C	M	
O.S.a	O.S.c	B.I.b	B.S.a	O.I.a	W	B.I.a	B.I.c

Block 2

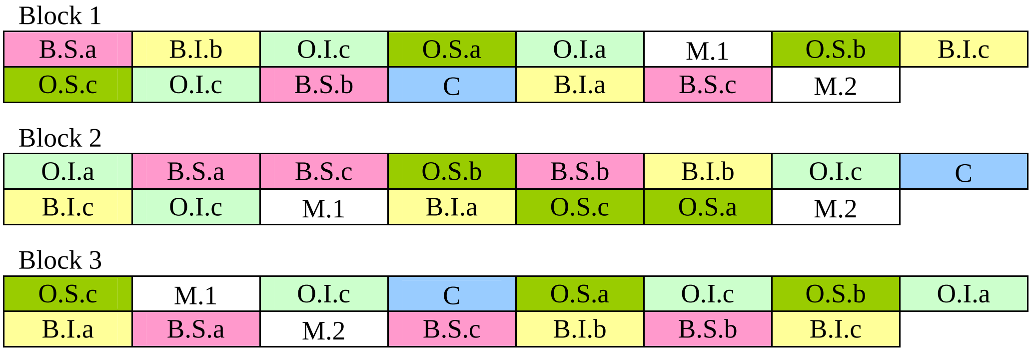
B.S.a	C	O.S.b	B.I.c	O.I.a	O.I.c	O.I.b	
W	M	B.I.a	B.I.b	O.S.c	O.S.a	B.S.c	B.S.b

Block 3

W	B.I.c	O.I.c	C	B.I.a	O.I.b	O.S.b	
B.I.b	O.S.a	O.I.a	M	O.S.c	B.S.c	B.S.a	B.S.b

Figure A.3.2.1 Pot experiment layout in 2005 (O: onion compost, B: biowaste compost, M: mineral fertilizer, C: control, W: water-control; I: incorporated, S: surface applied; a,b,c: compost application rates corresponding to 80, 165, 250 kg total N ha^{-1}, respectively).

Figure A.3.2.2 Soil moisture control equipment.



Block 1

B.S.a	B.I.b	O.I.c	O.S.a	O.I.a	M.1	O.S.b	B.I.c
O.S.c	O.I.c	B.S.b	C	B.I.a	B.S.c	M.2	

Block 2

O.I.a	B.S.a	B.S.c	O.S.b	B.S.b	B.I.b	O.I.c	C
B.I.c	O.I.c	M.1	B.I.a	O.S.c	O.S.a	M.2	

Block 3

O.S.c	M.1	O.I.c	C	O.S.a	O.I.c	O.S.b	O.I.a
B.I.a	B.S.a	M.2	B.S.c	B.I.b	B.S.b	B.I.c	

Figure A.3.2.3 Pot experiment layout in 2006 (O: onion compost, B: biowaste compost, M: mineral fertilizer, C: control; I: incorporated, S: surface applied; a, b, c: compost application rates corresponding to 100, 250, 400 kg total N ha^{-1}, respectively; 1, 2: mineral fertilizer N rates corresponding to 180, 360 kg N ha^{-1}, respectively).

Figure A.3.2.3 presents the pot experiment layout in 2006. Table A.3.2.1 summarizes the experimental establishment and monitoring by showing the dates of the experiment establishment practices, along with the soil and crop monitoring dates. Table A.3.2.2 shows the temperature range during the experimental periods in 2005 and 2006.

Table A.3.2.1 Pot experiment establishment and monitoring in 2005 and 2006.

Date	**Practice**
09/05/05	Establishment of the experiment
11/05/05	Leachate sampling
12/05/05	Leachate sampling
15/05/05	Leachate sampling
19/05/05	Crop emergence started
23/05/05	Crop growth measurements
25/05/05	Leachate sampling
31/05/05	Leachate sampling
01/06/05	Thinning & crop growth measurements
10/06/05	Leachate sampling
11/06/05	Crop growth measurements
20/06/05	Leachate sampling
21/06/05	Crop growth measurements
30/06/05	Leachate sampling
01/07/05	Crop growth measurements
10/07/05	Leachate sampling
11/07/05	Crop growth measurements
20/07/05	Leachate sampling
21/07/05	Crop growth measurements
30/07/05	Leachate sampling
31/07/05	Crop growth measurements
10/08/05	Leachate sampling

20/08/05	Leachate sampling
23/08/05	Harvest
06/09/05	soil sampling & leave soil to air-dry
12/05/06	Compost incorporation
01/06/06	Sulphate of potash slightly incorporated within the top about 2.5 cm soil in the mineral fertilizer treatments
01/06/05	Ryegrass sowing
02/06/05	Surface application of mineral fertilizer N (NH_4NO_3) and composts to the respective pots
05/06/06	The pots were moved outdoors
12/06/06	Crop emergence started
15/06/06	Leachate sampling
28/06/06	Leachate sampling
13/07/06	Leachate sampling
26/07/06	Leachate sampling
09/08/06	Leachate sampling
16/08/06	Crop sampling (1st cut)
23/08/06	Leachate sampling
06/09/06	Leachate sampling
21/09/06	Leachate sampling
27/09/06	Crop sampling (2nd cut)
28/09/06	The pots were placed back in the greenhouse
04/10/06	Leachate sampling
18/10/06	Leachate sampling
01/11/06	Leachate sampling
14/11/06	Crop sampling (3rd cut)
15/11/06	Leachate sampling
11/12/06	Leachate sampling
11/12/06	Crop sampling (4th cut)
11/12/06	Soil sampling

Table A.3.2.2 Minimum and maximum air temperature in the greenhouse during the experimental periods in 2005 and 2006. The temperatures in italics represent the air temperature when the experiment was held outdoors.

Month	**Temperature (°C)**			
	2005		2006	
	min	max	min	max
May	4.1	24.6	-	-
June	6.8	44.6	*11.3*	*21.7*
July	11.2	42.2	*14.2*	*26.6*
August	9.5	28.7	*12.6*	*20.5*
September	-	-	*13.5*	*21.8*
October	-	-	7.8	24.6
November	-	-	0.8	20.1
December	-	-	1.2	23.3

Figure A.3.2.4 presents the amounts of leachate produced in 2005 and 2006. The effect of the different irrigation strategy employed in 2005 and 2006, significantly affected the amount of leachate produced.

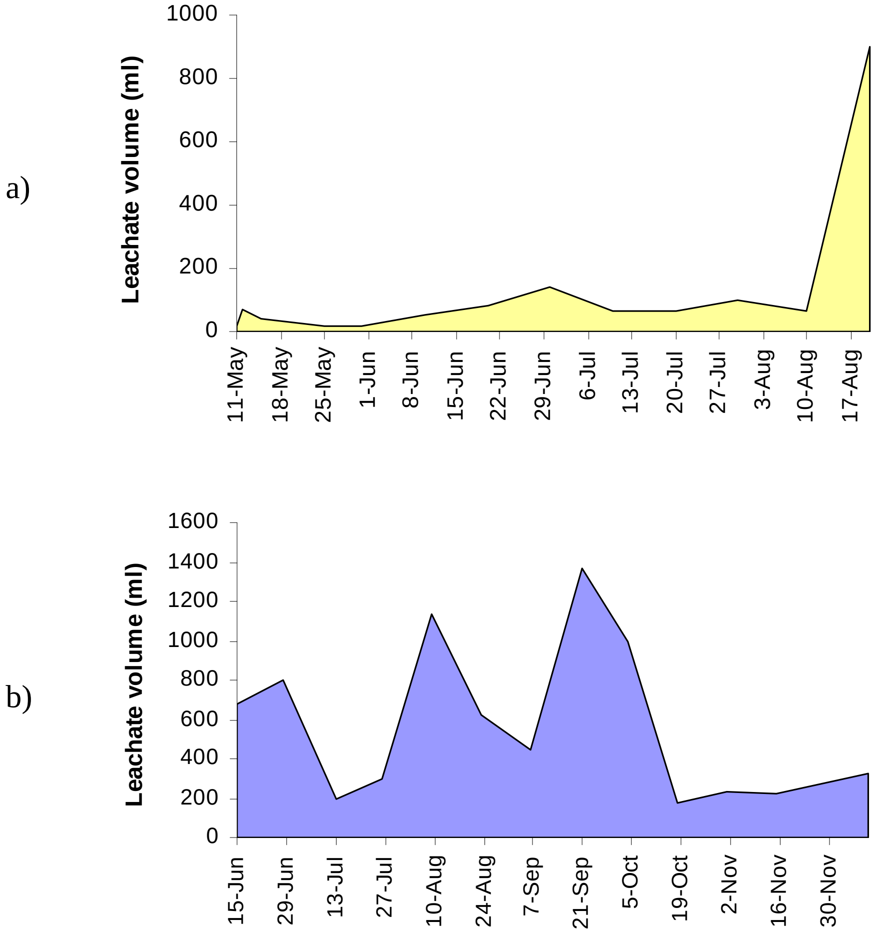


Figure A.3.2.4 The volume of leachate (average of all treatments) produced per pot between each sampling date (a) in 2005 and (b) in 2006.

Figure A.3.2.5 shows the effect of the different compost and mineral fertilizer treatments on forage maize growth parameters in 2005.

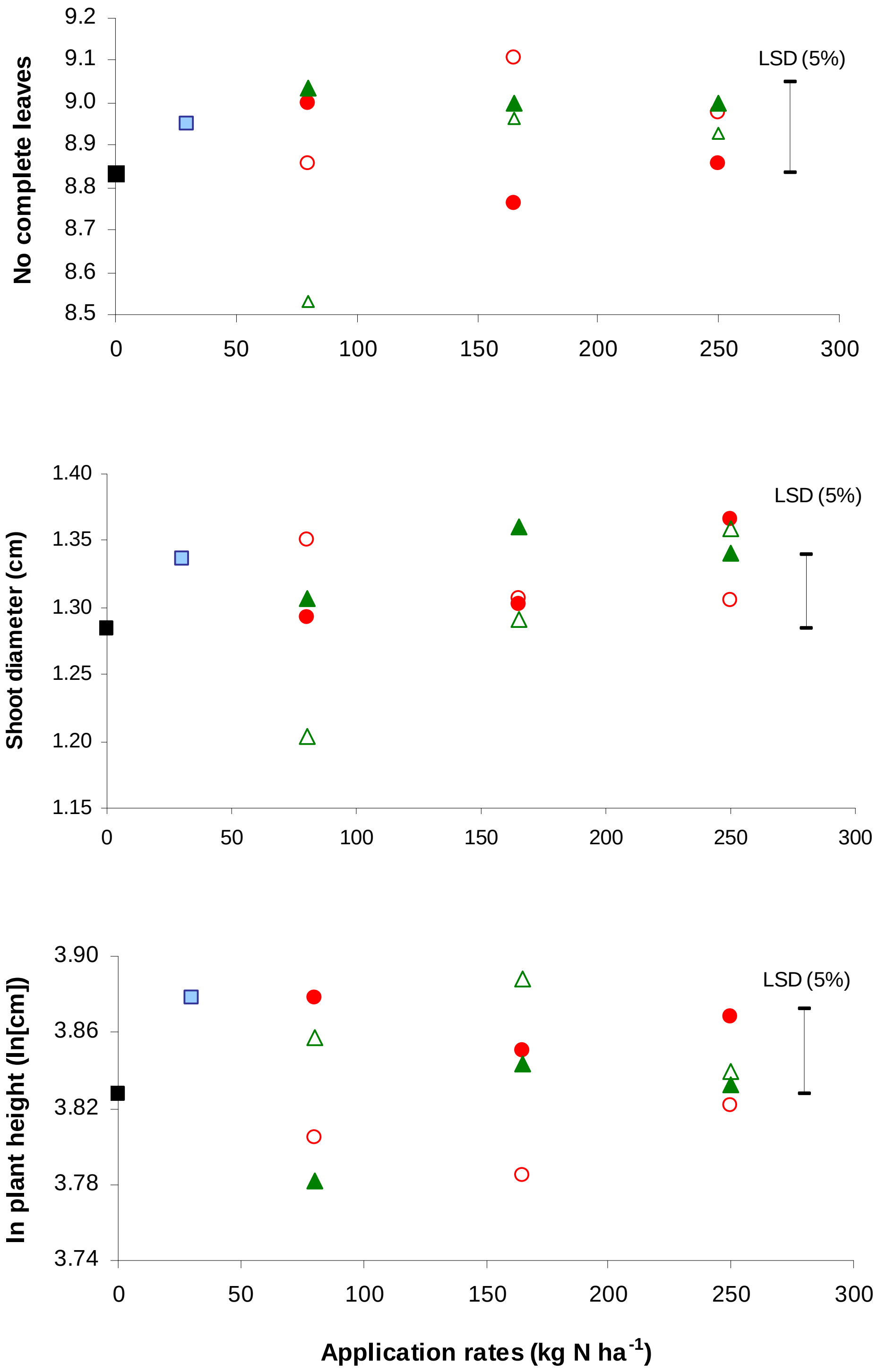


Figure A.3.2.5 The forage maize number of complete leaves (top), shoot diameter (middle) and plant height (bottom – natural logarithm) response to compost and mineral fertilizer application at different rates (means of the whole growing season in 2005) (■ : Control, 🟦 : Mineral, ○: Biowaste surface applied, ●: Biowaste incorporated, ▲: Onion incorporated, △: Onion surface applied).

A.3.3 Lysimeter experiment

Figure A.3.3.1 illustrates the layout of the lysimeter experiment. Table A.3.3.1 summarizes the experimental establishment and monitoring, showing the dates of the experiment establishment practices, along with the soil, crop and leachate sampling dates.

O_100	O_250	B_250	M_40	O_600	B_100
B_600	M_120	M_0	O_250	M_120	B_250
M_40	B_400	O_600	B_600	M_40	M_80
O_400	M_0	B_100	B_400	O_400	B_600
M_80	B_100	O_100	M_80	B_400	O_250
O_600	B_250	M_120	O_400	O_100	M_0

Figure A.3.3.1 Lysimeter experiment layout (M: mineral fertilizer, B: biowaste compost, O: onion compost; _0, 40, 80, 120: application rates in kg N ha^{-1}; _100, 250, 400, 600: application rates in kg total N ha^{-1}).

Figures A.3.3.2 illustrates the total amount of water (rainfall and irrigation) applied daily per lysimeter. Figure A.3.3.3 shows the total amount of water (rainfall and irrigation) applied per lysimeter, along with the amounts of leachate produced. Tables A.3.3.2 and A.3.3.3 present information regarding the climatic conditions during the experimental period.

Table A.3.3.1 Dates of the establishment of the lysimeter experiment and the monitoring of soil, crop and leachate.

Date	Practice
18/05/06	Compost incorporation
29/05/06	Leachate sampling 1st
30/05/06	Application of P & K fertilizers
01/06/06	Sowing (6 seeds per lysimeter)
02/06/06	N fertilizer surface application
08/06/06	Crop emergence started
08/06/06	Leachate sampling 2nd
15/06/06	Leachate sampling 3rd
21/06/06	Application of Mg fertilizers
26/06/06	Maize thinning to 3 plants lysim^{-1}
26/06/06	Maize growth measurements 1
26/06/06	P deficiency symptoms observed for the 1st time
27/06/06	Soil sampling
28/06/06	Leachate sampling 4th
13/07/06	Leachate sampling 5th
18/07/06	Supplementary fertilization with P & K
18/07/06	Maize growth measurements 2
26/07/06	Leachate sampling 6th
10/08/06	Leachate sampling 7th
23/08/06	Maize growth measurements 3
24/08/06	Leachate sampling 8th
05/09/06	Leachate sampling 9th
11/09/06	Maize growth measurements 4
11/09/06	Maize harvest
12/09/06	Leachate sampling 10th
12/09/06	Soil sampling
12/10/06	Leachate sampling 11th
06/11/06	Leachate sampling 12th
06/06/07	Leachate sampling
06/07/07	Leachate sampling
19/08/07	Leachate sampling

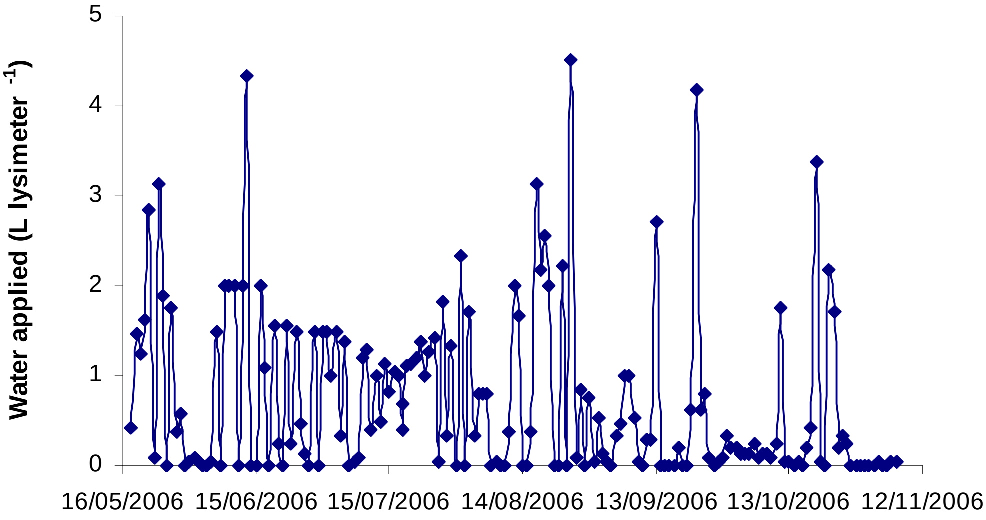


Figure A.3.3.2 The total water (sum of rain and irrigation water) applied daily per lysimeter during the experimental period.

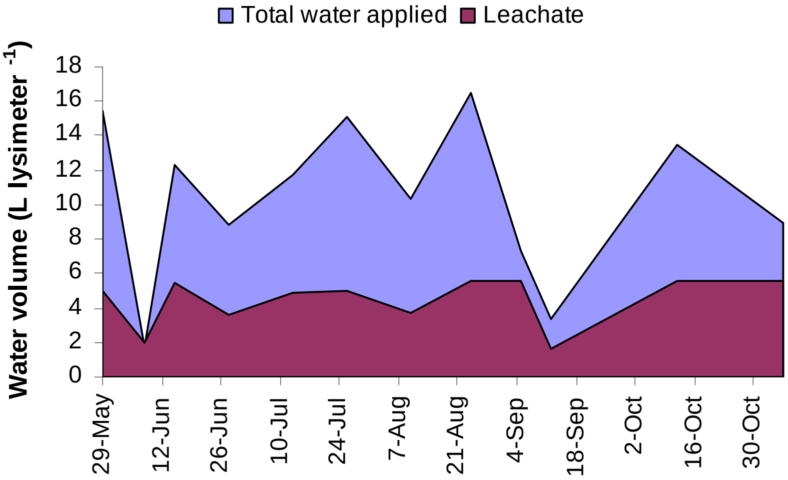


Figure A.3.3.3 The volume of total water applied (sum of rain and irrigation water) and the resulted volume of leachate (average of all treatments) per lysimeter between each sampling date.

Table A.3.3.2 Weather data in a monthly basis during the lysimeter experiment – mean values.

	Tmax (°C)	**Tmin** (°C)	**RH** (%)	**Sun** (hr)	**Solar** (MJ/m^2)	**Net** (MJ/m^2)	**Wind** (km/d)	**ET** (mm/d)	**Rain** (mm/d)
May-06	16.70	8.72	88.81	3.17	12.86	8.34	164.36	2.05	2.83
Jun-06	21.73	11.29	78.80	5.95	17.87	11.23	92.34	3.14	0.53
Jul-06	26.64	14.45	77.00	5.99	17.20	10.73	120.91	3.55	0.77
Aug-06	20.52	12.56	85.58	2.55	10.85	6.95	135.66	2.10	3.20
Sep-06	21.78	13.45	91.03	3.66	9.51	5.36	140.09	1.68	1.50
Oct-06	16.75	10.22	97.55	1.92	5.04	2.50	147.63	0.67	1.65
Nov-06	11.78	5.61	95.73	1.52	2.95	0.83	199.56	0.33	2.98

Table A.3.3.3 The water balance between the water inputs of rain and irrigation, and the output of evapotranspiration on a monthly basis.

	Rain (mm)	**Irrigation** (mm)	**ET** (mm)	**Balance** (mm)
May-06	87.8	0.0	63.4	24.40
Jun-06	16.0	92.6	94.2	14.43
Jul-06	23.8	101.3	109.9	15.23
Aug-06	99.2	18.0	65.0	52.16
Sep-06	45.0	14.0	50.4	8.64
Oct-06	51.2	0.0	20.8	30.40
Nov-06	89.4	0.0	9.8	79.60

Figure A.3.3.4 shows the lysimeter trial on site and Figure A.3.3.5 presents the leachate collection system. Effects of mineral fertilization and compost application on forage maize development are presented in Figure A.3.3.6.

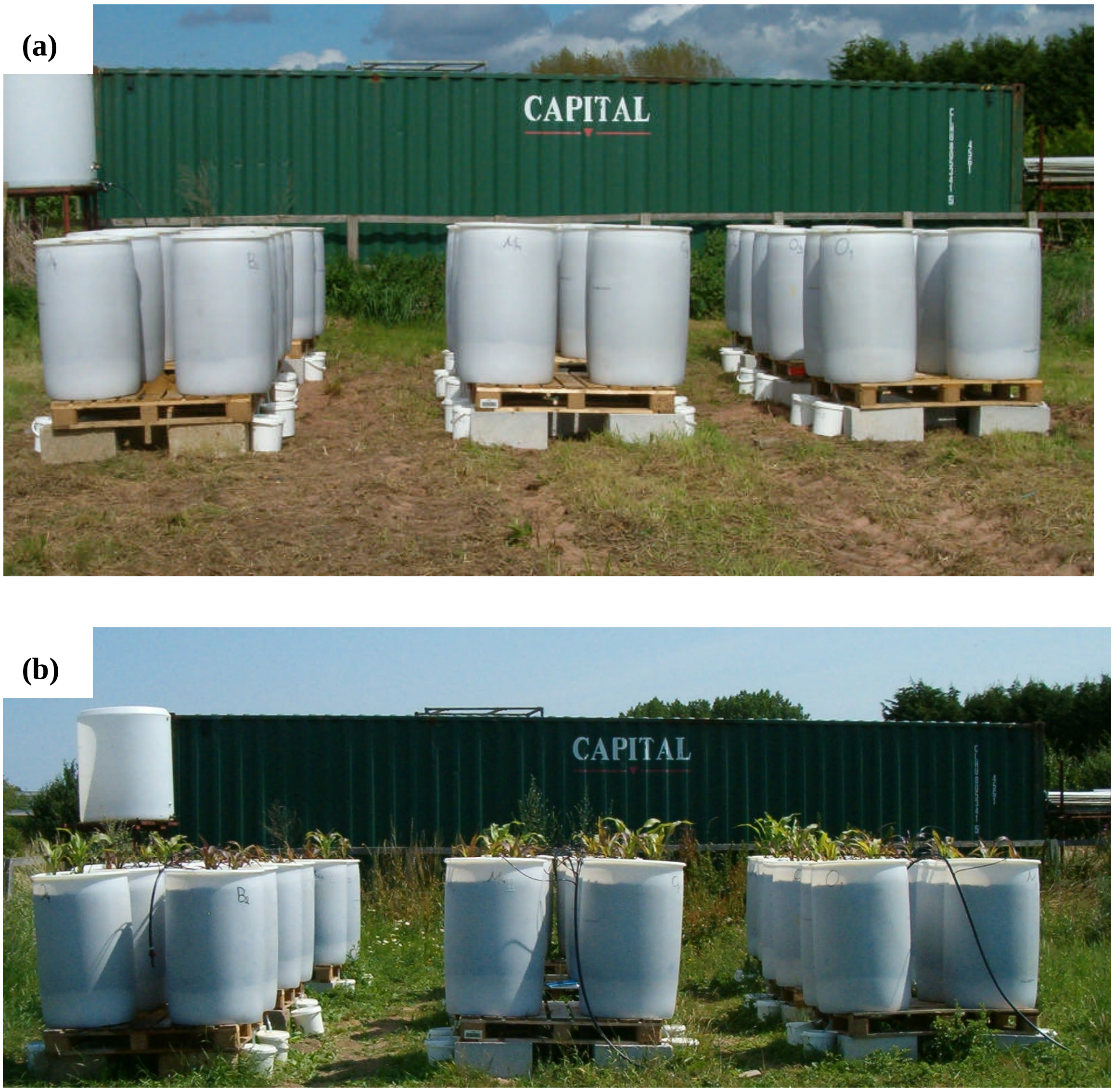


Figure A.3.3.4 View of the lysimeter trial on site: (a) before crop sowing (May 18) and (b) 5 weeks after crop emergence (July 12).

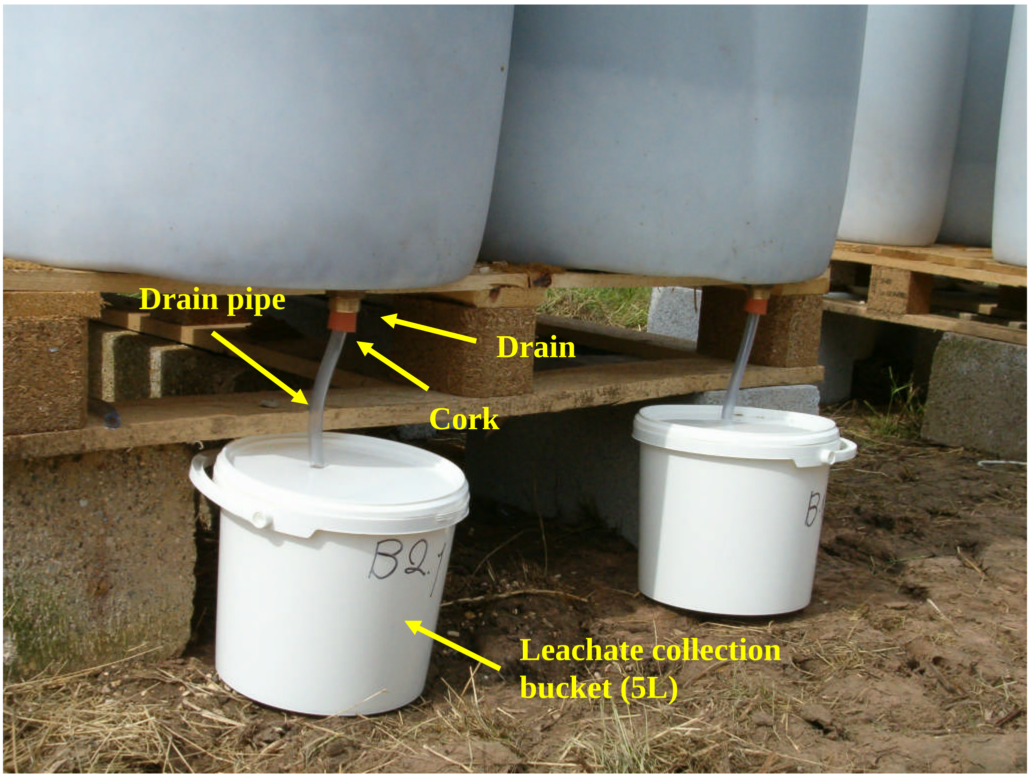


Figure A.3.3.5 Leachate collection system. Buckets were placed on the side of the lysimeters for the collection of the leachate. A plastic tube was attached in the drain by a cork and it ended in the bucket through a hole on the bucket lid.

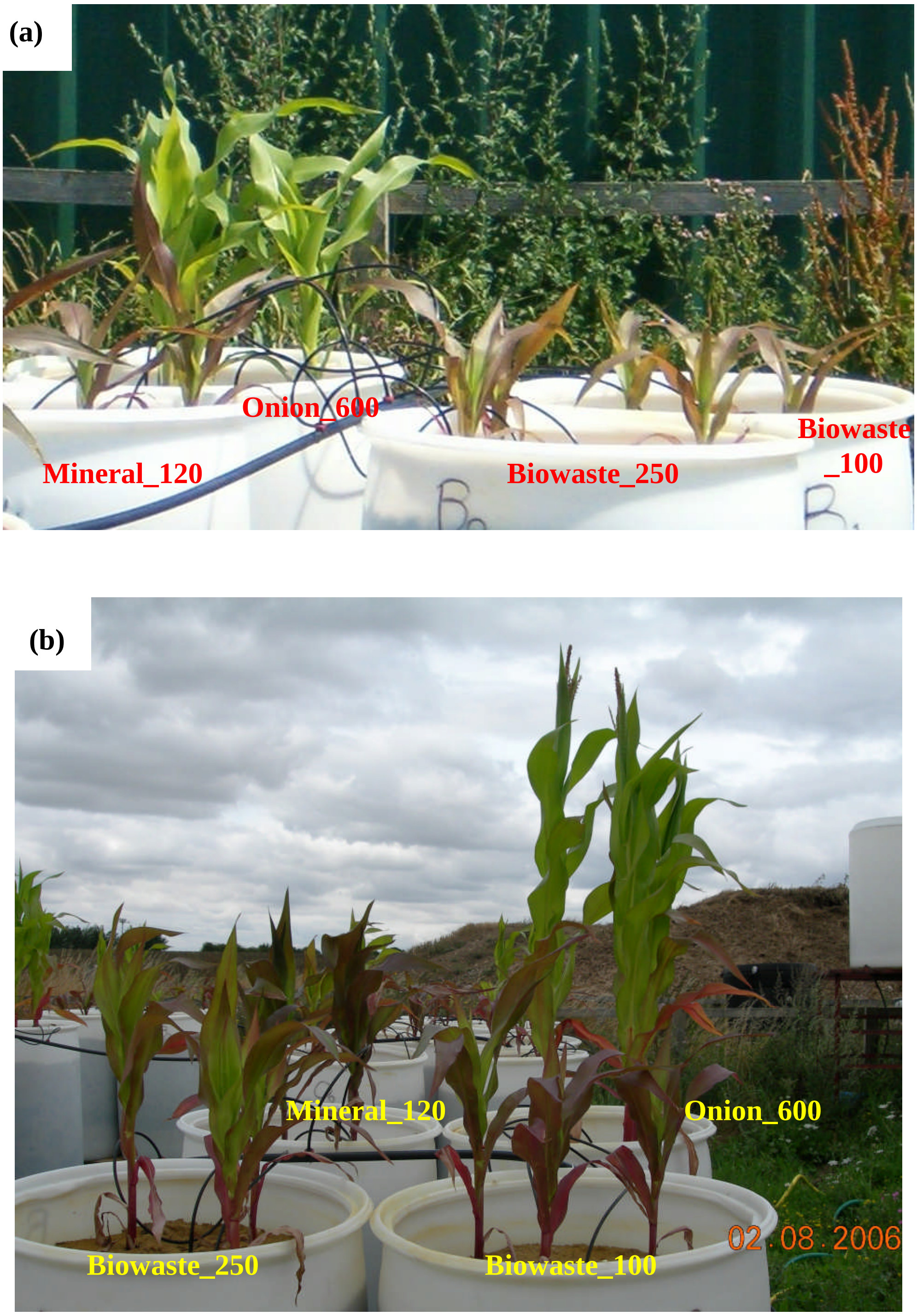


Figure A.3.3.6 Forage maize production resulted from mineral fertilized (120 kg N ha^{-1}), biowaste compost (100 and 250 kg total N ha^{-1}) and onion compost (600 kg total N ha^{-1}) amended sand (a) 5 weeks after crop emergence, before supplementary fertilization and (b) 8 weeks after crop emergence, 2 weeks after supplementary P fertilization.

A.4 Tables of means

This part of the Appendices presents the Tables of means of all soil, plant and leachate properties which they were not significantly affected by compost or mineral fertilizer application and were not presented in the main thesis.

Table A.4.1 Fluctuations of soil NH_4-N content (0-0.6 m depths) during the period of September 2005 to October 2006 in the field experiment (M: mineral fertilizer, B: biowaste compost, O: onion compost; I: incorporated, S: surface applied, 0, 40, 80: kg N ha^{-1}; a, b, c: 80, 165, 250 kg total N ha^{-1} in 2005, or 100, 250, 400 kg total N ha^{-1} in 2006).

	NH_4-N (kg ha^{-1})			
	Sep-05	Mar-06	Jul-06	Oct-06
M 0	16.8	1.2	0.0	0.7
M 40	-	0.3	1.0	0.0
M 80	22.0	0.1	0.0	0.0
BI a	11.7	2.1	0.8	1.2
BI b	0.0	0.1	0.0	0.0
BI c	15.7	0.1	0.0	0.0
BS a	2.5	0.3	0.8	0.5
BS b	0.4	1.6	2.5	0.0
BS c	6.0	4.0	0.0	3.2
OI a	11.4	0.4	0.0	0.0
OI b	5.1	0.6	0.0	1.6
OI c	0.5	1.2	0.8	0.4

Table A.4.2 Field experiment: Residual soil properties (0-0.3 m depths) in 2005 (M: mineral fertilizer, B: biowaste compost, O: onion compost; I: incorporated, S: surface applied, 0, 40, 80: kg N ha^{-1}; a, b, c: 80, 165, 250 kg total N ha^{-1}).

	K (kg/ha)	**P** (kg/ha)	**pH**	**OM** (%)	**EC_e** (dS/m)	**Total N** (%)	**Total C** (%)	**C: N**	**Na** (mg/L)
M 0	999	277	6.6	4.24	0.5	0.15	1.50	9.9	12.42
M 40	-	-	-	-	-	-	-	-	-
M 80	1036	292	6.7	4.41	0.6	0.16	1.56	9.9	12.84
BI a	1122	312	6.8	4.33	0.5	0.14	1.38	9.9	15.46
BI b	1013	290	6.7	4.56	0.5	0.15	1.53	10.0	13.43
BI c	1128	299	6.8	4.43	0.6	0.15	1.49	9.8	18.71
BS a	1043	275	6.7	4.22	0.4	0.14	1.44	9.9	13.97
BS b	1283	276	6.7	4.65	0.5	0.16	1.62	10.1	17.70
BS c	1082	287	6.7	4.19	0.5	0.16	1.58	10.0	17.17
OI a	1105	274	6.6	4.40	0.6	0.15	1.49	10.0	13.46
OI b	1074	279	6.8	4.12	0.5	0.13	1.31	10.1	14.06
OI c	1145	273	6.7	4.10	0.5	0.15	1.46	9.9	14.40

Table A.4.3 Field experiment: Residual soil properties (0-0.3 m depths) in 2006 (M: mineral fertilizer, B: biowaste compost, O: onion compost; I: incorporated, S: surface applied, 0, 40, 80: kg N ha^{-1}; a, b, c: 100, 250, 400 kg total N ha^{-1}).

	OM (%)	**P** (mg/kg)	**pH**	**C: N**	**EC$_e$** (dS/m)	**Total N** (%)	**Total C** (%)
M 0	3.70	83.9	7.1	10.0	0.6	0.14	1.43
M 40	3.61	83.5	7.0	9.9	-	0.15	1.43
M 80	3.70	84.9	7.0	9.9	0.9	0.14	1.38
BI a	3.57	87.8	7.2	10.0	-	0.13	1.31
BI b	3.88	89.4	7.1	10.1	-	0.15	1.52
BI c	3.85	89.1	7.2	10.1	1.0	0.15	1.54
BS a	3.68	79.2	7.1	10.0	-	0.13	1.34
BS b	3.80	81.3	7.1	9.8	-	0.16	1.54
BS c	3.97	89.3	7.1	10.6	1.0	0.15	1.64
OI a	3.85	82.2	7.1	10.0	-	0.15	1.49
OI b	3.42	100.6	7.1	9.8	-	0.14	1.32
OI c	3.75	89.7	7.1	9.9	1.4	0.15	1.46

Table A.4.4 Pot experiment: Residual soil properties in 2005 (M: mineral fertilizer, B: biowaste compost, O: onion compost; I: incorporated, S: surface applied, 0, 40, 80: kg N ha^{-1}; a, b, c: 80, 165, 250 kg total N ha^{-1}).

	C: N	**P** (mg/kg)	**pH**	**EC$_e$** (dS/m)	**TON-N** (mg/kg)	**NH$_4$-N** (mg/kg)	**ESP** (%)
M 0	10.3	55.94	6.7	1.8	0.49	1.63	2.3
M 40	-	-	-	-	-	-	-
M 80	10.4	56.34	6.5	1.8	0.14	1.30	2.6
BI a	10.4	58.81	6.6	1.7	0.71	1.80	2.4
BI b	10.3	57.77	6.7	2.0	2.27	1.58	2.7
BI c	10.2	55.77	6.7	1.6	0.39	1.07	2.4
BS a	10.3	56.4	6.7	1.7	0.68	1.72	2.3
BS b	10.3	57.43	6.7	1.5	0.75	1.46	2.1
BS c	10.5	53.82	6.7	1.6	0.59	1.26	2.2
OI a	10.4	54.39	6.7	1.8	1.07	1.74	2.4
OI b	10.4	58.17	6.7	1.8	0.85	1.68	2.4
OI c	10.3	58.40	6.7	2.0	1.04	1.79	2.6
OS a	10.4	57.37	6.7	1.7	0.86	1.09	2.0
OS b	10.4	55.54	6.7	1.7	1.01	1.18	1.9
OS c	10.2	56.17	6.6	1.8	0.79	2.05	2.0

Table A.4.5 Lysimeter experiment: Residual soil properties in 2006 (M: mineral fertilizer, B: biowaste compost, O: onion compost; 0-600: kg N ha^{-1}).

	Total P (mg/kg)	**P** (mg/kg)	**pH**	**OM** (%)	**EC$_e$** (dS/m)	**ESP** (%)
M 0	108.1	3.3	7.3	1.9	0.4	0.4
M 40	109.0	3.5	7.6	2.2	-	-
M 80	85.2	4.1	7.4	2.3	-	-
M 120	92.0	3.3	7.7	2.1	0.5	0.5
B 100	92.7	4.3	7.7	2.0	-	-
B 250	79.6	5.7	7.7	2.0	-	-
B 400	80.0	4.3	7.4	2.2	-	-
B 600	82.1	7.2	7.5	2.1	0.6	0.4
O 100	95.1	9.2	7.3	1.9	-	-
O 250	93.4	14.4	7.3	2.4	-	-
O 400	95.6	12.4	7.3	2.1	-	-
O 600	91.3	28.3	7.5	2.1	0.6	0.2

A.5 ANOVA: F probability tables

A.5.1 Compost analysis

A.5.1.1 Biowaste, onion, garden and MBT compost properties in 2005

Analysis of unbalanced designs using GenStat regression

	d.f.	s.s.	m.s.	v.r.	F pr.
Total N (%)					
+ Compost_type	3	8.51579	2.83860	36.98	<.001
Residual	86	6.60194	0.07677		
Total	89	15.11774	0.16986		
Total C (%)					
+ Compost_type	3	1775.34	591.78	42.11	<.001
Residual	86	1208.53	14.05		
Total	89	2983.87	33.53		
C: N ratio					
+ Compost_type	3	22.63	7.54	0.47	0.702
Residual	86	1370.35	15.93		
Total	89	1392.98	15.65		
DM (%)					
+ Compost_type	3	9116.56	3038.85	140.72	<.001
Residual	42	906.97	21.59		
Total	45	10023.53	222.75		
NH_4-N (mg/kg)					
+ Compost_type	3	194633.	64878.	0.89	0.461
Residual	22	1600469.	72749.		
Total	25	1795103.	71804.		
TON-N (mg/kg)					
+ Compost_type	3	28316.	9439.	0.97	0.425
Residual	22	214416.	9746.		
Total	25	242732.	9709.		
PO_4-P (mg/kg)					
+ Compost_type	3	159982.	53327.	2.74	0.068
Residual	22	428604.	19482.		
Total	25	588586.	23543.		
K (g/kg)					
+ Compost_type	3	188.733	62.911	14.07	<.001
Residual	22	98.382	4.472		
Total	25	287.115	11.485		
OM (%)					
+ Compost_type	3	901.03	300.34	8.12	<.001
Residual	22	813.37	36.97		
Total	25	1714.40	68.58		
bulk density (g/l)					
+ Compost_type	3	0.046402	0.015467	3.06	0.053
Residual	19	0.096164	0.005061		
Total	22	0.142565	0.006480		

	d.f.	s.s.	m.s.	v.r.	F pr.
pH					
+ Compost_type	3	18.356	6.1187	7.87	<.001
Residual	22	17.1102	0.7777		
Total	25	35.4662	1.4186		
EC_e (dS/m)					
Compost_type	3	9.43396	3.14465	206.32	<.001
Residual	8	0.12193	0.01524		
Total	11	9.55589			
Na (mg/l)					
Compost_type	3	1932423.9	644141.3	3971.11	<.001
Residual	8	1297.7	162.2		
Total	11	1933721.6			
EC (dS/m)					
+ Compost_type	1	0.01939	0.01939	1.82	0.211
Residual	9	0.09615	0.01068		
Total	10	0.11554	0.01155		
Na (mg/kg) - BS EN					
+ Compost_type	1	635941	635941	84.28	<.001
Residual	9	67907	7545		
Total	10	703848	70385		

A.5.1.2 Biowaste and onion compost properties in 2006

	d.f.	s.s.	m.s.	v.r.	F pr.
Total N (%)					
Compost	1	3.68294	3.68294	172.08	<.001
Residual	16	0.34245	0.0214		
Total	17	4.02539			
Total C (%)					
Compost	1	3334.186	3334.186	1660.77	<.001
Residual	16	32.122	2.008		
Total	17	3366.308			
C: N ratio					
Compost	1	416.068	416.068	118.62	<.001
Residual	16	56.122	3.508		
Total	17	472.19			
DM (%)					
Compost	1	947.893	947.893	176.39	<.001
Residual	8	42.99	5.374		
Total	9	990.883			
K (g/kg)					
Compost	1	8.6849	8.6849	32.03	<.001
Residual	8	2.1689	0.2711		
Total	9	10.8538			

	d.f.	s.s.	m.s.	v.r.	F pr.
NH_4-N (mg/kg)					
Compost	1	93362.2	93362.2	277.37	<.001
Residual	8	2692.8	336.6		
Total	9	96054.9			
TON-N (mg/kg)					
Compost	1	81720	81720	57.18	<.001
Residual	8	11433	1429		
Total	9	93153			
PO_4-P (mg/kg)					
Compost	1	63271	63271	1.87	0.208
Residual	8	270206	33776		
Total	9	333477			
pH					
Compost	1	0.225	0.225	15	0.005
Residual	8	0.12	0.015		
Total	9	0.345			
OM (%)					
Compost	1	12511.6	12511.6	2075.09	<.001
Residual	16	96.471	6.029		
Total	17	12608.071			
bulk density (g/l)					
Compost	1	0.5334083	0.5334083	1189.76	<.001
Residual	10	0.0044833	0.0004483		
Total	11	0.5378917			
EC (dS/m)					
Compost	1	0.484	0.484	322.67	<.001
Residual	8	0.012	0.0015		
Total	9	0.496			
Na (g/kg)					
Compost	1	5.17502	5.17502	162.28	<.001
Residual	8	0.25511	0.03189		
Total	9	5.43013			

A.5.2 Field experiment

A.5.2.1 Forage maize growth in 2005: plant height

Plant height (cm)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	91.83	45.92	1.72	
Block.Date stratum					
Date	3	722115.66	240705.22	9030.69	<.001
Residual	6	159.92	26.65	0.55	
Block.Date.*Units* stratum					
Controls	2	77.22	38.61	0.80	0.453
Date.Controls	6	116.49	19.42	0.40	0.875
Controls.Type_Method	2	33.00	16.50	0.34	0.711
Controls.Rate	2	94.25	47.13	0.98	0.381
Date.Controls.Type_Method	6	155.64	25.94	0.54	0.778
Date.Controls.Rate	6	185.51	30.92	0.64	0.697
Controls.Type_Method.Rate	4	872.02	218.01	4.52	0.002
Date.Controls.Type_Method.Rate	12	1064.64	88.72	1.84	0.055
Residual	80	3858.03	48.23		
Total	131	728824.22			

A.5.2.2 Crop DM yield, N, P and K uptake in 2005

DM (Mg ha^{-1})

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	56.315	28.157	4.23	
Block.*Units* stratum					
Control	2	3.950	1.975	0.30	0.746
Control.Type	2	9.557	4.779	0.72	0.500
Control.Rate	2	0.076	0.038	0.01	0.994
Control.Type.Rate	4	26.710	6.678	1.00	0.429
Residual	20	133.000	6.650		
Total	32	229.609			

N uptake (kg ha^{-1})

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	23862.	11931.	6.66	
Block.*Units* stratum					
Control	2	1722.	861.	0.48	0.625
Control.Type	2	1813.	907.	0.51	0.610
Control.Rate	2	542.	271.	0.15	0.861
Control.Type.Rate	4	6039.	1510.	0.84	0.515
Residual	20	35843.	1792.		
Total	32	69821.			

K uptake (kg ha^{-1})

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	4325.1	2162.6	3.21	
Block.*Units* stratum					
Contols	2	1752.1	876.0	1.30	0.295
Contols.Type_Method	2	121.3	60.7	0.09	0.914
Contols.Application_rate	2	83.0	41.5	0.06	0.941
Contols.Type_Method.Application_rate	4	298.9	74.7	0.11	0.977
Residual	20	13490.6	674.5		
Total	32	20071.0			

P uptake (kg ha^{-1})

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	323.74	161.87	3.88	
Block.*Units* stratum					
Contols	2	111.25	55.63	1.33	0.286
Contols.Type_Method	2	10.19	5.09	0.12	0.886
Contols.Application_rate	2	64.84	32.42	0.78	0.473
Contols.Type_Method.Application_rate	4	328.99	82.25	1.97	0.138
Residual	20	834.43	41.72		
Total	32	1673.44			

A.5.2.3 Crop DM yield and N uptake in 2006

DM yield (Mg ha^{-1})

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	4.874	2.437	0.72	
Block.*Units* stratum					
Control	3	14.030	4.677	1.38	0.274
Control.Type	2	6.805	3.403	1.01	0.381
Control.Rate	2	14.743	7.371	2.18	0.137
Control.Type.Rate	4	10.477	2.619	0.78	0.553
Residual	22	74.326	3.378		
Total	35	125.256			

N uptake (kg ha^{-1})

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	5196.	2598.	2.40	
Block.*Units* stratum					
Control	3	5685.	1895.	1.75	0.186
Control.Type	2	959.	480.	0.44	0.647
Control.Rate	2	4806.	2403.	2.22	0.132
Control.Type.Rate	4	2154.	538.	0.50	0.737
Residual	22	23793.	1082.		
Total	35	42593.			

A.5.2.4 Soil mineral nitrogen (TON-N and NH_4-N)

A.5.2.4.1 September 2005

TON-N (kg/ha) 0-0.6 m

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	162.42	81.21	2.18	
Block.*Units* stratum					
Control	2	121.37	60.68	1.63	0.221
Control.Type	2	48.79	24.39	0.65	0.530
Control.Rate	2	38.55	19.28	0.52	0.604
Control.Type.Rate	4	222.52	55.63	1.49	0.242
Residual	20	745.43	37.27		
Total	32	1339.07			

NH_4-N (kg/ha) 0-0.6 m

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	1210.4	605.2	3.07	
Block.*Units* stratum					
Control	2	943.9	472.0	2.39	0.117
Control.Type	2	169.5	84.7	0.43	0.656
Control.Rate	2	215.7	107.8	0.55	0.587
Control.Type.Rate	4	398.3	99.6	0.51	0.732
Residual	20	3942.2	197.1		
Total	32	6880.0			

A.5.2.4.2 March 2006

TON-N (kg/ha) 0-0.6 m

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	97.70	48.85	3.02	
Block.*Units* stratum					
Control	3	97.22	32.41	2.01	0.143
Control.Type	2	70.10	35.05	2.17	0.138
Control.Rate	2	236.54	118.27	7.32	0.004
Control.Type.Rate	4	96.61	24.15	1.49	0.238
Residual	22	355.52	16.16		
Total	35	953.69			

***Sqrt* [NH_4-N (kg/ha) 0-0.6 m]**

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.2826	0.1413	0.42	
Block.*Units* stratum					
Control	3	0.9581	0.3194	0.94	0.439
Control.Type	2	1.5494	0.7747	2.28	0.126
Control.Rate	2	0.4718	0.2359	0.69	0.510
Control.Type.Rate	4	6.6759	1.6690	4.91	0.006
Residual	22	7.4826	0.3401		
Total	35	17.4204			

A.5.2.4.3 July 2006

TON-N (0-0.6 m)

Source of variation	d.f.	(m.v.)	s.s.	m.s.	v.r.	F pr.
Block stratum	2		37233.	18617.	12.21	
Block.*Units* stratum						
Control	3		4118.	1373.	0.90	0.458
Control.Type	2		12921.	6461.	4.24	0.028
Control.Rate	2		5627.	2814.	1.84	0.183
Control.Type.Rate	4		5000.	1250.	0.82	0.527
Residual	21	(1)	32032.	1525.		
Total	34	(1)	96293.			

NH_4-N (kg/ha) 0-0.6 m

Source of variation	d.f.	(m.v.)	s.s.	m.s.	v.r.	F pr.
Block stratum	2		9.587	4.793	2.45	
Block.*Units* stratum						
Control	3		2.271	0.757	0.39	0.764
Control.Type	2		4.003	2.002	1.02	0.377
Control.Rate	2		1.380	0.690	0.35	0.707
Control.Type.Rate	4		10.518	2.629	1.34	0.287
Residual	21	(1)	41.101	1.957		
Total	34	(1)	68.855			

A.5.2.4.4 October 2006

***ln* [TON-N (kg/ha) 0-0.6 m]**

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	3.2559	1.6279	3.22	
Block.*Units* stratum					
Control	3	12.9109	4.3036	8.51	<.001
Control.Type	2	6.4047	3.2024	6.34	0.007
Control.Rate	2	0.2953	0.1476	0.29	0.750
Control.Type.Rate	4	1.2496	0.3124	0.62	0.654
Residual	22	11.1208	0.5055		
Total	35	35.2372			

NH_4-N (kg/ha) 0-0.6 m

Source of variation	d.f.	(m.v.)	s.s.	m.s.	v.r.	F pr.
Block stratum	2		0.306	0.153	0.07	
Block.*Units* stratum						
Control	3		2.881	0.960	0.44	0.728
Control.Type	2		3.421	1.710	0.78	0.470
Control.Rate	2		2.571	1.286	0.59	0.565
Control.Type.Rate	4		22.517	5.629	2.57	0.068
Residual	21	(1)	45.948	2.188		
Total	34	(1)	77.550			

A.5.2.5 Other Soil properties in 2005 and 2006

A.5.2.5.1 In 2005: Soil OM, total C, total N, C: N, extractable P, K, Na, pH and EC_e

K (kg/ha)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	33286.	16643.	1.88	
Block.*Units* stratum					
Control	2	44821.	22411.	2.53	0.105
Control.Type	2	10621.	5311.	0.60	0.559
Control.Rate	2	5803.	2901.	0.33	0.724
Control.Type.Rate	4	126171.	31543.	3.56	0.024
Residual	20	177094.	8855.		
Total	32	397797.			

P (kg/ha)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	298.4	149.2	0.23	
Block.*Units* stratum					
Control	2	328.9	164.4	0.25	0.779
Control.Type	2	3238.0	1619.0	2.49	0.108
Control.Rate	2	166.2	83.1	0.13	0.881
Control.Type.Rate	4	895.6	223.9	0.34	0.845
Residual	20	13017.6	650.9		
Total	32	17944.6			

pH

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.04424	0.02212	1.53	
Block.*Units* stratum					
Control	2	0.02391	0.01195	0.83	0.452
Control.Type_Method	2	0.02741	0.01370	0.95	0.404
Control.Application_rate	2	0.00963	0.00481	0.33	0.721
Control.Type_Method.Application_rate	4	0.02815	0.00704	0.49	0.745
Residual	20	0.28909	0.01445		
Total	32	0.42242			

Organic matter (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.1007	0.0503	0.40	
Block.*Units* stratum					
Control	2	0.0426	0.0213	0.17	0.845
Control.Type_Method	2	0.2502	0.1251	1.00	0.386
Control.Application_rate	2	0.1871	0.0936	0.75	0.486
Control.Type_Method.Application_rate	4	0.4573	0.1143	0.91	0.476
Residual	20	2.5045	0.1252		
Total	32	3.5424			

EC_e (dS/m)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.046309	0.023155	2.80	
Block.*Units* stratum					
Control	2	0.013881	0.006940	0.84	0.446
Control.Type_Method	2	0.011482	0.005741	0.69	0.511
Control.Application_rate	2	0.009527	0.004763	0.58	0.571
Control.Type_Method.Application_rate	4	0.042033	0.010508	1.27	0.314
Residual	20	0.165293	0.008265		
Total	32	0.288525			

Total N (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.0002856	0.0001428	0.60	
Block.*Units* stratum					
Control	2	0.0003080	0.0001540	0.65	0.535
Control.Type_Method	2	0.0006561	0.0003280	1.37	0.276
Control.Application_rate	2	0.0003159	0.0001580	0.66	0.527
Control.Type_Method.Application_rate	4	0.0010685	0.0002671	1.12	0.375
Residual	20	0.0047728	0.0002386		
Total	32	0.0074069			

Total C (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.01440	0.00720	0.35	
Block.*Units* stratum					
Control	2	0.02024	0.01012	0.49	0.622
Contro.Type_Method	2	0.07297	0.03649	1.75	0.199
Control.Application_rate	2	0.02544	0.01272	0.61	0.552
Control.Type_Method.Application_rate	4	0.12168	0.03042	1.46	0.251
Residual	20	0.41597	0.02080		
Total	32	0.67070			

C: N ratio

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.13268	0.06634	1.37	
Block.*Units* stratum					
Control	2	0.06092	0.03046	0.63	0.543
Control.Type_Method	2	0.04151	0.02076	0.43	0.657
Control.Application_rate	2	0.12300	0.06150	1.27	0.303
Control.Type_Method.Application_rate	4	0.05525	0.01381	0.29	0.884
Residual	20	0.96866	0.04843		
Total	32	1.38202			

Na (mg L^{-1})

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	1.842	0.921	0.11	
Block.*Units* stratum					
Control	2	37.182	18.591	2.30	0.126
Control.Type_Method	2	27.163	13.581	1.68	0.211
Control.Rate	2	28.597	14.299	1.77	0.196
Control.Type_Method.Rate	4	39.637	9.909	1.23	0.330
Residual	20	161.367	8.068		
Total	32	295.788			

A.5.2.5.2 In 2006: Soil OM, total C, total N, C: N, extractable P, K, Na, pH and EC_e

OM (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.1542	0.0771	0.68	
Block.*Units* stratum					
Control	3	0.0629	0.0210	0.18	0.906
Control.Type	2	0.0941	0.0470	0.41	0.667
Control.Rate	2	0.1509	0.0754	0.66	0.526
Control.Type.Rate	4	0.4640	0.1160	1.02	0.420
Residual	22	2.5097	0.1141		
Total	35	3.4358			

K (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	5972.	2986.	2.12	
Block.*Units* stratum					
Control	3	9339.	3113.	2.21	0.115
Control.Type	2	25624.	12812.	9.10	0.001
Control.Rate	2	24635.	12318.	8.74	0.002
Control.Type.Rate	4	2293.	573.	0.41	0.802
Residual	22	30988.	1409.		
Total	35	98852.			

Na (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	41.64	20.82	1.94	
Block.*Units* stratum					
Control	3	944.12	314.71	29.29	<.001
Control.Type	2	529.01	264.51	24.62	<.001
Control.Rate	2	849.63	424.81	39.53	<.001
Control.Type.Rate	4	203.99	51.00	4.75	0.007
Residual	22	236.40	10.75		
Total	35	2804.79			

C: N ratio

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.2473	0.1237	0.95	
Block.*Units* stratum					
Control	3	0.1172	0.0391	0.30	0.824
Control.Type	2	0.2902	0.1451	1.12	0.345
Control.Rate	2	0.4767	0.2384	1.84	0.183
Control.Type.Rate	4	0.6707	0.1677	1.29	0.303
Residual	22	2.8541	0.1297		
Total	35	4.6562			

Total C (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.00783	0.00392	0.13	
Block.*Units* stratum					
Control	3	0.02058	0.00686	0.24	0.871
Control.Type	2	0.03377	0.01689	0.58	0.568
Control.Rate	2	0.12665	0.06332	2.17	0.138
Control.Type.Rate	4	0.16261	0.04065	1.40	0.268
Residual	22	0.64103	0.02914		
Total	35	0.99247			

Total N (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.0002466	0.0001233	0.43	
Block.*Units* stratum					
Control	3	0.0001048	0.0000349	0.12	0.946
Control.Type	2	0.0001260	0.0000630	0.22	0.805
Control.Rate	2	0.0008622	0.0004311	1.50	0.245
Control.Type.Rate	4	0.0012279	0.0003070	1.07	0.396
Residual	22	0.0063223	0.0002874		
Total	35	0.0088898			

P (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	122.82	61.41	1.23	
Block.*Units* stratum					
Control	3	85.95	28.65	0.57	0.638
Control.Type	2	274.96	137.48	2.75	0.086
Control.Rate	2	285.87	142.94	2.86	0.079
Control.Type.Rate	4	404.61	101.15	2.03	0.126
Residual	22	1098.93	49.95		
Total	35	2273.14			

pH

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.01167	0.00583	0.42	
Block.*Units* stratum					
Control	3	0.07343	0.02448	1.75	0.187
Control.Type	2	0.03630	0.01815	1.29	0.294
Control.Rate	2	0.01185	0.00593	0.42	0.660
Control.Type.Rate	4	0.00593	0.00148	0.11	0.979
Residual	22	0.30833	0.01402		
Total	35	0.44750			

EC (dS/m)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.51600	0.25800	5.47	
Block.*Units* stratum					
control	1	0.45067	0.45067	9.55	0.015
control.Treatment	3	0.48000	0.16000	3.39	0.074
Residual	8	0.37733	0.04717		
Total	14	1.82400			

ESP (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.028000	0.014000	2.47	
Block.*Units* stratum					
control	1	0.037500	0.037500	6.62	0.033
control.Treatment	3	0.389167	0.129722	22.89	<.001
Residual	8	0.045333	0.005667		
Total	14	0.500000			

Moisture (%) (v/v)

Source of variation	d.f.	(m.v.)	s.s.	m.s.	v.r.	F pr.
Block stratum	2		203.185	101.592	53.95	
Block.Suction_kPa stratum						
Suction_kPa	7		23365.096	3337.871	1772.70	<.001
Residual	14		26.361	1.883	0.31	
Block.Suction_kPa.*Units* stratum						
Control	1		67.008	67.008	11.07	0.001
Suction_kPa.Control	7		9.783	1.398	0.23	0.977
Control.Type	2		309.231	154.615	25.54	<.001
Control.Rate	2		5.410	2.705	0.45	0.641
Suction_kPa.Control.Type	14		97.565	6.969	1.15	0.320
Suction_kPa.Control.Rate	14		110.983	7.927	1.31	0.210
Control.Type.Rate	4		170.153	42.538	7.03	<.001
Suction_kPa.Control.Type.Rate	28		102.185	3.649	0.60	0.940
Residual	134	(10)	811.285	6.054		
Total	229	(10)	24090.074			

A.5.3 Pot experiment

A.5.3.1 First period (2005)

A.5.3.1.1 Forage maize

No complete leaves

Source of variation	d.f.	(m.v.)	s.s.	m.s.	v.r.	F pr.
Block stratum	2		0.0038	0.0019	0.03	
Block.Date stratum						
Date	6		7088.2415	1181.3736	21079.36	<.001
Residual	12		0.6725	0.0560	0.45	
Block.Date.*Units* stratum						
Control_v_treated	2		0.1729	0.0865	0.70	0.500
Date.Control_v_treated	12		0.7900	0.0658	0.53	0.892
Control_v_treated.Type	1		0.0159	0.0159	0.13	0.721
Control_v_treated.method	1		0.1429	0.1429	1.15	0.285
Control_v_treated.rate	2		0.4985	0.2492	2.01	0.138
Date.Control_v_treated.Type	6		0.9216	0.1536	1.24	0.291
Date.Control_v_treated.method	6		0.5238	0.0873	0.70	0.647
Control_v_treated.Type.method	1		1.5125	1.5125	12.18	<.001
Date.Control_v_treated.rate	12		0.9699	0.0808	0.65	0.795
Control_v_treated.Type.rate	2		0.5154	0.2577	2.08	0.129
Control_v_treated.method.rate	2		2.5663	1.2832	10.34	<.001
Date.Control_v_treated.Type.method	6		0.5390	0.0898	0.72	0.631
Date.Control_v_treated.Type.rate	12		2.0650	0.1721	1.39	0.179
Date.Control_v_treated.method.rate	12		1.8204	0.1517	1.22	0.273
Control_v_treated.Type.method.rate	2		0.1170	0.0585	0.47	0.625
Date.Control_v_treated.Type.method.rate	12		0.9855	0.0821	0.66	0.786
Residual	147	(35)	18.2497	0.1241		
Total	258	(35)	6280.6313			

***ln* [plant height (cm)]**

Source of variation	d.f.	(m.v.)	s.s.	m.s.	v.r.	F pr.
Block stratum	2		0.023037	0.011519	6.19	
Block.Date stratum						
Date	6		299.690097	49.948350	26859.34	<.001
Residual	12		0.022316	0.001860	0.35	
Block.Date.*Units* stratum						
Control_v_treated	2		0.035244	0.017622	3.27	0.041
Date.Control_v_treated	12		0.018149	0.001512	0.28	0.992
Control_v_treated.Type	1		0.002109	0.002109	0.39	0.532
Control_v_treated.method	1		0.005983	0.005983	1.11	0.294
Control_v_treated.rate	2		0.006809	0.003405	0.63	0.533
Date.Control_v_treated.Type	6		0.011859	0.001976	0.37	0.899
Date.Control_v_treated.method	6		0.063747	0.010624	1.97	0.073
Control_v_treated.Type.method	1		0.171165	0.171165	31.79	<.001
Date.Control_v_treated.rate	12		0.074642	0.006220	1.16	0.321
Control_v_treated.Type.rate	2		0.057686	0.028843	5.36	0.006
Control_v_treated.method.rate	2		0.004469	0.002234	0.41	0.661
Date.Control_v_treated.Type.method	6		0.141319	0.023553	4.37	<.001
Date.Control_v_treated.Type.rate	12		0.039060	0.003255	0.60	0.836

Date.Control_v_treated.method.rate	12		0.128674	0.010723	1.99	0.029
Control_v_treated.Type.method.rate	2		0.024010	0.012005	2.23	0.111
Date.Control_v_treated.Type.method.rate	12		0.062610	0.005218	0.97	0.481
Residual	147	(35)	0.791487	0.005384		
Total	258	(35)	265.935433			

Diameter (cm)

Source of variation	d.f.	(m.v.)	s.s.	m.s.	v.r.	F pr.
Block stratum	2		0.021006	0.010503	2.95	
Block.Date stratum						
Date	5		46.646006	9.329201	2623.04	<.001
Residual	10		0.035566	0.003557	0.52	
Block.Date.*Units* stratum						
Control_v_treated	2		0.025227	0.012614	1.85	0.162
Date.Control_v_treated	10		0.018870	0.001887	0.28	0.985
Control_v_treated.Type	1		0.005938	0.005938	0.87	0.353
Control_v_treated.method	1		0.035587	0.035587	5.21	0.024
Control_v_treated.rate	2		0.105200	0.052600	7.70	<.001
Date.Control_v_treated.Type	5		0.034998	0.007000	1.02	0.406
Date.Control_v_treated.method	5		0.052071	0.010414	1.52	0.187
Control_v_treated.Type.method	1		0.035809	0.035809	5.24	0.024
Date.Control_v_treated.rate	10		0.133952	0.013395	1.96	0.043
Control_v_treated.Type.rate	2		0.085089	0.042544	6.23	0.003
Control_v_treated.method.rate	2		0.001590	0.000795	0.12	0.890
Date.Control_v_treated.Type.method	5		0.171715	0.034343	5.03	<.001
Date.Control_v_treated.Type.rate	10		0.065431	0.006543	0.96	0.484
Date.Control_v_treated.method.rate	10		0.036961	0.003696	0.54	0.858
Control_v_treated.Type.method.rate	2		0.131955	0.065978	9.66	<.001
Date.Control_v_treated.Type.method.rate	10		0.082913	0.008291	1.21	0.288
Residual	126	(30)	0.860735	0.006831		
Total	221	(30)	43.077436			

DM (g/pot)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	180.93	90.46	1.41	
Block.*Units* stratum					
Control_v_treated	2	258.92	129.46	2.01	0.154
Control_v_treated.Type	1	176.23	176.23	2.74	0.110
Control_v_treated.rate	2	315.77	157.88	2.45	0.106
Control_v_treated.method	1	338.62	338.62	5.26	0.030
Control_v_treated.Type.rate	2	232.64	116.32	1.81	0.184
Control_v_treated.Type.method	1	33.12	33.12	0.51	0.479
Control_v_treated.rate.method	2	99.94	49.97	0.78	0.470
Control_v_treated.Type.rate.method	2	52.56	26.28	0.41	0.669
Residual	26	1672.30	64.32		
Total	41	3361.01			

K uptake (g/pot)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.11774	0.05887	2.32	
Block.*Units* stratum					
Control_v_treated	2	0.09703	0.04851	1.91	0.168
Control_v_treated.Type	1	0.14304	0.14304	5.63	0.025
Control_v_treated.rate	2	0.13946	0.06973	2.75	0.083
Control_v_treated.method	1	0.10915	0.10915	4.30	0.048
Control_v_treated.Type.rate	2	0.10864	0.05432	2.14	0.138
Control_v_treated.Type.method	1	0.08681	0.08681	3.42	0.076
Control_v_treated.rate.method	2	0.07023	0.03511	1.38	0.269
Control_v_treated.Type.rate.method	2	0.13717	0.06858	2.70	0.086
Residual	26	0.66047	0.02540		
Total	41	1.66973			

N uptake (g/pot)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.077432	0.038716	4.12	
Block.*Units* stratum					
Control_v_treated	2	0.000684	0.000342	0.04	0.964
Control_v_treated.Type	1	0.022356	0.022356	2.38	0.135
Control_v_treated.rate	2	0.000840	0.000420	0.04	0.956
Control_v_treated.method	1	0.018232	0.018232	1.94	0.175
Control_v_treated.Type.rate	2	0.032251	0.016125	1.72	0.200
Control_v_treated.Type.method	1	0.000041	0.000041	0.00	0.948
Control_v_treated.rate.method	2	0.000901	0.000451	0.05	0.953
Control_v_treated.Type.rate.method	2	0.006438	0.003219	0.34	0.713
Residual	26	0.244380	0.009399		
Total	41	0.403556			

P uptake (g/pot)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.0027524	0.0013762	4.12	
Block.*Units* stratum					
Control_v_treated	2	0.0000362	0.0000181	0.05	0.947
Control_v_treated.Type	1	0.0064789	0.0064789	19.37	<.001
Control_v_treated.rate	2	0.0004622	0.0002311	0.69	0.510
Control_v_treated.method	1	0.0000081	0.0000081	0.02	0.877
Control_v_treated.Type.rate	2	0.0001952	0.0000976	0.29	0.749
Control_v_treated.Type.method	1	0.0006072	0.0006072	1.82	0.189
Control_v_treated.rate.method	2	0.0015011	0.0007505	2.24	0.126
Control_v_treated.Type.rate.method	2	0.0002879	0.0001439	0.43	0.655
Residual	26	0.0086950	0.0003344		
Total	41	0.0210241			

A.5.3.1.2 Leachate

Total mineral N (mg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	101.38	50.69	1.03	
Block.*Units* stratum					
C_v_treated	2	272.77	136.39	2.77	0.081
C_v_treated.Type	1	367.54	367.54	7.45	0.011
C_v_treated.rate	2	178.83	89.42	1.81	0.183
C_v_treated.method	1	442.26	442.26	8.97	0.006
C_v_treated.Type.rate	2	555.22	277.61	5.63	0.009
C_v_treated.Type.method	1	665.17	665.17	13.49	0.001
C_v_treated.rate.method	2	81.62	40.81	0.83	0.448
C_v_treated.Type.rate.method	2	8.45	4.23	0.09	0.918
Residual	26	1281.97	49.31		
Total	41	3955.20			

***ln* [Mineral N (mg)]**

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	4.9449	2.4724	4.12	
Block.Date stratum					
Date	11	1493.3004	135.7546	226.07	<.001
Residual	22	13.2108	0.6005	0.97	
Block.Date.*Units* stratum					
C_v_Treated	2	0.3564	0.1782	0.29	0.750
Date.C_v_Treated	22	22.7697	1.0350	1.67	0.031
C_v_Treated.Type	1	6.2445	6.2445	10.10	0.002
C_v_Treated.Rate	2	5.6460	2.8230	4.57	0.011
C_v_Treated.Method	1	5.3338	5.3338	8.63	0.004
Date.C_v_Treated.Type	11	13.2729	1.2066	1.95	0.033
Date.C_v_Treated.Rate	22	23.8417	1.0837	1.75	0.021
C_v_Treated.Type.Rate	2	9.8874	4.9437	8.00	<.001
Date.C_v_Treated.Method	11	14.9423	1.3584	2.20	0.014
C_v_Treated.Type.Method	1	9.3979	9.3979	15.21	<.001
C_v_Treated.Rate.Method	2	4.7473	2.3736	3.84	0.022
Date.C_v_Treated.Type.Rate	22	9.5836	0.4356	0.70	0.835
Date.C_v_Treated.Type.Method	11	15.2398	1.3854	2.24	0.012
Date.C_v_Treated.Rate.Method	22	8.2961	0.3771	0.61	0.916
C_v_Treated.Type.Rate.Method	2	1.7422	0.8711	1.41	0.246
Date.C_v_Treated.Type.Rate.Method	22	10.0435	0.4565	0.74	0.798
Residual	312	192.8118	0.6180		
Total	503	1865.6132			

A.5.3.1.3 Soil

pH

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.035576	0.017788	1.98	
Block.*Units* stratum					
Control	2	0.080048	0.040024	4.46	0.022
Control.Type	1	0.001225	0.001225	0.14	0.715
Control. rate	2	0.009739	0.004869	0.54	0.588
Control.Application_method	1	0.001003	0.001003	0.11	0.741
Control.Type. rate	2	0.007817	0.003908	0.44	0.652
Control.Type.Application_method	1	0.009025	0.009025	1.00	0.325
Control.rate.Application_method	2	0.022172	0.011086	1.23	0.307
Control.Type.rate.Application_method	2	0.012717	0.006358	0.71	0.502
Residual	26	0.233490	0.008980		
Total	41	0.412812			

Total N (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.00004444	0.00002222	0.26	
Block.*Units* stratum					
Control	2	0.00032064	0.00016032	1.88	0.173
Control.Type	1	0.00002210	0.00002210	0.26	0.615
Control. rate	2	0.00039202	0.00019601	2.30	0.121
Control.Application_method	1	0.00050166	0.00050166	5.88	0.023
Control.Type. rate	2	0.00012312	0.00006156	0.72	0.496
Control.Type.Application_method	1	0.00000437	0.00000437	0.05	0.823
Control. rate.Application_method	2	0.00009882	0.00004941	0.58	0.568
Control.Type.rate.Application_method	2	0.00049200	0.00024600	2.88	0.074
Residual	26	0.00221974	0.00008537		
Total	41	0.00421890			

Total C (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.00211	0.00106	0.10	
Block.*Units* stratum					
Control	2	0.03372	0.01686	1.60	0.221
Control.Type	1	0.00258	0.00258	0.24	0.625
Control.rate	2	0.03187	0.01594	1.51	0.239
Control.Application_method	1	0.04668	0.04668	4.43	0.045
Control.Type. rate	2	0.01469	0.00734	0.70	0.508
Control.Type.Application_method	1	0.00212	0.00212	0.20	0.658
Control.rate.Application_method	2	0.00801	0.00400	0.38	0.688
Control.Type.rate.Application_method	2	0.07240	0.03620	3.43	0.048
Residual	26	0.27421	0.01055		
Total	41	0.48839			

TON-N (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	2.9749	1.4874	3.71	
Block.*Units* stratum					
Control	2	2.0646	1.0323	2.57	0.096
Control.Type	1	0.0118	0.0118	0.03	0.865
Control. rate	2	1.7347	0.8673	2.16	0.136
Control.Application_method	1	0.6628	0.6628	1.65	0.210
Control.Type. rate	2	1.7573	0.8787	2.19	0.132
Control.Type.Application_method	1	0.2745	0.2745	0.68	0.416
Control. rate.Application_method	2	0.7482	0.3741	0.93	0.407
Control.Type. rate.Application_method	2	2.0039	1.0019	2.50	0.102
Residual	26	10.4378	0.4015		
Total	41	22.6706			

OM (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.51677	0.25839	3.72	
Block.*Units* stratum					
Control	2	0.56966	0.28483	4.10	0.028
Control.Type	1	0.00624	0.00624	0.09	0.767
Control.rate	2	0.49198	0.24599	3.54	0.044
Control.Application_method	1	0.50443	0.50443	7.27	0.012
Control.Type.rate	2	0.20287	0.10144	1.46	0.250
Control.Type.Application_method	1	0.00105	0.00105	0.02	0.903
Control.rate.Application_method	2	0.02507	0.01253	0.18	0.836
Control.Type.rate.Application_method	2	0.06784	0.03392	0.49	0.619
Residual	26	1.80526	0.06943		
Total	41	4.19116			

NH_4-N (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.6484	0.3242	1.84	
Block.*Units* stratum					
Control	2	0.1943	0.0972	0.55	0.583
Control.Type	1	0.1019	0.1019	0.58	0.454
Control.rate	2	0.0746	0.0373	0.21	0.811
Control.Application_method	1	0.2077	0.2077	1.18	0.288
Control.Type. rate	2	2.0027	1.0013	5.67	0.009
Control.Type.Application_method	1	0.1898	0.1898	1.08	0.309
Control. rate.Application_method	2	0.6386	0.3193	1.81	0.184
Control.Type. rate.Application_method	2	0.1665	0.0833	0.47	0.629
Residual	26	4.5877	0.1765		
Total	41	8.8122			

K (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	500.69	250.35	4.83	
Block.*Units* stratum					
Control	2	1603.55	801.78	15.46	<.001
Control.Type	1	3205.43	3205.43	61.80	<.001
Control. rate	2	940.40	470.20	9.07	0.001
Control.Application_method	1	1350.16	1350.16	26.03	<.001
Control.Type. rate	2	263.47	131.74	2.54	0.098
Control.Type.Application_method	1	168.85	168.85	3.26	0.083
Control. rate.Application_method	2	88.83	44.41	0.86	0.436
Control.Type. rate.Application_method	2	274.86	137.43	2.65	0.090
Residual	26	1348.61	51.87		
Total	41	9744.86			

ESP (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.37452	0.18726	3.15	
Block.*Units* stratum					
Control	2	0.18742	0.09371	1.58	0.226
Control.Type	1	0.15638	0.15638	2.63	0.117
Control. rate	2	0.00166	0.00083	0.01	0.986
Control.Application_method	1	1.43616	1.43616	24.14	<.001
Control.Type. rate	2	0.06579	0.03290	0.55	0.582
Control.Type.Application_method	1	0.04804	0.04804	0.81	0.377
Control. rate.Application_method	2	0.21463	0.10732	1.80	0.185
Control.Type. rate.Application_method	2	0.11936	0.05968	1.00	0.380
Residual	26	1.54667	0.05949		
Total	41	4.15063			

EC_e (dS/m)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.52825	0.26412	3.86	
Block.*Units* stratum					
Control	2	0.04958	0.02479	0.36	0.699
Control.Type	1	0.11845	0.11845	1.73	0.200
Control.rate	2	0.02270	0.01135	0.17	0.848
Control.Application_method	1	0.14150	0.14150	2.07	0.162
Control.Type.rate	2	0.15422	0.07711	1.13	0.339
Control.Type.Application_method	1	0.00060	0.00060	0.01	0.926
Control.rate.Application_method	2	0.12034	0.06017	0.88	0.427
Control.Type.rate.Application_method	2	0.16810	0.08405	1.23	0.309
Residual	26	1.77819	0.06839		
Total	41	3.08192			

Moisture (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	3.5194	1.7597	2.51	
Block.*Units* stratum					
Control	2	5.7707	2.8853	4.12	0.028
Control.Type	1	4.8762	4.8762	6.96	0.014
Control.rate	2	0.1312	0.0656	0.09	0.911
Control.Application_method	1	5.2590	5.2590	7.51	0.011
Control.Type.rate	2	0.4082	0.2041	0.29	0.750
Control.Type.Application_method	1	2.0289	2.0289	2.90	0.101
Control.rate.Application_method	2	3.1242	1.5621	2.23	0.128
Control.Type.rate.Application_method	2	2.0685	1.0343	1.48	0.247
Residual	26	18.2183	0.7007		
Total	41	45.4046			

P (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	25.130	12.565	1.42	
Block.*Units* stratum					
Control	1	1.446	1.446	0.16	0.689
Control.Between_controls	1	0.241	0.241	0.03	0.870
Control.Type	1	0.001	0.001	0.00	0.992
Control.rate	2	8.580	4.290	0.49	0.620
Control.Application_method	1	10.863	10.863	1.23	0.277
Control.Type. rate	2	29.187	14.593	1.65	0.211
Control.Type.Application_method	1	1.972	1.972	0.22	0.640
Control.rate.Application_method	2	9.178	4.589	0.52	0.601
Control.Type.rate.Application_method	2	23.805	11.903	1.35	0.277
Residual	26	229.444	8.825		
Total	41	339.847			

A.5.3.2 Second period (2006)

A.5.3.2.1 Annual Ryegrass

Total DM yield (g)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.1442	0.0721	0.08	
Block.*Units* stratum					
Control	3	41.4469	13.8156	15.40	<.001
Control.Type	1	9.3636	9.3636	10.44	0.003
Control.Rate	2	46.1483	23.0741	25.72	<.001
Control.Method	1	9.9856	9.9856	11.13	0.002
Control.Type.Rate	2	2.3620	1.1810	1.32	0.284
Control.Type.Method	1	3.1565	3.1565	3.52	0.071
Control.Rate.Method	2	1.5712	0.7856	0.88	0.428
Control.Type.Rate.Method	2	2.0585	1.0292	1.15	0.332
Residual	28	25.1151	0.8970		
Total	44	141.3519			

ln [DM yield (g)]

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.57488	0.28744	1.38	
Block.Date stratum					
Date	3	139.86375	46.62125	223.66	<.001
Residual	6	1.25069	0.20845	8.33	
Block.Date.*Units* stratum					
Control	3	0.61847	0.20616	8.24	<.001
Date.Control	9	1.72468	0.19163	7.66	<.001
Control.Type	1	0.58320	0.58320	23.30	<.001
Control.Rate	2	1.90427	0.95214	38.05	<.001
Control.Method	1	0.42221	0.42221	16.87	<.001
Date.Control.Type	3	0.10321	0.03440	1.37	0.254
Date.Control.Rate	6	0.20011	0.03335	1.33	0.249
Control.Type.Rate	2	0.05988	0.02994	1.20	0.306
Date.Control.Method	3	0.00822	0.00274	0.11	0.954
Control.Type.Method	1	0.34368	0.34368	13.73	<.001
Control.Rate.Method	2	0.00523	0.00262	0.10	0.901
Date.Control.Type.Rate	6	0.14859	0.02477	0.99	0.436
Date.Control.Type.Method	3	0.18106	0.06035	2.41	0.071
Date.Control.Rate.Method	6	0.10404	0.01734	0.69	0.656
Control.Type.Rate.Method	2	0.21024	0.10512	4.20	0.017
Date.Control.Type.Rate.Method	6	0.42454	0.07076	2.83	0.013
Residual	112	2.80294	0.02503		
Total	179	151.53387			

N uptake (mg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	52.09	26.04	0.64	
Block.Date stratum					
Date	3	48316.46	16105.49	397.84	<.001
Residual	6	242.89	40.48	2.87	
Block.Date.*Units* stratum					
Control	3	738.33	246.11	17.46	<.001
Date.Control	9	2346.43	260.71	18.50	<.001
Control.Type	1	400.25	400.25	28.40	<.001
Control.Rate	2	1600.48	800.24	56.77	<.001
Control.Method	1	238.72	238.72	16.94	<.001
Date.Control.Type	3	72.83	24.28	1.72	0.166
Date.Control.Rate	6	682.39	113.73	8.07	<.001
Control.Type.Rate	2	70.95	35.48	2.52	0.085
Date.Control.Method	3	14.86	4.95	0.35	0.788
Control.Type.Method	1	287.42	287.42	20.39	<.001
Control.Rate.Method	2	68.20	34.10	2.42	0.094
Date.Control.Type.Rate	6	89.01	14.83	1.05	0.396
Date.Control.Type.Method	3	39.24	13.08	0.93	0.430
Date.Control.Rate.Method	6	61.15	10.19	0.72	0.632
Control.Type.Rate.Method	2	129.95	64.98	4.61	0.012
Date.Control.Type.Rate.Method	6	197.50	32.92	2.34	0.037
Residual	112	1578.67	14.10		
Total	179	57227.82			

Total N uptake (mg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	208.4	104.2	1.02	
Block.*Units* stratum					
control	3	2953.3	984.4	9.64	<.001
control.Type	1	1601.0	1601.0	15.68	<.001
control.Rate	2	6401.9	3201.0	31.35	<.001
control.method	1	954.9	954.9	9.35	0.005
control.Type.Rate	2	283.8	141.9	1.39	0.266
control.Type.method	1	1149.7	1149.7	11.26	0.002
control.Rate.method	2	272.8	136.4	1.34	0.279
control.Type.Rate.method	2	519.8	259.9	2.55	0.096
Residual	28	2859.1	102.1		
Total	44	17204.6			

A.5.3.2.2 Leachate

Mineral N (mg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	50.61	25.31	0.10	
Block.Date stratum					
Date	1	67477.61	67477.61	260.79	0.004
Residual	2	517.49	258.74	3.20	
Block.Date.*Units* stratum					
Control	3	18033.09	6011.03	74.29	<.001
Date.Control	3	370.35	123.45	1.53	0.218
Control.Type	1	2383.78	2383.78	29.46	<.001
Control.Rate	2	54.96	27.48	0.34	0.713
Control.Method	1	187.32	187.32	2.32	0.134
Date.Control.Type	1	586.58	586.58	7.25	0.009
Date.Control.Rate	2	170.33	85.16	1.05	0.356
Control.Type.Rate	2	14.97	7.48	0.09	0.912
Date.Control.Method	1	57.45	57.45	0.71	0.403
Control.Type.Method	1	858.47	858.47	10.61	0.002
Control.Rate.Method	2	78.32	39.16	0.48	0.619
Date.Control.Type.Rate	2	51.07	25.54	0.32	0.731
Date.Control.Type.Method	1	379.62	379.62	4.69	0.035
Date.Control.Rate.Method	2	374.50	187.25	2.31	0.108
Control.Type.Rate.Method	2	102.23	51.12	0.63	0.535
Date.Control.Type.Rate.Method	2	221.46	110.73	1.37	0.263
Residual	56	4530.95	80.91		
Total	89	96501.18			

ln [mineral N (mg)]

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.01451	0.00726	0.37	
Block.*Units* stratum					
Control	3	3.06795	1.02265	52.74	<.001
Control.Type	1	0.86623	0.86623	44.67	<.001
Control.Rate	2	0.02335	0.01168	0.60	0.555
Control.Method	1	0.02795	0.02795	1.44	0.240
Control.Type.Rate	2	0.00878	0.00439	0.23	0.799
Control.Type.Method	1	0.28715	0.28715	14.81	<.001
Control.Rate.Method	2	0.03487	0.01744	0.90	0.418
Control.Type.Rate.Method	2	0.03948	0.01974	1.02	0.374
Residual	28	0.54297	0.01939		
Total	44	4.91323			

TSN (mg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.752	0.376	0.25	
Block.*Units* stratum					
Control	3	9.007	3.002	1.99	0.139
Control.Type	1	20.817	20.817	13.77	<.001
Control.Rate	2	6.756	3.378	2.23	0.126
Control.Method	1	3.882	3.882	2.57	0.120
Control.Type.Rate	2	4.895	2.447	1.62	0.216
Control.Type.Method	1	5.487	5.487	3.63	0.067
Control.Rate.Method	2	6.432	3.216	2.13	0.138
Control.Type.Rate.Method	2	2.554	1.277	0.84	0.440
Residual	28	42.324	1.512		
Total	44	102.905			

OSN (mg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.606	0.303	0.21	
Block.*Units* stratum					
Control	3	9.648	3.216	2.20	0.111
Control.Type	1	21.548	21.548	14.71	<.001
Control.Rate	2	7.145	3.573	2.44	0.106
Control.Method	1	3.798	3.798	2.59	0.119
Control.Type.Rate	2	4.587	2.294	1.57	0.227
Control.Type.Method	1	5.277	5.277	3.60	0.068
Control.Rate.Method	2	6.029	3.014	2.06	0.147
Control.Type.Rate.Method	2	2.856	1.428	0.97	0.390
Residual	28	41.014	1.465		
Total	44	102.508			

A.5.3.2.3 Soil

C: N ratio

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.74057	0.37029	11.05	
Block.*Units* stratum					
Control	3	0.18979	0.06326	1.89	0.155
Control.Type	1	0.09893	0.09893	2.95	0.097
Control.Rate	2	0.00522	0.00261	0.08	0.925
Control.Method	1	0.12500	0.12500	3.73	0.064
Control.Type.Rate	2	0.02027	0.01014	0.30	0.741
Control.Type.Method	1	0.00222	0.00222	0.07	0.799
Control.Rate.Method	2	0.01612	0.00806	0.24	0.788
Control.Type.Rate.Method	2	0.11250	0.05625	1.68	0.205
Residual	28	0.93866	0.03352		
Total	44	2.24927			

Total N (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.00064386	0.00032193	4.32	
Block.*Units* stratum					
Control	3	0.00400202	0.00133401	17.89	<.001
Control.Type	1	0.00007854	0.00007854	1.05	0.314
Control.Rate	2	0.00156859	0.00078430	10.52	<.001
Control.Method	1	0.00023463	0.00023463	3.15	0.087
Control.Type.Rate	2	0.00003213	0.00001606	0.22	0.808
Control.Type.Method	1	0.00013293	0.00013293	1.78	0.193
Control.Rate.Method	2	0.00004885	0.00002442	0.33	0.723
Control.Type.Rate.Method	2	0.00001208	0.00000604	0.08	0.922
Residual	28	0.00208821	0.00007458		
Total	44	0.00884184			

Total C (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.157179	0.078590	9.54	
Block.*Units* stratum					
Control	3	0.373062	0.124354	15.09	<.001
Control.Type	1	0.001054	0.001054	0.13	0.723
Control.Rate	2	0.188667	0.094334	11.45	<.001
Control.Method	1	0.008461	0.008461	1.03	0.320
Control.Type.Rate	2	0.001111	0.000555	0.07	0.935
Control.Type.Method	1	0.016163	0.016163	1.96	0.172
Control.Rate.Method	2	0.002850	0.001425	0.17	0.842
Control.Type.Rate.Method	2	0.008860	0.004430	0.54	0.590
Residual	28	0.230678	0.008239		
Total	44	0.988086			

OM (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.59902	0.29951	10.90	
Block.*Units* stratum					
Control	3	0.80705	0.26902	9.79	<.001
Control.Type	1	0.05076	0.05076	1.85	0.185
Control.Rate	2	0.91106	0.45553	16.59	<.001
Control.Method	1	0.22651	0.22651	8.25	0.008
Control.Type.Rate	2	0.00312	0.00156	0.06	0.945
Control.Type.Method	1	0.09320	0.09320	3.39	0.076
Control.Rate.Method	2	0.16446	0.08223	2.99	0.066
Control.Type.Rate.Method	2	0.21421	0.10711	3.90	0.032
Residual	28	0.76905	0.02747		
Total	44	3.83844			

K (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	7437.0	3718.5	15.31	
Block.*Units* stratum					
Control	3	11397.8	3799.3	15.64	<.001
Control.Type	1	7095.0	7095.0	29.22	<.001
Control.Rate	2	13898.0	6949.0	28.61	<.001
Control.Method	1	3981.3	3981.3	16.39	<.001
Control.Type.Rate	2	1911.2	955.6	3.93	0.031
Control.Type.Method	1	357.2	357.2	1.47	0.235
Control.Rate.Method	2	594.1	297.1	1.22	0.310
Control.Type.Rate.Method	2	1290.5	645.2	2.66	0.088
Residual	28	6799.9	242.9		
Total	44	54762.0			

Na (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	414.14	207.07	10.07	
Block.*Units* stratum					
Control	3	51.51	17.17	0.83	0.486
Control.Type	1	4.03	4.03	0.20	0.661
Control.Rate	2	170.97	85.49	4.16	0.026
Control.Method	1	1.75	1.75	0.09	0.772
Control.Type.Rate	2	5.85	2.92	0.14	0.868
Control.Type.Method	1	0.18	0.18	0.01	0.926
Control.Rate.Method	2	221.50	110.75	5.39	0.011
Control.Type.Rate.Method	2	60.00	30.00	1.46	0.250
Residual	28	575.86	20.57		
Total	44	1505.80			

ln [NH4 (mg/kg)]

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	1.1290	0.5645	4.36	
Block.*Units* stratum					
Control	3	1.2137	0.4046	3.13	0.041
Control.Type	1	0.2734	0.2734	2.11	0.157
Control.Rate	2	0.1986	0.0993	0.77	0.474
Control.Method	1	0.0541	0.0541	0.42	0.523
Control.Type.Rate	2	0.1062	0.0531	0.41	0.667
Control.Type.Method	1	0.7333	0.7333	5.67	0.024
Control.Rate.Method	2	0.4215	0.2108	1.63	0.214
Control.Type.Rate.Method	2	0.3364	0.1682	1.30	0.288
Residual	28	3.6213	0.1293		
Total	44	8.0875			

ln [P (mg/kg)]

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	0.08401	0.04200	3.62	
Block.*Units* stratum					
Control	3	0.76500	0.25500	21.99	<.001
Control.Type	1	0.91625	0.91625	79.01	<.001
Control.Rate	2	0.48343	0.24171	20.84	<.001
Control.Method	1	0.00143	0.00143	0.12	0.728
Control.Type.Rate	2	0.32718	0.16359	14.11	<.001
Control.Type.Method	1	0.00412	0.00412	0.36	0.556
Control.Rate.Method	2	0.04717	0.02358	2.03	0.150
Control.Type.Rate.Method	2	0.04430	0.02215	1.91	0.167
Residual	28	0.32472	0.01160		
Total	44	2.99761			

Moisture (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Block stratum	2	44.4696	22.2348	33.24	
Block.*Units* stratum					
Control	3	12.3482	4.1161	6.15	0.002
Control.Type	1	1.2142	1.2142	1.81	0.189
Control.Rate	2	25.0212	12.5106	18.70	<.001
Control.Method	1	6.9332	6.9332	10.36	0.003
Control.Type.Rate	2	3.5243	1.7622	2.63	0.089
Control.Type.Method	1	0.4713	0.4713	0.70	0.408
Control.Rate.Method	2	4.2623	2.1311	3.19	0.057
Control.Type.Rate.Method	2	4.3504	2.1752	3.25	0.054
Residual	28	18.7312	0.6690		
Total	44	121.3257			

A.5.4 Lysimeter experiment

A.5.4.1 Forage maize plant height

***ln* [Plant height (cm)]**

Source of variation	d.f.	(m.v.)	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum						
Control	4		0.753618	0.188405	3.67	0.018
Control.Type	1		4.080647	4.080647	79.47	<.001
Control.Rate	3		0.677379	0.225793	4.40	0.013
Control.Type.Rate	3		1.558978	0.519659	10.12	<.001
Residual	24		1.232326	0.051347	5.38	
Lysim_No.Time stratum						
Time	3		232.718108	77.572703	8133.92	<.001
Time.Control	12		0.445553	0.037129	3.89	<.001
Time.Control.Type	3		0.210955	0.070318	7.37	<.001
Time.Control.Rate	9		0.335870	0.037319	3.91	<.001
Time.Control.Type.Rate	9		0.318440	0.035382	3.71	<.001
Residual	71	(1)	0.677123	0.009537		
Total	142	(1)	236.178692			

A.5.4.2 Forage maize production at thinning

DM yield (g)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control	4	0.8572	0.2143	1.22	0.330
control.type	1	2.2448	2.2448	12.73	0.002
control.rate	3	0.7256	0.2419	1.37	0.275
control.type.rate	3	2.4287	0.8096	4.59	0.011
Residual	24	4.2321	0.1763		
Total	35	10.4885			

***Sqrt* [N uptake (mg)]**

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control	4	21.808	5.452	4.83	0.005
control.type	1	22.299	22.299	19.77	<.001
control.rate	3	6.048	2.016	1.79	0.176
control.type.rate	3	16.808	5.603	4.97	0.008
Residual	24	27.066	1.128		
Total	35	94.030			

A.5.4.3 Forage maize production at harvest

***Sqrt* [DM (g)]**

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control	4	47.8113	11.9528	14.61	<.001
control.type	1	94.7237	94.7237	115.76	<.001
control.rate	3	22.1621	7.3874	9.03	<.001
control.type.rate	3	27.9287	9.3096	11.38	<.001
Residual	24	19.6380	0.8182		
Total	35	212.2638			

***Sqrt* [N uptake (mg)]**

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control	4	0.21614	0.05403	5.24	0.004
control.type	1	0.58774	0.58774	57.05	<.001
control.rate	3	0.21413	0.07138	6.93	0.002
control.type.rate	3	0.16293	0.05431	5.27	0.006
Residual	24	0.24727	0.01030		
Total	35	1.42821			

***Sqrt* [K uptake (g)]**

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control_v_T	1	1.55467	1.55467	144.40	<.001
control_v_T.b_controls	3	0.02117	0.00706	0.66	0.587
control_v_T.type	1	1.51002	1.51002	140.25	<.001
control_v_T.rate	3	0.41328	0.13776	12.80	<.001
control_v_T.type.rate	3	0.35068	0.11689	10.86	<.001
Residual	24	0.25840	0.01077		
Total	35	4.10822			

***Sqrt* [P uptake (mg)]**

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control_v_T	1	135.2464	135.2464	188.85	<.001
control_v_T.b_controls	3	0.5031	0.1677	0.23	0.872
control_v_T.type	1	172.1633	172.1633	240.39	<.001
control_v_T.rate	3	55.7927	18.5976	25.97	<.001
control_v_T.type.rate	3	22.3244	7.4415	10.39	<.001
Residual	24	17.1882	0.7162		
Total	35	403.2180			

Total K (g/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control_v_T	1	186.937	186.937	21.17	<.001
control_v_T.b_controls	3	56.515	18.838	2.13	0.122
control_v_T.type	1	41.429	41.429	4.69	0.040
control_v_T.rate	3	9.480	3.160	0.36	0.784
control_v_T.type.rate	3	50.241	16.747	1.90	0.157
Residual	24	211.939	8.831		
Total	35	556.540			

Total P (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control_v_T	1	2091256.	2091256.	98.83	<.001
control_v_T.b_controls	3	13886.	4629.	0.22	0.882
control_v_T.type	1	769856.	769856.	36.38	<.001
control_v_T.rate	3	708484.	236161.	11.16	<.001
control_v_T.type.rate	3	55808.	18603.	0.88	0.466
Residual	24	507842.	21160.		
Total	35	4147131.			

A.5.4.3 Sand properties at thinning

P (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control_v_T	1	69.871	69.871	14.72	<.001
control_v_T.b_controls	3	15.390	5.130	1.08	0.376
control_v_T.type	1	74.276	74.276	15.65	<.001
control_v_T.rate	3	98.143	32.714	6.89	0.002
control_v_T.type.rate	3	32.282	10.761	2.27	0.106
Residual	24	113.889	4.745		
Total	35	403.851			

Sqrt [mineral N (mg/kg)]

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control	4	2.0893	0.5223	4.85	0.005
control.type	1	1.7892	1.7892	16.60	<.001
control.rate	3	1.0255	0.3418	3.17	0.043
control.type.rate	3	0.8004	0.2668	2.48	0.086
Residual	24	2.5865	0.1078		
Total	35	8.2908			

K (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control_v_T	1	15613.9	15613.9	38.22	<.001
control_v_T.b_controls	3	121.1	40.4	0.10	0.960
control_v_T.type	1	1169.7	1169.7	2.86	0.104
control_v_T.rate	3	14066.2	4688.7	11.48	<.001
control_v_T.type.rate	3	1939.9	646.6	1.58	0.220
Residual	24	9805.2	408.6		
Total	35	42716.0			

A.5.4.4 Sand properties at harvest

A.5.4.4.1 Olsen P, total P, extractable Ca, and pH

***ln* [P (mg/kg)]**

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control_v_T	1	7.0485	7.0485	24.66	<.001
control_v_T.b_controls	3	0.3511	0.1170	0.41	0.748
control_v_T.type	1	5.4813	5.4813	19.17	<.001
control_v_T.rate	3	3.0791	1.0264	3.59	0.028
control_v_T.type.rate	3	0.3287	0.1096	0.38	0.766
Residual	24	6.8611	0.2859		
Total	35	23.1499			

Total P (mg/kg)

Source of variation	d.f.	(m.v.)	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum						
control_v_T	1		775.7	775.7	6.17	0.021
control_v_T.b_controls	3		1263.9	421.3	3.35	0.037
control_v_T.type	1		633.5	633.5	5.04	0.035
control_v_T.rate	3		220.0	73.3	0.58	0.632
control_v_T.type.rate	3		153.6	51.2	0.41	0.749
Residual	23	(1)	2892.6	125.8		
Total	34	(1)	5929.3			

Ca (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control_v_T	1	43931.	43931.	7.60	0.011
control_v_T.b_controls	3	8942.	2981.	0.52	0.675
control_v_T.type	1	4938.	4938.	0.85	0.364
control_v_T.rate	3	80763.	26921.	4.66	0.011
control_v_T.type.rate	3	5965.	1988.	0.34	0.794
Residual	24	138689.	5779.		
Total	35	283228.			

pH

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control_v_T	1	0.00347	0.00347	0.04	0.848
control_v_T.b_controls	3	0.35000	0.11667	1.27	0.307
control_v_T.type	1	0.35042	0.35042	3.81	0.063
control_v_T.rate	3	0.10458	0.03486	0.38	0.769
control_v_T.type.rate	3	0.10125	0.03375	0.37	0.777
Residual	24	2.20667	0.09194		
Total	35	3.11639			

A.5.4.4.2 Mineral N, extractable K and Na, OM and total C

Average mineral N (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control	4	0.32508	0.08127	3.10	0.034
control.type	1	0.06705	0.06705	2.56	0.123
control.rate	3	0.00919	0.00306	0.12	0.949
control.type.rate	3	0.33323	0.11108	4.24	0.015
Residual	24	0.62904	0.02621		
Total	35	1.36358			

K (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control_v_T	1	3300.3	3300.3	9.50	0.005
control_v_T.b_controls	3	547.5	182.5	0.53	0.669
control_v_T.type	1	283.6	283.6	0.82	0.375
control_v_T.rate	3	2068.5	689.5	1.98	0.143
control_v_T.type.rate	3	1156.8	385.6	1.11	0.365
Residual	24	8338.0	347.4		
Total	35	15694.7			

Na (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control_v_T	1	69.284	69.284	7.41	0.012
control_v_T.b_controls	3	24.919	8.306	0.89	0.461
control_v_T.type	1	6.987	6.987	0.75	0.396
control_v_T.rate	3	71.043	23.681	2.53	0.081
control_v_T.type.rate	3	17.241	5.747	0.61	0.612
Residual	24	224.399	9.350		
Total	35	413.873			

OM (%)

Source of variation	d.f.	(m.v.)	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum						
control_v_T	1		0.00922	0.00922	0.38	0.542
control_v_T.b_controls	3		0.29730	0.09910	4.12	0.018
control_v_T.type	1		0.00540	0.00540	0.22	0.640
control_v_T.rate	3		0.27411	0.09137	3.80	0.024
control_v_T.type.rate	3		0.17374	0.05791	2.41	0.093
Residual	23	(1)	0.55370	0.02407		
Total	34	(1)	1.23571			

Total C (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control_v_T	1	0.089849	0.089849	71.01	<.001
control_v_T.b_controls	3	0.000057	0.000019	0.01	0.997
control_v_T.type	1	0.002958	0.002958	2.34	0.139
control_v_T.rate	3	0.065708	0.021903	17.31	<.001
control_v_T.type.rate	3	0.003813	0.001271	1.00	0.408
Residual	24	0.030369	0.001265		
Total	35	0.192752			

A.5.4.4.3 EC_e and ESP at the high application rates, and WRC

ESP (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Sample_No stratum					
Type	3	0.13533	0.04511	4.10	0.049
Residual	8	0.08802	0.01100		
Total	11	0.22335			

EC_e (dS/m)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Sample_No stratum					
Type	3	0.14149	0.04716	1.90	0.208
Residual	8	0.19866	0.02483		
Total	11	0.34015			

Available water (%) (v/v)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysimeter stratum					
Control	1	13.770	13.770	7.27	0.015
Control.Type	1	8.933	8.933	4.71	0.044
Control.Rate	3	60.923	20.308	10.72	<.001
Control.Type.Rate	3	16.283	5.428	2.86	0.066
Residual	18	34.114	1.895		
Total	26	134.022			

Moisture (v/v) (%)

Source of variation	d.f.	(m.v.)	s.s.	m.s.	v.r.	F pr.
Lysimeter stratum						
ControlvT	1		242.326	242.326	5.16	0.036
ControlvT.Type	1		87.422	87.422	1.86	0.189
ControlvT.Rate	3		803.681	267.894	5.70	0.006
ControlvT.Type.Rate	3		182.756	60.919	1.30	0.306
Residual	18		845.665	46.981	21.99	
Lysimeter.Suction_kPa stratum						
Suction_kPa	8		50227.566	6278.446	2938.23	<.001
Suction_kPa.ControlvT	8		68.657	8.582	4.02	<.001
Suction_kPa.ControlvT.Type	8		47.828	5.979	2.80	0.007
Suction_kPa.ControlvT.Rate	24		150.286	6.262	2.93	<.001
Suction_kPa.ControlvT.Type.Rate	24		85.405	3.559	1.67	0.036
Residual	142	(2)	303.427	2.137		
Total	240	(2)	52619.393			

A.5.4.5 Leachate

A.5.4.5.1 During the period from May 2006 to November 2006

***ln* [total mineral N (mg)]**

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control	4	73.6987	18.4247	36.76	<.001
control.type	1	5.1076	5.1076	10.19	0.004
control.rate	3	0.1769	0.0590	0.12	0.949
control.type.rate	3	2.1575	0.7192	1.43	0.257
Residual	24	12.0282	0.5012		
Total	35	93.1689			

***ln* [mineral N (mg)]**

Source of variation	d.f.	(m.v.)	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum						
control	4		842.791	210.698	32.57	<.001
control.type	1		22.873	22.873	3.54	0.072
control.rate	3		7.266	2.422	0.37	0.772
control.type.rate	3		2.132	0.711	0.11	0.954
Residual	24		155.259	6.469	4.51	
Lysim_No.date stratum						
date	11		590.250	53.659	37.42	<.001
date.control	44		556.947	12.658	8.83	<.001
date.control.type	11		101.105	9.191	6.41	<.001
date.control.rate	33		51.360	1.556	1.09	0.351
date.control.type.rate	33		83.783	2.539	1.77	0.008
Residual	263	(1)	377.152	1.434		
Total	430	(1)	2786.458			

A.5.4.5.2 During the period of June 2006 to November 2006

***ln* [TSN (mg)]**

Source of variation	d.f.	(m.v.)	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum						
control	4		427.2882	106.8220	72.67	<.001
control.rate	3		1.5725	0.5242	0.36	0.785
control.type	1		2.0415	2.0415	1.39	0.250
control.rate.type	3		4.0393	1.3464	0.92	0.448
Residual	24		35.2801	1.4700	3.30	
Lysim_No.date stratum						
date	7		209.0585	29.8655	67.03	<.001
date.control	28		128.3469	4.5838	10.29	<.001
date.control.rate	21		5.4602	0.2600	0.58	0.926
date.control.type	7		6.5153	0.9308	2.09	0.047
date.control.rate.type	21		6.0767	0.2894	0.65	0.876
Residual	167	(1)	74.4117	0.4456		
Total	286	(1)	899.0452			

***ln* [Total TSN (mg)]**

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control_v_T	4	83.9258	20.9814	173.44	<.001
control_v_T.type	1	0.0751	0.0751	0.62	0.438
control_v_T.rate	3	0.1119	0.0373	0.31	0.819
control_v_T.type.rate	3	0.3599	0.1200	0.99	0.414
Residual	24	2.9033	0.1210		
Total	35	87.3760			

***ln* [total PO_4-P (mg)]**

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control_v_T	1	12.0792	12.0792	19.12	<.001
control_v_T.b_controls	3	6.2731	2.0910	3.31	0.037
control_v_T.type	1	1.7932	1.7932	2.84	0.105
control_v_T.rate	3	2.7672	0.9224	1.46	0.250
control_v_T.type.rate	3	1.7455	0.5818	0.92	0.446
Residual	24	15.1598	0.6317		
Total	35	39.8179			

A.5.4.5.3 During summer 2007

ln **[mineral N (mg)]**

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control	4	8.3318	2.0829	5.33	0.003
control.type	1	0.0509	0.0509	0.13	0.721
control.rate	3	0.9544	0.3181	0.81	0.498
control.type.rate	3	0.1351	0.0450	0.12	0.950
Residual	24	9.3713	0.3905		
Total	35	18.8435			

ln **[TSN (mg)]**

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control	4	2.1338	0.5334	4.90	0.005
control.type	1	0.0830	0.0830	0.76	0.391
control.rate	3	0.0547	0.0182	0.17	0.917
control.type.rate	3	0.0206	0.0069	0.06	0.979
Residual	24	2.6122	0.1088		
Total	35	4.9043			

OSN (mg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Lysim_No stratum					
control	4	6.322	1.581	1.40	0.264
control.type	1	0.736	0.736	0.65	0.427
control.rate	3	11.774	3.925	3.48	0.032
control.type.rate	3	3.604	1.201	1.06	0.383
Residual	24	27.101	1.129		
Total	35	49.537			

A.5.5 Incubation

Mineral N (mg/kg)

Source of variation	d.f.	(m.v.)	s.s.	m.s.	v.r.	F pr.
No stratum						
Control_v_treated	1		613.792	613.792	74.48	<.001
Control_v_treated.Type	1		3644.822	3644.822	442.30	<.001
Control_v_treated.Rate	2		1097.656	548.828	66.60	<.001
Control_v_treated.Type.Rate	2		446.808	223.404	27.11	<.001
Residual	14		115.369	8.241	1.28	
No.Day stratum						
Day	9		5778.534	642.059	100.10	<.001
Day.Control_v_treated	9		56.953	6.328	0.99	0.455
Day.Control_v_treated.Type	9		154.356	17.151	2.67	0.007
Day.Control_v_treated.Rate	18		188.421	10.468	1.63	0.062
Day.Control_v_treated.Type.Rate	18		147.360	8.187	1.28	0.215
Residual	125	(1)	801.802	6.414		
Total	208	(1)	12667.175			

TSN (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
No stratum					
Control_v_treated	1	395.53	395.53	3.54	0.081
Control_v_treated.Type	1	1602.71	1602.71	14.35	0.002
Control_v_treated.Rate	2	2077.67	1038.84	9.30	0.003
Control_v_treated.Type.Rate	2	452.06	226.03	2.02	0.169
Residual	14	1563.76	111.70	1.48	
No.Day stratum					
Day	9	2571.39	285.71	3.78	<.001
Day.Control_v_treated	9	802.26	89.14	1.18	0.314
Day.Control_v_treated.Type	9	765.26	85.03	1.12	0.350
Day.Control_v_treated.Rate	18	2243.18	124.62	1.65	0.058
Day.Control_v_treated.Type.Rate	18	1172.09	65.12	0.86	0.626
Residual	126	9527.31	75.61		
Total	209	23173.22			

OSN (mg/kg)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
No stratum					
Control_v_treated	1	23.88	23.88	0.21	0.651
Control_v_treated.Type	1	413.64	413.64	3.70	0.075
Control_v_treated.Rate	2	322.91	161.45	1.45	0.269
Control_v_treated.Type.Rate	2	16.98	8.49	0.08	0.927
Residual	14	1564.26	111.73	1.42	
No.Day stratum					
Day	9	5659.80	628.87	8.01	<.001
Day.Control_v_treated	9	761.97	84.66	1.08	0.383
Day.Control_v_treated.Type	9	1083.28	120.36	1.53	0.143
Day.Control_v_treated.Rate	18	3055.10	169.73	2.16	0.007
Day.Control_v_treated.Type.Rate	18	978.36	54.35	0.69	0.813
Residual	126	9889.11	78.48		
Total	209	23769.29			

NMR$_{total}$ (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
No stratum					
Type	1	2557.164	2557.164	136.26	<.001
Rate	2	34.896	17.448	0.93	0.421
Type.Rate	2	95.218	47.609	2.54	0.121
Residual	12	225.199	18.767	2.93	
No.Day stratum					
Day	9	315.598	35.066	5.48	<.001
Day.Type	9	139.037	15.449	2.41	0.016
Day.Rate	18	139.060	7.726	1.21	0.269
Day.Type.Rate	18	159.279	8.849	1.38	0.155
Residual	108	691.549	6.403		
Total	179	4356.999			

NMR$_{organic}$ (%)

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
No stratum					
Type	1	100.288	100.288	4.92	0.047
Rate	2	40.251	20.125	0.99	0.401
Type.Rate	2	105.192	52.596	2.58	0.117
Residual	12	244.619	20.385	2.92	
No.Day stratum					
Day	9	345.943	38.438	5.51	<.001
Day.Type	9	155.860	17.318	2.48	0.013
Day.Rate	18	146.900	8.161	1.17	0.299
Day.Type.Rate	18	168.667	9.370	1.34	0.177
Residual	108	753.755	6.979		
Total	179	2061.475			

A.6 List of publications produced from this research

Results from this research project were presented at European and International conferences on agricultural engineering and waste management. The papers produced are listed below:

1. Kokkora M.I. and Hann M.J, 2005. The effects of using municipal solid waste (MSW) and vegetable compost in agriculture. In: Lowe, P., Horan, N. (Ed.), Proceedings of the 10th European Biosolids and Biowastes Conference and Workshop, 13-16 November 2005, Wakefield, UK, Aqua Enviro Technology Transfer, 51.1-51.7.
2. Kokkora M.I., Hann M.J. and Godwin R.J., 2006. Biowaste and vegetable compost utilization in agriculture: an agronomic and environmental assessment. In: Proceedings of ISTRO 17 Conference 'Sustainability - its Impact on Soil Management and Environment', 28 August - 3 September 2006, Kiel, Germany, 1431-1436.
3. Kokkora M.I., Hann M.J. and Godwin R.J., 2006. Biowaste and vegetable compost utilization in agriculture: an agronomic and environmental assessment. In: Soil Management for Sustainability, Advances in GeoEcology, 38, Catena Verlag, Reiskrichen, Germany, 388-394.
4. Kokkora M.I. and Hann M.J., 2007. Crop production and nitrogen leaching from biowaste and vegetable compost amended sand. In: Proceedings of the Sardinia Symposium 2007 – 11th International Waste Management and Landfill Symposium, 1-5 October 2007, S.Margherita di Pula (Cagliari), Italy.
5. Kokkora M.I. and Hann M.J., 2008. Agricultural use of bio- and vegetable- waste composts: effects on nitrogen leaching and crop production. In: Proceedings of the AgEng 2008 Conference 'Agricultural and Biosystems Engineering for a Sustainable World', 23-25 June 2008, Hersonissos (Crete), Greece, *in press*.
6. Kokkora M.I., Hann M.J. and Tyrrel, S.F., 2008. Organic waste compost parameters in relation to soil properties. In: Proceedings of the AgEng 2008 Conference 'Agricultural and Biosystems Engineering for a Sustainable World', 23-25 June 2008, Hersonissos (Crete), Greece, *in press*.

7. Kokkora M.I., Hann M.J. and Godwin R.J., 2008. The use of vegetable- and bio-waste composts: effects on forage maize production and soil properties. In: Proceedings of the ASABE 2008 Annual International Meeting, 29 June – 2 July 2008, Providence (Rhode Island), USA, *in press*.