DIAS report

June 1999

13 • Plant Production

J. Petersen & S. O. Petersen (eds.)

Use of municipal organic waste

Proceedings of NJF seminar no. 292, November 23-25, 1998
Agricultural Research Centre, Jokioinen, Finland

Ministry of Food, Agriculture and Fisheries
Danish Institute of Agricultural Sciences

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DIAS report Plant Production no. 13 • June 1999 • 2nd volume

Publisher:	Danish Institute of Agricultural Sciences Research Centre Foulum P.O. Box 50 DK-8830 Tjele	Tel. +45 89 99 19 00 Fax +45 89 99 19 19
Sale by copies: (incl. VAT)	up to 50 pages up to 100 pages more than100 pages	50,- DKK 75,- DKK 100,- DKK
Subscription:	Depending on the number of reports sent but equivalent to 75% of the price of sale by copies.	

04/06/99, 9:54

Preface

Section I, *Soil and Fertilization*, of The Nordic Association of Agricultural Scientists (in Swedish: Nordiska Jordbruksforskares Förening, abbreviated NJF) was interested in a seminar concerning the use of organic waste in agriculture. In advance it was decided to hold the seminar in Finland, and Section I appointed the following organizing committee:

Torleiv Næss Ugland, Apelsvoll Research Centre, The Norwegian Crop Research Institute,
Sirkka Malkki, TTS-Institute, Finland,
Ylva Eklind, Horticultural Research Station/Uppsala, Swedish Univ. of Agricultural Sciences
Olof Thomsson, Dept. of Agricultural Engineering, Swedish Univ. of Agricultural Sciences
Olafur R. Dyrmundsson, The Farmers Association of Iceland, and
Jens Petersen, Dept. of Crop Physiology and Soil Science, Danish Inst. of Agric. Sciences.

The organizing committee met in Oslo in January 1998, where Torleiv Næss Ugland was voted as chairman, and in Helsinki just before the seminar. In the meantime 3 telephone meetings were held. The theme of the seminar was decided to be:

Municipal organic waste, such as toilet waste, kitchen and garden waste, is a potential source of plant nutrients and soil improvement in agriculture and horticulture. However, risks to hygiene and soil contamination must be minimized. The main waste type is sewage sludge, but composted or anaerobically digested municipal wastes are on the increase. Some of these waste types are a result of the demand for waste treatment rather than the recycling of nutrients. Therefore, special interest will be given to waste products from newly developed, "alternative" treatment systems focusing on increased recycling of plant nutrients. The seminar will deal with the following topics:

- *The current situation in the Nordic countries concerning the use of waste products, including legislation (opening session).*
- *The effects of waste product application on soil and plants.*
  - *effect on soil physical properties and microbiology*
  - *effect of contaminants*
  - *prediction of mineral-N dynamic following addition of organic waste*
- *The waste quality and hygienic aspects of waste product use.*
- *The principles of systems for recycling municipal organic waste to agriculture.*
  - *farmer operated systems*
  - *source separating systems*
  - *composting lavatories in permanent houses*
  - *filter media used as fertilizers*
- *Alternative use of waste in forests and grass for energy-production.*

In total 58 scientists from Iceland, Sweden, Norway, Denmark, Finland and Russia participated in the seminar. There were 20 oral presentations and 10 posters presented. Futhermore, there were two invited speakers: Ann Albihn, National Veterinary Institute, and Staffan Steineck, Institute of Agricultural Engineering, both from Uppsala, Sweden.

The seminar was held on November 23-25, 1998 at the Agricultural Research Centre, Jokioinen, Finland. Oral and poster presentations on November 23-24, and an excursion to two treatment plants on November 25. Ämmässuo Waste Handling Centre at Espoo is owned by The Helsinki Metropolitan Area. The handling centre operates a reception for usable and hazardous waste, a waste station for small deliveries, a composting plant for separately collected biowaste and a composting area for sewage sludge. The metropolitan region's only civic waste landfill is sited in the vicinity of the handling centre. Viikinmäki Wastewater Treatment Plant is located in Helsinki, and treats the wastewater of both the 700,000 inhabitants and the industry in its sewerage district. The seven parallel process lines, facilities for pretreatment, sludge treatment, machinery and equipment have been excavated in the rock.

The organizing committee thanks the participants of the seminar and the speakers for their contribution, particularly the key-note speakers. Further the committee thanks Kaija Laaksonen, TTS-Institute, for taking care of the registration and the technical editing of the programme and website. Also a warm thank to the staff at the Resource Management Research, Agricultural Research Centre of Finland, for a pleasant time at Jokioinen, and for taking care of co-ordination of pracitical seminar arrangements, excursion, conference dinner (Ritva Mäklelä-Kurtto and Jouko Sippola), contact person during the seminar, management of audiovisual equipments, e-mail facilities and poster exhibition (Mikael Lindholm), reception and service (Tiina Rämö and Leif Söderlund).

The editors thanks Margit Schacht, Danish Institute of Agricultural Sciences, for invaluable technical editing with reference to uniformity of the papers and generel help to move the Proceedings though to completion.

This report replaces *NJF-utredning no. 125* (ISSN 0333-1350).

Danish Institute of Agricultural Sciences
Research Centre Foulum
May 1999

J. Petersen and S.O. Petersen

Contents

Summary

Torleiv Næss Ugland

The seminar brought together researchers with backgrounds in different scientific disciplines to discuss their latest findings concerning municipal organic waste. The program consisted of a wide range of items including legislation, effects on soil and microbial life, quality and use of waste, and systems for waste treatment.

Speaking about legislation, it was pointed out that especially the hygienic aspect was treated differently in the Nordic Countries. It may be a common goal to hygienize the waste to reduce the potential for contamination of water and crops. It is also important to diminish insecurity and fear amongst the farmers.

Concerning technology, it seems to be important that researchers with biological background are at the front and deliver basic knowledge to the constructors of wastewater plants and equipment for waste handling. Lack of knowledge about biological processes has often led to process failure in process and diminished product quality.

Meanwhile, the waste products are not only used in traditional ways in agriculture, but are finding new uses like reclaiming deserts in Island, energy production on canary grass in Finland and spreading on agricultural land through so-called biobeds in Finland.

Farmer operated systems for both sludge and kitchen waste are in their infancy in Norway, but seem to be interesting solutions to close the link between farmers and consumers of agricultural products (waste users and waste producers).

A good waste quality is important for the end users, and a lot of work is carried out concerning toxic organics and heavy metals. It is important to find the sources of these pollutants and minimise the content in waste. But although the inhabitants of cities are very concerned about clean products from agriculture, they are not able to serve the same agriculture with clean waste.

The quality aspects of waste discussed during the seminar included effects on soil properties like bulk density, C and N mineralization potential, soil water holding capacity and aggregate stability as well as microbiological properties like antifungal activity and possibility of suppressing the growth of phytopathogenic fungi by using compost. These items put an increased focus on the positive effects of different wastes, looking for possibilities and advantages and making a good balance between risks and advantages with waste use.

The challenge for further research may be – not only to treat the situation we have today, but play an active role in trying to improve the systems for:

- the collection of waste
- the treatment of waste
- the regulations and restrictions of waste-use given by the authorities
- the use of waste
- the utilisation of new waste-based products
- the utilisation of methods to estimate the quality of waste

- according to updated biological and ecological knowledge.

Session 1 - Municipal organic waste and legislation in the Nordic Countries

Jens Petersen

Council directives of the European Community (EC) has the purpose to 1) reduce the environmental effect of effluents from waste water treatment plants and 2) re-use of the nutrients in organic wastes by application to agricultural land. By the agreement on the European Economic Area, which has been in force since 1994, the EC-directives are joined by the states in the European Free Trade Association (EFTA). Hence, the directives are valid in Denmark (EC), Iceland (EFTA), Finland (EC), Norway (EFTA) and Sweden (EC). The directives order each state to elaborate national legislation.

Each of the Nordic countries has retrictions for heavy metals content in the waste and soil. Furthermore, there are often restrictions on application rate and options for crops. Since the early 1980s a great effort has been done to reduce the content of heavy metals in waste water, and today heavy metals is a minor problem in agricultural use of sewage sludge.

Among the different types of municipal organic waste, sewage sludge is the main product. Between 30 and 70% is used in agriculture, but the content of nutrients available to the plants is low due to the operation of the waste water treatment plants, which is designed to fulfil the policy of the EC-directives.

A much smaller quantity of composted household waste is used in agriculture. The quantity depends on the collecting system, and today about 10% of the potential is collected, but with an increasing tendency. Compared with sewage sludge the compost has a low content of nitrogen and phosphorus, but some potassium. Neither compost nor sewage sludge makes a significant contribution to the nutrient supply in agriculture.

In recent years the presence of organic contaminants in municipal waste has raised concerns. Some countries (Sweden and Denmark) have provisions concerning this topic, resulting in a reduction of the amount of waste suitable for application to agricultural land.

The requirement for hygienic certification was discussed in the light of the key-note speech of this session. A hygienic treatment has to prevent disease carriers to be transmitted to humans, as well as domestic and wild animals. A hygienic certification may be based on both the treatment procedure and microbiological analysis for select indicator organisms. Denmark, Norway and Finland already have provisions handling this topic.

Session 2 - The effects of waste product application on soil and plants

Olof Thomsson

The subjects presented and discussed in this session were mainly different soil quality aspects, although the title also mentions plants. “Soil micro-organisms are an important factor when determining the impact of anthropogenic activities on soil quality”; it was stated by one of the speakers. Based on a 16-year Swedish trial it was concluded that spreading of sewage sludge on agricultural land affected several investigated parameters, chemical as well as biological. However, no obvious negative effects on soil microorganisms were detected. Danish experiments presented have investigated the effects of organic micro-pollutants in sludge. They showed that different microbial parameters may respond differently to such pollutants and thus change the composition of the soil microbial community. Another Danish study presented discussed the microbial breakdown of DEHP (di(2-ethylhexyl)phtalate). It was shown to adsorb strongly to the organic fraction of the soil. Temperature and oxygen as well as topsoil aggregate size were important factors for its degradation. Continued application of sludge containing the substance will lead to increased levels of DEHP in topsoil due to limited microbial metabolism of the substance.

Soil microbial activity can also be used for sewage sludge treatment. Fields with efficient drainage systems for collection of leakage water, which is led to a sewage plant, have been investigated in a Finnish study. The vegetation of the field may promote the treatment process and also be harvested for energy or fibre purposes. It is said to be a cost-effective method, although the investment costs are quite high.

In order to predict nitrogen dynamics following application of organic waste to agricultural land, mathematical models of soil C and N dynamics may be a useful tool. SOILN_NO, a Norwegian version of the Swedish SOILN model, was evaluated by comparing the model simulation results to results obtained by incubation experiments. It seemed to be adequate for simulation of C and N dynamics in the decomposition of quite different organic wastes.

The concluding discussion of this session also included hygienic aspects of organic waste utilisation in agriculture. The sanitary risk associated with organic waste is obvious since there are organisms, e.g. bacteria spores, viruses, and priones, that one knows can survive in soil for a very long time (at least 100 years). This aspect has to be considered in the discussion of when and how organic waste should/could be used as fertiliser. It was concluded that there is a big need for further research in this field.

Session 3 - Poster presentation

Olafur R. Dyrmundsson

Being a poster session with 10 contributions, a wide range of subjects relating to municipal organic waste was covered. The posters were well presented and attracted a great deal of attention. The main subject matters were composting processes, technical aspects of composting, hygienic considerations and utilization of compost and sludge products. The results reported ranged from detailed laboratory and field experimental findings to information gathered by surveys under practical conditions. There is a generally positive attitude towards composting and recycling of organic waste; the products are commonly being used for gardening in urban areas and for agricultural production in rural areas. This activity is clearly seen as one means of supporting sustainable development. Other problems addressed in this session were: Freezing in household composters, heavy metal contamination, liquid seepage in the bottom of biowaste containers (drainage may be restricted by a compressed peat layer), lack of information on composting and the usefulness of the products. This calls for better general education based on sound scientific principles. References were made to both aerobic and anaerobic processes as well as biogas production in addition to the production of composts for soil improvement. Amongst technical developments reported were large-scale flat bed composting instead of the traditional windrow composting system. On the whole this session highlighted several positive developments and efforts to advance present knowledge on the treatment and use of municipal organic waste.

Session 4 - Waste quality

Jouko Sippola

The theme of the session is very central for waste utilisation in crop production. Apart from heavy metals and organic contaminants the papers concentrated on changes occurring during the composting process and maturity of the composts. Maturity is of great importance when the compost is used before sowing or is mixed into growth substrates. Immature compost may have harmful effects on the soil environment and plant growth. Therefore maturity tests, which are used to predict the possible negative impact of the compost, are both numerous and varied. In a survey among Nordic compost producers the most commonly used maturity parameter was length of treatment time followed by temperature measurements. Several chemical parameters were also used in many cases. Plant assays were used by less than one third of the composting facilities. However, based on a literature review plant assays were considered to provide the most valuable information on compost maturity, and it was suggested that it should be used as a first action when evaluating compost maturity. Mature composts of different origin vary in their chemical, physical and microbiological properties. Due to biological activity antibiotic substances are produced. These suppress the growth of phytopathogenic fungi and micro-organisms, making such composts very useful. Therefore microbiological parameters are also recommended for evaluation of compost quality.

Other experiments reported showed that despite differences in the method of composting the product quality may be similar in some respects, but differ in others. Composting straw and pig slurry in a box system with natural aeration resulted in higher temperature for a longer period compared to a reactor system with forced aeration. This resulted in higher losses of C and N in the box system, but the C/N ratio in the final product from both systems was similar. When composting sewage sludge with wood chips in a tunnel composting system, changing the aeration rhythm affected microbial activity profoundly and affected concentrations of water soluble nutrients, but did not affect the hygienic status of the compost.

Session 5 - Use of waste

Olafur R. Dyrmundsson

Although only three papers were presented in this session they gave a useful overview of the use of human urine, household compost and sewage sludge under a range of conditions. This included information on plant nutrient supply, quality aspects and plant growth. Methods of application were addressed in all the papers. In two of them critical levels of application rates were demonstrated in relation to the leaching of N and P. Application of sewage sludge on a poor gravely soil in Iceland indicated that ploughing-in was superior to surface spreading. These papers and the discussion which followed clearly outlined the potential value of municipal organic waste as a plant nutrient. However, some questions remained unanswered giving rise to speculation and, presumably, suggestions for further research.

Session 6 - Principles of systems for waste treatment
Torleiv Næss Ugland

The environmental authorities recommend that organic wastes from society should be recycled to agriculture as a source of plant nutrients. To improve the interest among farmers, and to improve the environmental benefits, a technology and management system that allows the farmer to process the waste anaerobically on the farm was discussed. The system has been used with good results in processing sludge, and a new system for handling food waste is now being developed in Norway. An overall goal is to put the farmer in the centre, to make a more cost-effective system, and to make a closer link between the consumer/waste-producer and the end-user of the waste products.

In conventional sludge treatment plants it may be a problem to find a suitable way of recycling phosphorus from filter media. It was found that materials like opoka and crystalline slag containing Ca were the most suitable of the investigated filter materials from an agricultural point of view, since they possessed high P-sorption capacity and the sorbed P was highly plant available.

Urine separation is looked upon as an interesting solution for waste treatment in Sweden in the future, because nearly 100% of the nitrogen and 50% of the phosphorus is bound in the urine fraction. Urine separation may for this reason be an alternative to the building of wastewater treatment plants with nitrogen removal.

In Finland there is a lot of interest in composting toilets for permanent houses. This is due to large areas with scattered settlement, and problems with pollution risks to groundwater due to faeces from toilets.

J. Petersen & S.O. Petersen (eds.) 1999. Use of municipal organic waste
Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
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Legislation in Denmark and nutrient value of waste products

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Abstract

The legislation in Denmark concerning the agricultural use of sewage sludge and composted waste is based on council directives of the European Community (EC, formerly the EEC) which aim to prevent adverse environmental effects and to ensure a sensible recycling of nutrients. In Denmark the environment is protected by maximum thresholds for heavy metals and organic contaminants in the waste products suitable for application to agricultural land. Furthermore there is a threshold for the application rate, both in terms of dry matter and phosphorus. The legislation specifies when and how to apply the waste. The common method is spring application followed by ploughing before sowing of spring cereals (typically barley).

Sewage sludge is the main waste product (140.000 tonnes of DM/year). Different types of compost are produced in small quantities only and are not commonly used in agriculture. The level of phosphorus in Danish soils is in generally high and the application of phosphorus in sewage sludge has to be considered only as fertilization for maintaining the nutrient status of the soil. Therefore the application rate is limited by the phosphorus threshold, which corresponds to about 1.5 t DM/ha. Due to the low application rate combined with the low content of plant available nitrogen and the brief growing season for spring barley, the nutrient value of nitrogen is moderate. The potassium content in sewage sludge is very low and is not included in the fertilizer budget.

Production of waste and quantity of nutrients

The aim of EC-directive 91/271/EEC (Anonymous, 1991) is that more than 80% of the nitrogen and phosphorus has to be removed from the urban waste water at the treatment plants. In modern plants this aim is achieved by nitrification-denitrification of nitrogen and precipitation of phosphorus by Al or Fe (Anonymous, 1994). Another EC-directive 86/278/EEC (Anonymous, 1986) prescribes that the nutrients in the sludge 'shall be used in such a way that account is taken of the nutrient need of the plants (crops) and that the quality of the soil and of the surface and ground water is not impaired'. Further, 'where sludge is used on soils of which the pH is below 6, Member States shall take into account the increased mobility and availability to the crop of heavy metals'. In Denmark these aims are laid down in statutory order no. 823 (Table 1) (Anonymous, 1996a).

Table 1. Thresholds and cut-off values (from 1 July 2000) (Anonymous, 1996a)

	Waste		Soil
	ppm DM	ppm P	Ppm DM
Threshold values for metals and heavy metals in waste, and soil quality criteria			
Cd	0.4	100	0.5
Hg	0.8	200	0.5
Pb	120	10,000	40
Ni	30	2,500	15
Cr	100	-	30
Zn	4000	-	100
Cu	1000	-	40
Cut-off values for organic contaminants in waste			
LAS Linear alkylbenzene sulphonates	1300	-	-
ΣPAH Polycyclic aromatic hydrocarbons	3	-	-
NPE Nonylphenol	10	-	-
DEHP Di(2-ethylhexyl)phthalate	50	-	-

About 70% of the sewage sludge, corresponding to 104.000 tonne of DM, is within the threshold values for heavy metals and therefore a suitable potential for re-circulation (Anonymous, 1998d). More than 94% of this potential is applied to agricultural land.

The potential for collection of organic household refuse for composting is 350.000 tonnes (Anonymous, 1998c). Today only 10% of this potential is collected and supplied to about 20 central composting plants (Domela, 1996). The 34.000 tonnes of organic household refuse, corresponding to about 11,000 tonnes of dry matter, is composted together with 43.000 tonnes of garden/park waste, which results in 21.000 tonnes of compost (I. Domela, pers. comm.). Less than 1% of this type of compost is applied to agricultural land. In contrast, about 19.000 tonnes of compost based on garden/park waste, corresponding to 9% of the total amount of compost, is applied to agricultural land (Domela, 1996). Compost is mainly returned to private gardens and today there is no connection between organic household refuse collected for composting and re-use of compost in agriculture. The threshold values for heavy metal may be observed by careful sorting at source (Kjølholt *et al.*, 1998).

An overview of quantities of dry matter and nutrients in types of organic waste in Denmark is shown in Table 2. The quantities of nutrients in organic waste are not substantial compared with the quantities of nutrients in animal manure and mineral fertilizer. The sludge from fish farming is not suitable for application to agricultural land due to excess of the heavy metal thresholds. About 85-90% of the total amount of N and P in sludge from the industries originates from two major industries. The contents of heavy metals and organic contaminants are low in these two types of sludge.

Table 2. Quantities per year of dry matter and nutrients in organic wastes in Denmark. Values in brackets are assumptions

Source	Dry matter [1000 tonnes]	Nitrogen [tonnes]	Phosphorus [tonnes]	Potassium [tonnes]
Vegetable	76	1,500	275	1,700
Fish farming	5	70	40	4
Animal	9	320	100	45
Household 1)	11	100	20	35
Sewage sludge 2)	104	4,600	3,400	(300)
Industries	109	2,260	1,760	2,200
Animal manure 3)	(4,400)	201,000	49,000	156,000
Mineral fertilizer 4)	1,273	283,000	22,000	86,000

1) Collected for composting, corresponding to about 10% of the total potential (Anonymous, 1998d).

2) Suitable for application to agricultural land corresponding to 70% of the total amount (Anonymous, 1998c).

3) Total (Knudsen, 1997).

4) Total consumption (Anonymous, 1998b).

The two following sections concerning the provision of agricultural use of organic waste may be divided into conditions and restrictions for application. This paper is based on legislation valid from July 1st 2000.

Conditions for application

The content of heavy metals in the waste has to be below the threshold values. The treatment plants have the option of choosing one set of the thresholds values related to either DM or P (Table 1). Application is not allowed when the soil quality criteria for the field are exceeded (Table 1).

The potential amount of sewage sludge suitable for application to agricultural land may in a few years be reduced to 35% of the total amount of sewage sludge, due to the cut-off values for organic contaminants (Anonymous, 1998c). This change is not only due to the cut-off values being exceeded, but also due to increased analytical costs caused by accredited sampling and the complicated methods of analysis. The results may cause the batch to be rejected for application to agricultural land. This element of uncertainty and the increased administration may change the disposal procedure. The threshold values have to be observed for each waste product in a mixture (Anonymous, 1996a), which is based on the point of view that the solution to pollution is not dilution.

Furthermore the waste water treatment plants are required to have a storage capacity of more than nine months of operation, and each year the treatment plants have to make arrangements with farmers to receive the waste (Anonymous, 1996a). The farmers have to include the nitrogen value of the waste in the fertilizer account, which is also subject to legislation (Anonymous, 1998a).

Restrictions in application

The principal regulation is a maximum application rate of 7 tonnes of DM/ha/year or as a mean of 10 years (Anonymous, 1996a). For parks and forests the corresponding restriction is 15 tonnes/ha/year or as a mean of 10 years. These rates are often reduced by the restrictions for nitrogen and phosphorus application. The maximum nitrogen rate is 250 kg N/ha/year (presumably reduced to 210 before year 2000), and simultaneously the maximum phosphorus rate is 30 kg P/ha/year or as a mean of 3 years. Both restrictions are valid for the total application of nutrients in waste and animal manure. Therefore, an application of about 30 tonnes/ha of animal slurry leaves no space for application of waste. Hence it follows that waste application will not take place at intensive livestock husbandries but only on farms specializing in plant production.

Due to physical and engineering limits for how small the application rate can be, it is most likely that the maximum application rate will be used at each application. Furthermore the application costs per tonne of waste will be minimized. In the light of the average content of nutrients in sewage sludge and composted household waste, the maximum application rate has been calculated (table 3).

Table 3. Mean application rate of dry matter, nitrogen, phosphorus and potassium in two waste products. The limiting parameter is underlined

Waste type and application rate/frequency	Rate [ton DM/ha]	Total-N [kg/ha]	Mineral-N [kg/ha]	P [kg/ha]	K [kg/ha]
Sewage sludge, max. each 3rd year	3	120	35	<u>90</u>	10
Composted household waste, max. each 2nd year	12.5	<u>250</u>	25	40	125
Composted household waste, each year	<u>7</u>	140	15	20	70
Spring barley			100-130	20	50
Beets			120-180	35	150
Potatoes			120	30	140

Options for crops

Based on the sanitary considerations, the legislation (Anonymous, 1996a) prescribes how and when the waste can be applied and restricts the options for crops.

More than 80% of the sewage sludge is stabilized anaerobically, aerobically or by the addition of lime before application to agricultural land (Anonymous, 1998d), which has to be followed by incorporation within 12 hours (Anonymous, 1996a). Hence the sludge has to be applied to land without crops, and for the following 12 months only the cultivation of cereals for grain production, crops for use in industrial fodder production and non-food crops is allowed.

The same restrictions apply for stabilized composts regarding the options for crops, but without the time restrictions. If the household waste passes through a controlled composting process (55°C for 2 weeks) or a biogas plant, then the requirement of incorporation and the limitations in the options for crops are omitted.

The application of waste before sowing of winter crops (wheat, rye or rape) is restrained by the requirement to include the nitrogen-value in the fertilizer account. For sewage sludge the options will be restricted to spring sown cereals, mainly barley, in which incorporation may be carried out efficiently by ploughing before sowing the seed. When using compost it is more difficult to predict the choice of crop. Crops with a high requirement of potassium (beets, potatoes, whole-crop for silage or vegetables) will be obvious choices when using household waste that have passed a controlled composting process.

Fertilizer effect

The fertilizer effect of phosphorus and potassium is difficult to demonstrate in the particular growing season due to the frequently high nutrient status of Danish soils. Fertilization of phosphorus, but also potassium, has to be regarded as maintenance of the soil nutrient status, and the application rate has to equal the removal by the crop. Spring barley cultivated in a crop rotation of mainly cereals has a requirement of about 20 kg P/ha and 50 kg K/ha. These requirements may be met by an annual application of composted household waste, but only the phosphorus requirement may be met by application of sewage sludge (Table 3).

The fertilizer effect of organic manure is normally due to the nitrogen. But by nitrification-denitrification at the waste water treatment plants nitrogen is removed, and in this way the level of plant-available nitrogen in the sludge is reduced. The remaining nitrogen is mainly present in organic compounds with a low turnover. Likewise, the great majority of nitrogen in compost is present in organic compounds. The part of mineral nitrogen, which may be taken up by the crop, is often just 10 and 20% of the total nitrogen in compost and sewage sludge, respectively.

In Denmark spring barley has a mineral nitrogen demand of 110-130 kg/ha depending on region, soil type and previous crop (Anonymous, 1998a). The nitrogen effect of waste is often moderate (Petersen, 1996) due to the low content of mineral nitrogen, the low application rate and the brief growing season of spring barley. Therefore the use of wastes has to be combined with the application of a considerable amount of mineral fertilizer.

The application of organic manure/waste in crops with a relatively brief growing season will increase the risk of increased nitrogen leaching due to increased mineralisation after harvest (e.g. Petersen, 1996). The nitrogen leaching may be counteracted by under-sown grass in the main crop (spring barley).

Economical value

The purchase of nutrients in mineral fertilizer equivalent to the amount in Table 3 results in a value of 270 and 70 DKK/tonne DM for sewage sludge and compost, respectively. However, the value is not determined by nutrient content only, but also depends on transport and delivery conditions. In contrast, the expenses for alternative disposal of the waste, e.g. incineration, are considerable and amount to a total of 1250-2500 DKK/ton DM (Anonymous, 1996b).

Conclusion

In Denmark, the total quantity of nutrients in organic waste is moderate compared with the total use of plant nutrients in agriculture. Sewage sludge will mainly be applied at farms specializing in crop production, and the application may take place in the spring before sowing of spring barley. Supplemental mineral nitrogen is required to comply with the demand of a cereal crop when using organic waste.

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The current situation concerning the use of municipal organic waste in Iceland

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Introduction

As in other Nordic countries there is a clear trend in Iceland, linked to national policy, towards sustainable development. Although Iceland is not a member of the European Union the Icelandic Government is obliged to implement several EU directives being a member of the European Economic Area, as is Norway. Consequently, steps are being taken to implement both national and international regulations relating to the management, treatment and use of municipal organic waste. However, these matters have not received much attention until recent years. With growing numbers of the 270.000 people living in urban areas, particularly in the City of Reykjavik and adjacent towns in SW-Iceland, there is clearly a need to seek ways and means of handling waste products more efficiently and in harmony with environmental criteria. The fact that Iceland is an island of 103.300 km^2 just south of the Arctic Circle surrounded by rough seas, and with low population density, has indeed influenced general attitudes and official policy but certain developments witnessed during this decade indicate changes likely to occur in the immediate future.

Small scale composting

While composting of vegetable matter has been practised on an individual basis in urban gardens for several years, composting of organic household waste began on a trial basis in a few municipalities soon after 1990 (Gíslason, 1998). Several types of compost boxes and bins, both insulated and uninsulated, have been tested with variable results. However, these trials have shown that with experience, which takes into account factors such as the dry matter content of the waste and the litter amendments, satisfactory results can been achieved.

This development led to a larger scale trial of organized collection and composting of organic household waste in 1994-1995 in Hafnarfjörður Town near Reykjavík (Jónsson, 1996a). The composting took place in open-air windrows aerated by a tractor-mounted windrow turner. The results were generally satisfactory, and it was of interest to note the willingness and positive attitude of those participating in the trial. Unfortunately, source separation of household waste on a routine basis is still only carried out on a small scale in a few municipalities, all outside Reykjavík. A good example of a successful collection and composting of source separated organic household waste is found in the Hvanneyri

Community in W-Iceland. A drum composter (ALE trumman, SWEDEN) has been in operation since February 1997 handling some 150-250 kg of kitchen waste from a population of 260 fortnightly (Brynjólfsson, 1998).

Large scale composting

At the beginning of this decade decisive steps were taken to collect and recycle solid, inorganic waste materials such as bottles, cans and scrap metals throughout the country. In 1994 the company SORPA, responsible for the collection and treatment of all waste materials, except sewage, in Reykjavík and neighbouring towns, pioneered a garden waste composting project. Until then all garden waste in the area, estimated at 6.000-9.000 tons per year, had been tipped into landfills together with household garbage. This project was in operation at the same time as the household waste trial in Hafnarfjörður Town (Jónsson, 1996a), and the same windrow method was used. The main ingredients of the compost were grass and tree clippings from private and municipal gardens supplemented with horse manure available locally. The windrows were approximately 1-1.5 m high and 2-3 m wide and in most cases heating above 60°C was achieved and maintained for 12-14 days. Normally the composting process was completed 10 weeks after the raw materials had been mixed in the windrows.

This large scale composting trial proved so successful that composting of all garden waste available in the Reykjavík area has become a standard practice using the method developed during the course of the project.

Sewage sludge

The majority of the Icelandic population lives in coastal areas and adjacent valleys, while the central highlands are uninhabited. The general practice in urban areas has been to pipe the raw sewage into the sea. Attitudes are changing, and official authorities are realising the growing need for anti-pollution measures, particularly in the most densely populated SW part of the country. Thus, the city of Reykjavik has invested heavily in sophisticated pumping stations on the shore which may be expanded so as to serve as sewage plants for the treatment of raw sewage in the future. The fact remains, however, that sewage sludge is not yet available for agricultural use, and research in this area is in its infancy in Iceland.

Compost and sewage sludge utilization

As indicated above, composts from municipal organic waste have been successfully produced in Iceland in recent years, mainly from garden waste, while kitchen waste is also being included in some localities.

The use of municipal composts from garden waste has now become quite widespread in the Reykjavik area. Trials carried out during the first few years indicated that the product, called „molta" in Icelandic, can be applied successfully as a soil improver and fertilizer raising the NPK levels substantially (Jónsson, 1996b). A survey carried out amongst members of the Horticultural Society of Iceland showed that after the summer of 1995 a total of 83% of those participating expressed satisfaction with the compost, giving further evidence of its value for growing under a range of conditions.

The compost produced from the household waste in the drum composter in the Hvanneyri Community is used for organic vegetable growing trials. There are already indications of a high N content in that compost.

Since 1993 Mógilsá Forestry Research Station has pioneered trials using municipal sewage sludge in tree plantations in S-Iceland. Results from these trials will be reported at this seminar (Ásgeirsson, 1998).

Conclusions

It is clear that composts from garden and household waste as well as sewage sludge are potential sources of plant nutrients, for both growing crops and land reclamation under Icelandic conditions. Such resources may have an important role to play in efforts to revegetate barren land and to stop soil erosion (Dýrmundsson, 1993). Furthermore, these potential sources of plant nutrients could, if properly and efficiently utilised, supplement the farmyard manure available as a supply of organic fertilisers. Lack of N, and to a lesser extent of P in an organic form, clearly limits large scale conversion to organic agriculture in Iceland (Dýrmundsson, 1996 & 1997). In the absence of statistics on amounts and availability of municipal organic waste in the country, let alone estimates of their value in terms of safely usable products, speculations only will have to suffice at this stage.

Finally, policy measures and growing public awareness of the growing need to protect the environment will greatly influence future development in Iceland as in other countries. Several EU directives relating to pollution, waste and its products were incorporated into Icelandic law with the Anti - Pollution Regulation Nr. 48/1994, with subsequent amendments. Important steps have already been taken to fulfill these statutory requirements. However, as for the use of municipal organic waste and research on this resource is concerned, Iceland is lagging somewhat behind Denmark, Finland, Norway and Sweden.

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J. Petersen & S.O. Petersen (eds.) 1999. Use of municipal organic waste
Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
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The current situation concerning the use of municipal organic waste in Norway

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According to the Ministry of Environment, the overall goal with respect to municipal organic waste is to solve the waste problem in such a way that it leads to as few problems as possible for human beings and the environment, while occupying only a limited part of community resources. Organic waste from the households has to be regarded as a valuable resource, and used preferentially in agriculture for fertilising and soil improvement.

Wastewater treatment in Norway

Due to political decisions, a lot of wastewater treatment plants have been built during the last 20 years, and Norway has now about 2050 such plants. The major part of the wastewater undergoes chemical treatment, and in plants that treat wastewater from larger cities, both chemical and biological treatment is used. The total amount of sludge produced in Norway in 1995 was 92 900 tons dry matter.

Fifty five percent of the population in Norway is situated in the coastal area from the Swedish border to Kristiansand in the south. Due to the connection with the North Sea, it has been a political goal to clean the wastewater in this area for as much phosphorus as possible. By the use of both chemical and biological treatment the average removal of phosphorus reached 89 % in 1995. The total retention of phosphorus in the wastewater treatment plants was 1495 tons that year.

To reach this goal, the wastewater plants use different phosphorus precipitating chemicals, such as lime and salts with iron or aluminium. According to a research project carried out as a co-operation between The Agricultural University of Norway, The Norwegian Crop Research Institute and 15 wastewater plants, the use of phosphorus precipitation chemicals made the phosphorus nearly non-available for the plants. The binding of phosphorus was most severe when using aluminium and less severe when using lime. (Ugland *et al.* 1998).

The Norwegian pollution authorities consider sludge a resource that ought to be used as a fertiliser or soil conditioner. The objective is to return 75 % of the sludge to agricultural - land. Sludge used in agriculture requires a declaration of hygienic parameters, stability and content of heavy metals. The possibility for using sludge in agriculture according to heavy metal content is shown in table 1. Quality class 0 is a proposal still being evaluated by the authorities, and class 0 and 1 concern only compost. The use of sludge in agriculture is also limited to cereals, green fodder and establishment of ley – and it is not possible to grow vegetables until 3 years after sludge disposal. Sewage sludge may not be spread without being hygienically treated according to treatment criteria, and samples have to be taken for bacteriological tests once a month. The samples must not contain salmonella bacteria or parasite eggs, and the content of thermo-tolerant coliforms should be below 2500 per g DM.

Table 1. Limits for heavy metal content in organic waste to be used in agriculture and for the landscape sector in Norway. Mean values of heavy metal content in Norwegian soil (mg/kg DM).

	Cd	Pb	Hg	Ni	Zn	Cu	Cr	Potential use
Quality-class 0	0,4	40	0,2	20	150	50	50	Suggested new class, free use of compost according to agronomic valuation.
Quality-class I	0,8	60	0,6	30	400	150	60	Agriculture, private gardens and landscape sector. Max. 40 t DM/ha/10 years.
Quality-class II	2,1	80	3	50	800	650	100	Agriculture, private gardens and landscape sector. Max. 20 t DM/ha/10 years.
Quality-class III	5	200	5	80	1500	1000	150	Only landscape sector. Max. 20 t DM/ha/10 years.
Mean values in soil	0,22	23,9	0,05	21,1	63,9	19,2	27,1	

The authorities have done a lot of work to reduce the content of heavy metals in sludge during the last 15 years. The result is a 50-90% reduction in the concentration of Cd, Cr, Pb, Ni, Zn and Hg, but only a 3% reduction in the content of copper.

In 1995, only 48% of the sludge was used in agriculture. One important reason is the low demand for extra organic manure in the northern and western part of Norway, where there is mainly grassland, a huge surplus of animal manure and much humus in the soil. In the large barley-districts of Sør-Trøndelag the natural content of nickel in the clay is so high that use of sludge is prohibited. Also a certain reservation towards the use of sludge is present among farmers in Norway, due to insecurity about consumers' opinion and the uncertainty about possible regulations and restrictions in the future, that can represent a problem for farms that have used sewage sludge. The use of phosphorus precipitation chemicals has also reduced the value of sludge as an actual organic fertilizer.

Table 2. Content of creosols, nonylphenol, phthalates, LAS, dioxins, PCB and PAH in Norwegian sewage sludge, compost and animal manure (mg/kg DM (Paulsrud *et al.* 1997).

	Creosols	Nonylphenol	Phthalates	LAS	Dioxines	PCB	PAH
Sewage sludge	-	140	60	55	0,06626	0,0422	3,13
Compost	-	-	8	40	0,00367	0,0187	0,62
Animal manure	151	-	5	70	0,0003	0,002	-

The interest in potentially toxic organics in organic waste has increased over the last years. According to a research programme carried out for The Norwegian Pollution Control Authority (Paulsrud et al 1997) in 1989 and 1996/97 it was concluded that the content of toxic organics in sewage sludge and compost was so low that it was found unnecessary to establish limiting standards.

Source separated household waste

There has been a remarkable increase in the source separation of household waste as a waste management system in Norway over the last 5 years. 77% of the population had a system for source separation in 1997, with paper as the most important fraction. The collection of organic kitchen waste is still in its infancy. Only 4% of the population had the possibility of delivering kitchen waste to composting plants in 1995, but this had increased to 16% in 1997. During 1999 it will be prohibited to take organic household waste to landfill sites in Norway. As a consequence local authorities are now building plants to handle 4-500 000 tons organic waste per year.

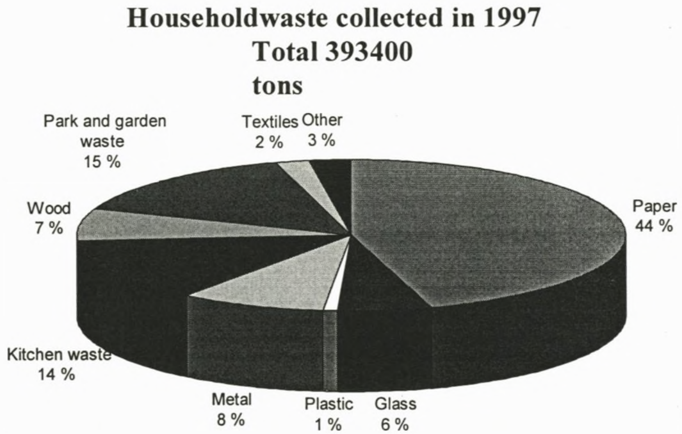

Figure 4. Collection of waste from the households in Norway in 1997 (Statistics 1998a)

Compost will be tested for heavy metals, Salmonella and thermo-tolerant coliforms in the same way as sludge. 40 tons DM/ha may be used during a period of 10 years, if the compost satisfies the demands in class 1 (table 1). The authorities in Norway are now evaluating a new class 0 for compost with especially good quality without restrictions for use in agriculture.

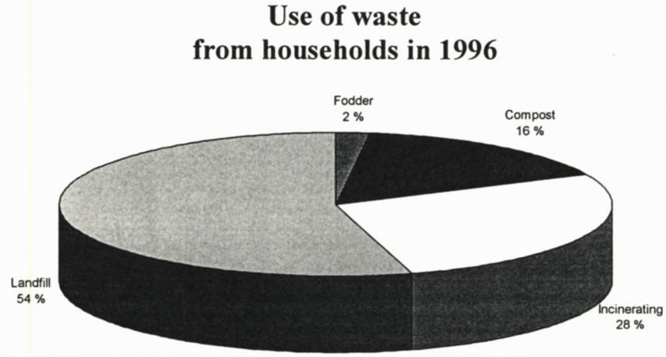

Figure 5. Use of waste from households in 1996 (Statistics 1998b)

The compost from local plants is estimated to constitute 100 000 tons per year. There is a fairly large demand within agriculture and the landscape sector for compost, and also organic farmers are allowed to use some compost of non-organic origin (100 kg total nitrogen/ha/year in compost).

The authorities regulate the sale of raw and processed compost. The compost or products made of compost have to fulfill the demands of the authorities for quality, documentation and declaration before marketing. The products have to fulfill the limiting standards for heavy metals, hygienic quality, maturity, weed seeds and foreign matter, and they need a quality declaration for nutrients, pH-value, liming value, electric conductivity, organic matter, dry matter, particle size, C/N ratio and heavy metals

The challenge for the composting plants in Norway will be to produce compost with good quality and declarations for stability, fertilising value, and low content of heavy metals and organic pollutants. For marketing purposes it would be desirable to go even further and classify the compost for soil improving value and plant protection effect, and develop specially designed products for different markets.

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J. Petersen & S.O. Petersen (eds.) 1999. Use of municipal organic waste
Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
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Danish Institute of Agricultural Sciences report no. 13

The current situation concerning the use of municipal organic waste in Sweden

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Environmental policies

The Swedish parliament has stated that the overall goal for the waste policies is good housekeeping with resources and maintenance of a healthy environment. All resources, both financial and materials, should be well managed with respect to energy, water and soil quality. Already in 1993 a decision was made to prefer reuse before recycling, energy extraction, and landfilling, in the order mentioned.

Amount of total organic waste in Sweden

The total annual amount of easily degradable organic waste generated in Sweden is about 30 million tonnes (Brolin *et al.* 1996). Three quarters of that is animal manure that can be assumed to be 100 % used by agriculture. Organic wastes from the food industry is about one tenth of the total and is to a very large extent used as fodder in the agriculture. Stabilised sewage sludge amounts to about only 3 % of the total, which is equivalent to about 1 million tonnes (Brolin *et al.* 1996). The household organic waste represents between 800 000 tonnes (RVF 1998) and 1 million tonnes (Brolin *et al.* 1996). The distribution between different sources is shown in Figure 1 and some approximate total amount of plant nutrients is given in Table 1.

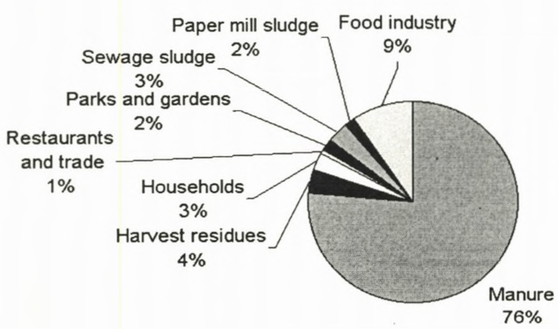

Figure 1. Distribution of different fractions of organic waste in Sweden.

Table 1. Tonnes dry matter and plant nutrients in different organic waste fractions.

Organic waste product	Tonne dry matter/year	Tonne N/year	Tonne P/year	Tonne K/year
Sewage sludge	200,000[1]	6,600[2]	4,800[2]	600[2]
Household org. waste[3]	350,000	7,000	1,050	3,150
Manure[4]	2,786,000	135,000	23,500	124,500

[1] Brolin *et al.* 1996; Pettersson 1992

[2] Pettersson 1992

[3] Brolin *et al.* 1996; RVF 1998 give 800 000 tonnes total, equiv. to 280 000 t dry matter if water content 65%

[4] Brolin *et al.* 1996 (calculated from official statistics, SCB)

As can be seen the municipal part of the organic wastes is quite small. It is, however, important to recycle also these plant nutrients and organic matter when a long-term perspective is considered. For quite a long time it may be possible to misplace these resources and still have a productive system, but at length the accumulated quantity removed will be so large that the productivity of the system will drop.

Sewage sludge

The major part of the sewage sludge in Sweden, 90 % according to Brolin *et al.* 1996, is produced in sewage plants with both biological and chemical treatment. All raw sludge produced in sewage plants is dewatered and almost all is stabilised. 70 % is stabilised by digestion, 15 % by liming, and 12 % by aerobic treatments (wet or dry). Only 3 % of the dewatered sludge is not stabilised (Diedrich 1992).

Only about 30 % of the sewage sludge are nowadays used in agriculture (Brolin *et al.* 1996) but there are large regional variations. In the north of Sweden almost no sludge is used in agriculture, while in some areas in the southern part, e.g. Malmö and Stockholm, a large part is used. Up till now most of the remaining part is deposited in landfills; only a minor part is used for soil construction purposes. Since a discussion about a landfill tax has increased the interest in alternatives to landfilling, experiments with incineration of sewage sludge have been carried out in e.g. Linköping.

Regional sludge quality variations (mostly heavy metal content) and the food industry's negative attitude towards use of sludge in agriculture explain the low rate of utilisation. For example, the dominant dairy co-operative Arla has forbidden all use of sewage sludge on the farms of their suppliers. However, for many years there have been discussions going on regarding the use of sewage sludge. The authorities want more of the sludge to be used in agriculture, and so does the Federation of Swedish Farmers (LRF). In an agreement committed 1994 the Swedish Environmental Protection Agency (SNV), LRF, and the Swedish Water & Wastewater Association (VAV) agreed to promote the recycling of nutrients and organic matter in sludge to agricultural land (Naturvårdsverket *et al.* 1995). The parties also decided to use the existing limits for heavy metal content in sludge (SFS 1993; SNFS 1994) (see Table 2).

Table 2. Maximum allowable heavy metal content in sludge and maximum annual supply for sludge spread on agricultural land.

Metal	mg/kg dry matter in sludge (SFS 1993:1271)	g/ha/year[1] from 1995 (SNFS 1994:2)	g/ha/year[2] from 1995 (KRAV)[3]	g/ha/year[1] from 2000 (SNFS 1994:2)
Pb	100	100	50	25
Cd	2	1.75	1	0.75
Cu	600	600	500	300
Cr	100	100	50	40
Hg	2.5	2.5	1	1.5
Ni	50	50	50	25
Zn	800	800	700	600

[1] Average of a 7-year period, i.e. 7 times the amount can be spread once in 7 years
[2] Average of a 5-year period, i.e. 5 times the amount can be spread once in 5 years
[3] KRAV, a member of IFOAM, is a certifying organisation for organic farming. PLEASE NOTE that only sludge from the sludge separator at the household on the farm can be used, not municipal sewage sludge.

Although there are no scientific evidence that the sludge contains harmful organic substances that are taken up by the crop (Naturvårdsverket 1993) it was decided to be extra careful. To avoid accumulation of long-lived substances in the soil, limits have been set for the content of some organic "indicator" substances in sludge (see Table 3), and rules have been set for which crops can be used for sludge application, as well as a minimum time span between sludge spreading and harvest. Furthermore, it was decided to promote information, discussions and spread of knowledge in the society about how to get a cleaner sludge.

Table 3. Maximum allowable content of "indicator" organic substances in sludge spread on agricultural land.

Substance	Content (mg/kg dry matter)
Nonylphenol	50
Toluene	5.0
Sum PAH	3.0
Sum PCB	0.4

Source separated human urine

There is a growing interest in source separation of human urine in Sweden. The background is the recollection that urine holds the larger part of the nutrients secreted with the excrements and at the same time a very small part of the total sewage water volume (Figure 2). The idea is that by using source separated urine in agriculture one can handle much plant nutrients in a fairly small volume. Furthermore, urine separating toilets use considerably less water than conventional WCs. The urine is transported from the toilet via a separate pipe to a tank where it is stored. Of hygienic reasons it is recommended that it is stored for 6 months before spreading on farmland. At present (autumn 1998) there are two brands of urine separating toilets on the Swedish market. Some 2000-3000 toilets have been installed in Sweden so far. This type of toilet can be combined with any sort of system for handling of faeces and water fractions.

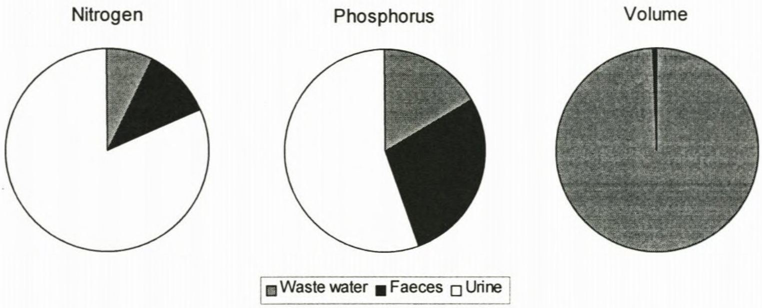

Figure 2. Nitrogen, phosphorus and volume for different sewage water fractions. (pers. comm. Björn Vinnerås)

Urine separation has been investigated in a large research project with researchers from Dept. of Agr. Engineering, SLU (Håkan Jönsson, Björn Vinnerås, Anna Burström and Jan Svensson); Dept. of Soil Sciences, SLU (Holger Kirchmann and Pernilla Kvarnmo); Dept. of Biochemistry and Biotechnology, KTH (Gunnel Dalhammar); Swedish Institute for Infectious Disease Control, SMI (Thor Axel Stenström and Caroline Höglund). The different aspects covered are nutrient content in urine solution, practical experiences with separating toilets, hygiene, environmental effects, fertiliser effects, toxicity for crops, and concentration of the urine solution.

Household organic waste

There has been no formal quality regulation for organic wastes so far. The "sewage sludge agreement" (Naturvårdsverket *et al.* 1995) have been the only rules to look at for a quality measure. Therefore a project has been underway to establish environmental and quality certification rules for organic waste. The first draft of the rules was recently published (Lundeberg *et al.* 1998). These rules agree in most cases with demands that already exist and are relevant for Swedish conditions.

As can be seen in Table 4, only a small part of the municipal organic waste is biologically treated (composted or digested). Of the remaining part about half is incinerated and half is deposited in landfills. It should, however, be noted that the amount given for parks and gardens does not include material composted locally in private gardens or elsewhere in parks. This amount could be fairly high.

Table 4. Total amounts organic waste and amounts biologically treated from different sources in Sweden 1997 (from RVF 1998)

Organic waste from:	Total amount Tonnes/year	Biologically treated Tonnes/year	Biologically treated %
Households	800,000	110,000 [1]	14
Restaurants and trade	180,000	15,000	8
Parks and gardens	530,000	150,000 [2]	28
Total	1,510,000	275,000	18

[1] Including approximated 40 000 tonnes home-composted

[2] Excluding locally composted waste

The trend goes towards more source separation of household organic waste, which in time will probably give a higher utility rate. Today about 15 % of the local communities have extended source separation of household organic waste coupled to central composting or digestion plants. Another third of the local communities plan to introduce this kind of source separation. 60 % of the local communities have regulations on home composting. In year 1997 about 5 % of the households – about 200 000 households – practised home composting of their organic household waste. (RVF 1998)

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J. Petersen & S.O. Petersen (eds.) 1999. Use of municipal organic waste
Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
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The current situation and the legislation concerning the use of municipal organic waste in Finland

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Waste policy

The aim of Finnish waste policy is to support sustainable development by promoting the sensible use of natural resources and preventing hazards and ill-effects caused by waste to people's health and the environment. The legislation of 1994 concerning the use of waste includes provisions on waste reclamation, the organisation of waste management, waste reduction, and soil protection.

Waste policy focuses on waste avoidance. Existing waste has to be reclaimed wherever it is technically possible and costs remain reasonable. Materials recovery followed by energy recovery are conducted if possible; if not, the waste is tipped onto a landfill site.

Municipalities are obliged to organise the transportation of household waste or waste parallel to household waste. Municipalities can also decide for special reasons not to organise municipal waste transportation in certain areas. The holder of waste has to utilise the organised waste transportation unless otherwise agreed with the municipality.

The Council of State has the right to issue national waste management regulations, and, correspondingly, municipalities may issue municipal regulations. Municipalities may also collect a waste charge for the waste management, which should at least cover the expenses caused by the founding, use, and abolishing of treatment sites. As a result of this kind of charging based on the causation principle, waste charges have increased and will continue to increase.

Sewage sludge

In Finland, 23% of the population live outside municipal wastewater treatment areas and therefore treat their toilet waste as best as they can and according to local regulations (oral statement by E. Santala, June 4, 1996). The number of such people is surprisingly high in other European countries also: for example, approximately 10 % of the population in Germany live outside municipal wastewater treatment areas (Kollatsch 1997). Furthermore, a large part of the population spend the holidays or weekends in rural areas.

In areas of scattered settlement, wastewater is usually treated in septic tanks. The annual amount of septic tank sludge, including the waters of closed tanks, totals a million cubic metres. Since the septic tank sludge contains a lot of water, its dry matter content is only 15,000 tons. About half of septic tank sludge is treated in municipal wastewater treatment plants; the rest is tipped onto landfill sites or spread on fields. According to the National Waste Plan Until 2005 drawn up by the Ministry of the Environment, the spreading of septic tank sludge should be discontinued due to the environmental damage caused by the sludge (Anon. 1998).

In 1993, there were altogether 563 wastewater treatment plants serving more than 200 clients and some 5,000 plants serving less than 200 clients in Finland. The annual amount of sewage sludge totals approximately a million cubic metres, its dry matter content being 150,000 tons. In 1996, this amount included 38% anaerobically digested matter, 1% decayed matter, 13% lime-stabilised matter, and 23% unstabilised matter (Statistics of the Finnish Environment Institute). The dry matter content of sewage sludge varies considerably depending on the drying method; at its highest it is approximately 30% (Anon. 1998).

The quality of sewage sludge has been enhanced during recent decades by regulating the amount and quality of hazardous chemicals and waste let into sewers with connection agreements and waste management regulations. Today a major part of sludge meets the target values that have been set for the use of sludge in agriculture.

In the mid-80s, up to 70% of the dry matter content of municipal sewage sludge was used in agriculture and landscaping; the rest ended up in landfill sites. The attitudes of the agricultural population and the more strict conditions regarding the use of sludge during the early 90s has decreased the use of sludge in agriculture. The degree of sludge utilisation has simultaneously reduced. The increase in the composting of sludge and its use in landscaping have not compensated for the decreasing use of sludge in agriculture. In 1994, some 60% of sewage sludge was utilised (Anon. 1998).

The use of sewage sludge in agriculture is regulated by the government resolution on the use of sewage sludge in agriculture (282/94). According to the resolution, the sludge has to be treated before use so that it will not cause health hazards or environmental damage. Its heavy metal content must not exceed the maximum amounts allowed, and it can only be spread on agricultural land where the heavy metal contents do not exceed the maximum amounts allowed (Table 1). Most of the Finnish limitations for sewage sludge are substantially stricter than those in the EU directives for sludge. The stricter limitations have been justified by the cold Finnish climate and the sensitivity of water bodies to pollution. We also want to protect the soil, since it remains relatively clean.

Table 1. The maximum amounts of heavy metals allowed in sludge and agricultural land. (Government resolution 282/94)

	Maximum amount (mg/kg/dry matter)						
	Cd	Cr	Cu	Hg	Ni	Pb	Zn
Sludge	3,0	300	600	2,0	100	150	1500
Agricultural land	0,5	200	100	0,2	60	60	150

Sewage sludge may only be used on agricultural land where people grow corn, sugarbeet, oil plants, or plants that are not used for food or feed. Sludge may only be spread on lawns when setting up a lawn with a companion crop and after mixing the sludge carefully with earth. On agricultural land where sludge has been used one may not grow potatoes, root crops, or vegetables for five years after the application of sludge.

Sewage sludge is usually composted heaped in a field. The quality of composting has varied; there have been odour problems and shortcomings in the treatment of seep waters. Finding suitable places for composting has also been difficult. What is more, sludge basins in landfill sites have sometimes filled up and caused the dams to collapse. In order to solve the problems in the treatment of sludge we need to develop alternative ways of composting (Yli-Kauppila 1998).

The use of compost products made of sludge is regulated by the decision on fertiliser products made by the Ministry of Agriculture and Forestry (46/94). The monitoring of the production, marketing, and control of fertiliser products, including soil conditioners, rests with the Ministry of Agriculture and Forestry, while the Plant Production Inspection Centre monitors compliance with the statutes and regulations enacted on the basis of the law on fertilisers.

In Finland, the utilisation target of sewage sludge is 70% of the total amount of sludge by the year 2005. Increasing the utilisation of sludge should be based on the continuous and reliable enhancement of the quality of sludge, the development of new ways of utilisation, and the increasing use of sludge in agriculture (Anon. 1998).

Toilet waste

The water closet is the most significant single water pollutant in households. With modern technology, wastewater treatment plants are able to decrease the phosphorus load and biological oxygen consumption by more than 90%, whereas the nitrogenous load reduces by 35% on average (Rontu & Santala 1995). If the household uses a composting toilet, all the wastewater is 'grey water', which requires substantially less treatment than 'black water' and can usually be treated safely on the site.

Between 3,500 and 4,000 composting toilets are sold annually in Finland. Most of them are situated in the privies of summer cottages; about a fifth of them are acquired for round-the-year use in one-family houses, mostly in areas of scattered settlement. There are also home-made models, the number of which is difficult to estimate. As environmental awareness has gained foothold, the demand for composting toilets has grown during recent years. They are particularly recommended in areas of scattered settlement, where wastewater is absorbed into the ground and forms a pollution risk to nearby water bodies. The lack of fresh water in households outside water distribution systems also provides reasons for interest in composting toilets (Malkki & al. 1997).

The usual way to utilise the composted toilet waste is to spread it on the yard under bushes or on wasteland. People rarely use it as a soil conditioner or fertiliser for vegetables, due to the biased attitudes towards composted faeces. With separating toilets, the urine is not usually collected and used as a fertiliser in the garden, but instead led into grey waters and absorbed into the ground through a septic tank or a sand filter. (Malkki & Vanhala 1994; Malkki & al. 1997).

In Finland, constructing a composting toilet in a population centre is subject to licence, but in areas of scattered settlement no licence is needed. Instead, water closets in areas of scattered settlement are subject to licence, as are the toilets, which lead substantial amounts of seeping liquids into the drain.

Municipal environmental committees are responsible for the general legality control of waste management. Waste management authorities and environment committees deal with matters concerning wastewater, and therefore the conduction of waters from composting toilets or the post-composting process of waste is under their control. Building authorities may interfere with the toilet's location or other details during the construction phase, while public health authorities control the condition, maintenance, and hygiene of the toilet.

Municipal waste management regulations may include separate regulations for the treatment of toilet waste. In populated areas, toilet waste may only be processed in composters. Before starting the composting, one has to notify the public health authorities, who provide the necessary instructions. In areas of scattered settlement, however, the composting of toilet waste is permitted without a licence.

Organic household waste

It is estimated that approximately 2.1 million tons of municipal waste is produced annually in Finland; about a third or, at the most, half of it originates from households, and the rest is produced by services and businesses, small industries, and building sites. During this century, the consumption of households has increased manifold, while changes in the structure of this consumption have resulted in the production of increasingly larger amounts of waste. The number of households has also grown as family sizes have diminished (Anon. 1998).

The accumulation of household waste has been estimated to total 180 kg/person/year in populated areas and 120 kg/person/year in areas of scattered settlement; a good third of the waste is biowaste (Figure 1), some 40% is paper and cardboard waste, and some 7% is glass (Anon. 1998).

In Finland, the municipal waste management regulations oblige people to separate their wastes. However, since the implementation of separation is poorly monitored, reclaimable waste is still taken to landfill sites with other waste. Independent composting is regulated by the municipal waste management regulations, but the practice varies: in some municipalities one needs a municipal licence for the composting of biowaste, and in others a simple notification is enough.

The reclamation rate of organic household waste varies regionally, but the national average is below 10%. The rate will increase, however, once the separate biowaste collection that has now begun in several municipalities starts to pay off. For the time being, a major part of reclaimable organic household waste is treated in thermocomposters in people's own houses, and the end product is used for their own purposes.

Nowadays biowaste treatment in several towns with large waste accumulations is concentrated. There are 17 waste plants or sites for concentrated biological reclamation or treatment in Finland. Organised biowaste treatment has been active for the longest time in the Helsinki Metropolitan area and Tampere. In the Vaasa region only, biowaste is treated through anaerobic digestion, while in the Turku region there is an incineration plant for municipal solid waste.

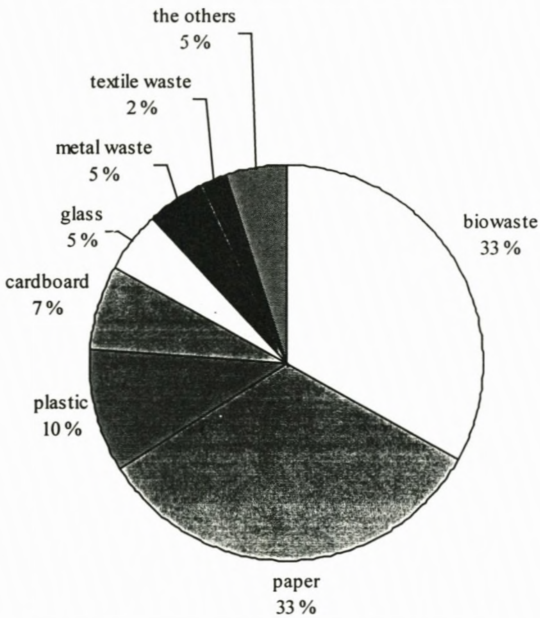

Figure 1. The average composition of household waste in populated areas (Anon. 1998).

The separate biowaste collection and composting covered the whole Helsinki Metropolitan area (about 900,000 people) in 1998. According to the waste management regulations, residential buildings in the area have to separate their biowaste if there are at least 10 apartments in the building or if more than 50 kg of biowaste is produced weekly. Moreover, smaller buildings have to compost biowaste as far as possible on their own plots. Separately collected biowaste is composted in the composting plant at the Ämmässuo landfill site. A private enterprise processes the resulting earth into complete products and sells them to households and towns, which use earth compost in planting arrangement.

Organised treatment of the biowaste of town areas will be necessary in the future. In the treatment plants, the biological process can be conducted in a way that meets the requirements of environmental protection and hygiene. The network of treatment plants will complement the small-scale composting of biowaste, which is a better method in areas of scattered settlement (Anon. 1998).

About 90% of organic household waste is taken to landfill sites with other waste (Figure 2). In the future, biodegradable material and waste suitable for energy recovery should no longer be taken to landfill sites, since it may cause substantial environmental and hygienic problems.

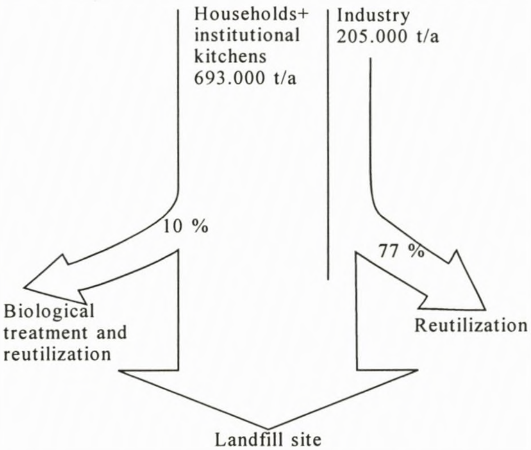

Figure 2. The amount, treatment, and reclamation of organic household and industrial waste (Anon. 1998)

In Finland, there were altogether 390 landfill sites for municipal waste in 1996; in international terms, the number is high with regard to the population. A major part of the sites are small in size and their equipment, maintenance, and supervision do not meet the EU requirements. The quality of landfill sites will be enhanced according to the government resolution (861/97), and their number will be decreased to 50–80 by the year 2005.

Conclusion

Biowaste reclamation should be increased considerably in Finland. The quality of sewage sludge has improved during recent years, but its use in agriculture has decreased. The composting of organic household waste is still based on independent small-scale composting, with the exception of a few cities and the Helsinki Metropolitan area, where the biowaste of nearly 900,000 people is composted concentratedly in waste plants.

In order to increase the demand for sewage sludge and composted biowaste, their quality must be improved to meet the requirements. Quality control has to be strict to ensure the consumer that the use of a composted product will not cause problems.

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Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
Session 1
Danish Institute of Agricultural Sciences report no. 13

Hygienic aspects of organic waste use in agriculture

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Abstract

By utilising biowaste, its potentially negative impacts on the environment can be minimised in a profitable way. However, a variety of pathogens can be found in organic waste. As a result, its use as fertiliser on arable land can inadvertently facilitate the spread of infectious diseases affecting human and animals. In addition, new routes of disease transmission are created between animals and humans. Furthermore, the disease situation is always very dynamic; thus there is a risk that pathogens recently introduced into a country or an area will be further spread in connection with organic waste applications. In the Nordic countries both the very good health status of our domestic animals and our society's generally high sanitary standards must be maintained.

Pathogens in organic waste vary considerably in terms of their survival time, population growth potential in the environment and infectious dose. Some of these pathogens cause zoonoses, such as EHEC and salmonella, and may thereby be transmitted between humans and animals. Some cause epizootic diseases in livestock, such as classical swine fever, and are thereby considered of enormous importance for animal production.

Pathogens from organic waste spread on farm land can be transmitted by vector animals, to surrounding populations of animals and humans. Vectors may carry pathogens on their bodies, or in their gut. Infection may also result directly when animals are grazing on or passing over the land. Other ways of transmission of pathogens may be on dust particles transported by wind, in surface water, and in food or feed harvested from the treated land. It can thus be very difficult to trace a disease outbreak in domestic animals or humans back to biological waste spread on farmland.

The effectiveness of a hygienic treatment of organic waste often depends on the temperature, treatment duration, pH and oxygen availability. The reduction of pathogen numbers varies greatly depending on the method used and species of pathogens involved. Goals of a suppressive treatment can also vary: For some serious or exotic pathogens a complete kill-off is desirable, whereas for many others a marked reduction is sufficient. Biological treatment methods such as anaerobic digestion, aerobic composting and long-term storage are frequently used today. End-product hygienic status can be evaluated by monitoring the treatment process or by microbiological analyses. Such analyses may focus on indicator organisms or on the pathogens themselves. Although analyses based on indicator organisms

tend to be simple and inexpensive, the results however are limited since their capability of survival may differ from the pathogens. On the other hand, the direct analysis of pathogens can be complicated and costly, but is more reliable. Often it is easier to specify a process rather than an end-product. As an extra safety precaution the use of a particular end-product on farmland may be restricted. Due to such restrictions, there may be little demand for them. There is still much to learn with regard to treatment efficacy and the fate of pathogens after spreading treated waste on farmland.

In Sweden, there is no biosecurity-related legislation regulating the use of organic waste as fertilisers. By contrast, such legislation has been passed in several other countries, including Norway and Denmark. If organic waste is to gain widespread acceptance as a fertiliser in agriculture its hygienic standard will have to be assured. In other words, the organic waste will have to be treated and the end-products certified, based either on a treatment of known efficacy or on a microbiological analysis.

Background

The collection and utilisation of organic waste are central components of modern waste management policy. Unfortunately, a variety of pathogens can be found in organic wastes such as manure, slaughter house offal, sewage sludge and wastewater as well as household and restaurant swill (Löfgren *et al.*, 1978). In connection with the use of biological wastes as fertilisers on arable land, new diseases can be introduced to an area or a country. Large-scale changes in waste treatment strategies may also result in the creation of new routes of disease transmission between animals, humans and the environment. As a consequence, risks of transmission may increase for certain diseases which could have serious societal impacts.

In many developed nations, such as the Nordic countries, the classical waste-related and water-borne diseases have largely been controlled by investing in infrastructure for handling sewage and distributing drinking water. Consequently, the sanitary standard in the Nordic countries is high. Furthermore, the disease situation in the Nordic livestock industry is generally very favourable, largely owing to the geographical location of Scandinavia and to strict regulations on imports of animals and animal products. In addition, comprehensive action programmes for controlling certain diseases of domestic animals, such as for example tuberculosis and brucellosis, have been implemented in Sweden. Since Sweden entered the European Union, it has been necessary to relax import regulations to some extent. The disease situation is very dynamic, with new diseases appearing and old ones turning up now and again. In Sweden about 60 new agents of animal disease were discovered within a 20-year period (Wierup, 1989). In view of widespread adoption of waste recycling strategies, we expect that interest will continue to increase in evaluating the associated risks of pathogen spread and taking preventive measures. Thus, effective systems for disease surveillance and control as well as for maintaining a high level of biosecurity will be needed.

Pathogenic micro-organisms

Pathogens are micro-organisms that may cause disease in humans, animals or plants. Plant pathogens will not be further mentioned here, even though they can also be present in organic waste (Kroon, 1997). The pathogens that can cause problems in connection with the recirculation of organic waste include viruses, bacteria, fungi, helminths and protozoa. The species and their numbers are unlimited. In addition, prions, which include the probable agents of "mad cow disease", need to be considered (OIE, 1998). The increasing trend towards globalisation, e.g. increased international trade in domestic animals, animal products, food and feedstuffs, has elevated the risks for introducing new pathogens in organic wastes.

The risk of infection associated with the use of waste materials is determined by a rather complex set of factors. Here the focus is on how the degree of risk for pathogen spread is affected by the initial concentration of pathogens in the organic waste and the degree to which their numbers are reduced during handling and storing. If pathogens are spread out in the environment, it is also essential to be able to predict how long they can survive and whether they will be able to multiply. In addition, for each pathogen, information on potential hosts and the infectious dose for humans and/or animals needs to be obtained.

Post-spreading survival time varies considerably between species of pathogens. Some bacteria, such as *Bacillus* and *Clostridia* species, may produce spores that can remain viable in the environment for decades (Mitscherlich & Marth, 1984). In many cases spores can tolerate pasteurisation as well as other treatments that generally reduce numbers of most other micro-organisms (Mitscherlich & Marth, 1984). Eggs of some parasitic worms, *Ascaris*, in particular, are very persistent in the environment and have been reported remaining viable in soil for up to 7 years (Feachem *et al.*, 1983). Several viruses are also very persistent, and in certain extreme environments they can survive longer than indicator bacteria (Scarpino, 1982).

Under favourable circumstances some bacteria and fungi are able to multiply in the environment. For example, enterohaemorrhagic *Escherichia coli* (EHEC) in manure were shown capable of multiplying from an initial 10^3/g manure to $10^{5\text{-}6}$/g manure within a few days, the actual rate varying depending on temperature (Wang *et al.*, 1996).

The number of micro-organisms needed to cause illness, i.e. the infectious dose (ID), varies considerably between pathogen species and also depends on the infection route (e.g oral, inhalation, through a wound, etc.) (Crozier & Woodward, 1961). For some species of salmonella the ID is very low, i.e. fewer than 100 bacteria (D'Aoust & Pivnick, 1976), and low IDs have also been reported for EHEC (Boyce *et al.*, 1995). For many helminths and protozoa, including *Taenia* and *Cryptosporidiae,* a single egg may suffice to initiate an infection (Feachem *et al.*, 1983). Some bacteria may persist in the body of an infected human/animal for many years before clinical disease (e.g. tuberculosis or paratuberculosis in the case of *Mycobacteria*) becomes evident. ID is also a function of susceptibility which vary between individuals depending on their age, nutritional status, medical treatment, other diseases, etc. The proportion of the human population considered to be immunosuppressed has increased over recent decades (Gerba *et al.*, 1996).

Viable but non-culturable bacteria

In most assessments of hygienic quality, 'kill-off' in the case of bacteria is defined as the level at which bacteria can no longer be detected, most often, using conventional culturing techniques on agar plates. It has been suggested that many bacterial species can enter a viable, but non-culturable, state (VBNC) and thus survive periods of nutritional or thermal stress (Oliver, 1993; Foster & Spector, 1995). Few attempts have been made to evaluate the importance of VBNC bacteria as pathogens in organic waste. Nevertheless, bacteria entering a VBNC state should be given serious consideration since animals can be infected after oral administration of VBNC bacteria (Saha *et al.*, 1991; Stern *et al.*, 1994). Kearney *et al.*, (1994) reported that numbers of pathogenic bacteria grown in a laboratory-scale digester rapidly decreased with 4-5 log of culturable bacteria (*E.coli, S.typhimurium, Y.enterocolitica* and *L. monocytogenes*), but after the addition of nutrients these bacteria will revert to culturable forms. A number of techniques, plagued by various problems, have been used for detecting bacterial cells in a VBNC-state (reviewed by Oliver, 1993).

Zoonoses

Many of the most troublesome pathogens are zoonotic agents; i.e. they can be transmitted between animals and humans. Among these, salmonella, campylobacter and EHEC are of particular importance concerning organic waste use in agriculture. Zoonotic helminths and protozoa of potential importance include *Ascaris*, *Taenia/Cysticerca* and *Giardia.* The virus species of concern with regard to biowaste are generally not zoonotic agents.

Salmonella is a well known problem all over the world and one of the most common zoonotic agents (Acha & Szyfres, 1989). Bacteria in this genus can infect most animal species, including humans. The prevalence of salmonella is very low in food as well as within the animal production chain in most Nordic countries (Hopp, 1996). In most cases where Swedish citizens have contracted the disease, they were infected abroad (Zoonos report, 1997). The low prevalence of this pathogen in Sweden needs to be maintained. It has long been known that salmonella is almost always present in wastewater from urban areas (Danielsson, 1977). Thus, the increased use of sewage sludge as fertiliser may raise the risk of salmonella dissemination, especially to farm animals, if the sludge is not properly treated to kill-off pathogenic bacteria. Subsequent transmission to humans is thought to occur mainly through contaminated food items (Acha & Szyfres, 1989). The possibility of salmonella resistant to antibiotics (multi-resistance) spreading in the Nordic countries has raised special concern.

One zoonotic infection that recently (1995) emerged in Sweden is EHEC. This infection is caused by an intestinal bacteria with certain toxin-producing entities. In humans it can produce severe infections characterised by gastrointestinal disturbances and kidney malfunction. The infection may be fatal in children and the elderly. Ingestion of contaminated food is the main route of infection in humans, but persons can also become infected through contact with contaminated soil or bathing water. The EHEC-bacteria is of special interest from an environmental point of view since it can tolerate changes in the environment much better than other *E. coli* serotypes (Arnold & Kaspar, 1995). Animals, especially cattle, can act as asymptomatic carriers (Beutin *et al.*, 1996). In Sweden slightly more than 1% of the cattle are considered to be infected with EHEC (Albihn *et al.*, 1997).

Epizootic diseases

Epizootic diseases can seriously affect animal welfare and cause severe production losses. Furthermore, the country harbouring the infection may be subject to restrictions concerning the export of animals and animal products, and thereby adversely affect the national economy. A disease outbreak has to be dealt with immediately by implementing an eradication programme and passing special legislation. Often, all livestock on infected farms – and even possibly infected farms – has to be destroyed, and the infected areas are placed under quarantine. All possible means are employed in an attempt to stop the spread of the disease, and in some respects the situation resembles emergencies caused by floods, wildfires and earthquakes. In EU-countries the costs are covered in part by the EU-community and also by the agricultural board in the affected country. From a veterinary point of view, the rise in international trade and travel has increased the risk of introducing exotic pathogens in connection with the application of organic waste to arable land. Newcastle disease virus, a respiratory disease affecting birds, and classical swine fever (CSF) are two diseases of special concern in this respect.

CSF, one of the most serious epizootics affecting swine, can be very costly for farmers and society (Van Oirschot & Terpstra, 1989). The latest outbreak in the Netherlands, involving 429 swine herds, has cost in excess of 20 billion Swedish crowns (Engvall, 1998). In Sweden, increasing numbers of domestic swine are being kept outdoors owing to concerns about their welfare. This increases the risk for contracting an infection from the environment. The CSF-virus is very resistant and may remain viable in smoked or deep-frozen meat (Van Oirschot & Terpstra, 1989). Consequently, organic waste such as slaughterhouse offal as well as household and restaurant swill and sewage sludge may be contaminated with CSF virus.

BSE (bovine spongiform encephalopathy) or "mad cow disease" is probably the most frightening example of what can happen when insufficiently treated organic waste is recycled. Carcasses from scrapie-infected sheep as well as from the first wave of infected cattle processed at to low temperature were used as a component of cattle feedstuff. Since the incubation period is several years the disease had spread widely before the source of the infection was found. The ingestion of meat from infected cattle was most probably the cause of infection in humans. As of October 1998, 29 confirmed cases had been reported (OIE, 1998). Incineration at high temperature is necessary to totally destroy the prions, the most likely agents of BSE (OIE,1998).

Spread of pathogens from organic wastes

Diseases can be transmitted from organic waste to animals or humans in two main ways: First, directly from the treatment system when servicing or repairing the system or when handling or transporting waste. In such situations it is aerosolised micro-organisms and toxins that pose the most serious threat to the health of workers and residents in the surroundings of a treatment plant or site. Second, pathogens may be transmitted after the organic waste, or residues thereof, has been spread on farmland as a fertiliser. From a veterinary point of view this is a more serious problem. Vectors, such as insects, birds or mammals, may transmit pathogens from organic waste to humans or other animal species. Wild animals may also act as reservoirs and transmit the infection back to domestic animals and humans. Infection may also result directly when domestic or wild animals are grazing on or passing over the land. Pathogens can also be transmitted to humans or animals on dust particles transported by wind, in surface water, and on or in food or feed harvested from the treated land.

It can be very difficult to trace an outbreak of a disease in domestic animals or humans back to biological wastes spread on farmland. However, it is often difficult to determine the source of an infection that led to a disease outbreak, even in cases where biological waste is not involved. Most often, it is not possible to confirm the source of infection. It is especially hard to track the pathogen back to the source of the infection in cases where vectors are responsible for a disease outbreak or the pathogen is persistent in the environment. One approach to finding routes of transmission of an infection is to establish the "identity" of the disease agent. Serotyping is a routine method used for identifying microbial isolates. However, it is often not possible to carry out a relevant epidemiological analysis based on serotyping alone. With DNA-fingerprinting the pathogen can be further characterised, thereby helping to identify epidemiologically relevant clones (Altekruse *et al.*, 1997). For the last several years workers at SVA have been using molecular techniques, i.e. pulsed-field gel electrophoresis, to trace sources of salmonella dissemination within Swedish agriculture. In connection with this work, a database consisting of DNA-patterns for different serotypes of salmonella was established and is currently used to trace clones within animal populations and the feed industry, etc. (Aspan, 1998). Similar databases for EHEC and other pathogens isolated from animals are presently being compiled. These databases should facilitate work in determining routes of transmission of these agents within society, i.e between animal, humans and the environment

Vector animals

Vectors have the potential to spread pathogenic micro-organisms from organic waste to surrounding populations of humans, domestic and wild animal species. Fractions of organic waste, such as sewage and food scraps, often attract vectors. They may then transmit pathogens from the waste material directly to a new host by biting or sucking blood, or indirectly by contaminating food, water reservoirs or the environment. In addition, vectors may be consumed by predators or humans.

Insects

Insects of different species may be attracted by the smell of organic waste. Most species come there to feed. A few species are able to take advantage of the "new" opportunities for breeding provided by the recycling of organic waste, but those which do so often appear in very large numbers. The most important disease vectors can be found among the species which breed in garbage and then come into contact with man or domestic animals by feeding on their food. Flies in the families *Muscidae* (including the housefly) and *Calliphoridae* (blowflies) are important in this respect. Also important here are fly species that cause a nuisance by invading homes and may land on food (e.g. *Psychodidae* and *Chironomidae*). Although several other species may be attracted to organic waste, e.g. rove beetles, dung beetles, they normally have little contact with humans or animals (Feachem *et al.*, 1983). Vectors may carry pathogens on their bodies, i.e. mechanical transport, or in their gut, in which case the pathogens can be spread in their faeces. Practically every enteric pathogen has, at one time or another, been isolated in a viable state from insects (Feachem *et al.*, 1983).

Insects may migrate actively or be passively transported by the wind or vehicles of various kinds. In one study where insects of many different species were labelled on a landfill site, some were recovered in traps located in an urban area 500 m from the dump. Some species, as *Psycodidae* were trapped in a slaughterhouse area 11 km away as well as in a village located 11-13 km away (Sveum & Sendstad, 1982). These distances are well within the documented migratory ranges of the species concerned.

Birds

Gulls (*Laridae*) and crows (*Corvidae)* frequently are attracted to organic waste. Like insects, birds have been known to carry pathogens in their gut or, for shorter periods, on their outer body surface (Feachem *et al.*,1983). *Salmonella* was isolated from gulls more often in areas with low hygienic standards, such as in some urban regions, than in more remote areas with higher standards (Aalvik & Rossebø, 1969). In gulls collected from coastal areas in Norway, the incidence of *Salmonella* was about 1% (Aalvik & Rossebø, 1969), whereas it was about 22% in gulls collected from landfill sites (Bö, 1980).

In a Norwegian study 567 gulls marked at a landfill site were later found as far as 1340 km away. In addition, more than 40 % of the re-sitings of the marked gulls were made more than 50 km *away* from the dump (Barikmo, 1980). Therefore, gulls should not be perceived as only a local problem. For birds, in general, the long-distance transmission of pathogens is most likely to occur when the young leave their families and during the winter season when many birds migrate to other regions or countries. Daily migrations over shorter distances can also contribute substantially to pathogen spread. Gulls feed during the daytime and thereafter migrate to neighbouring areas. Their daily need for fresh water can draw them to water supplies in the area surrounding arable land where organic waste has been used as fertiliser. Such water may then become contaminated by bird droppings. In Great Britain, gulls are suspected of contributing to the increased prevalence of *Cysticercus bovis* in cattle. This is the cyst stage of the human tapeworm, *Taenia saginata*. This observed increase in the incidence of infection coincided with an increase in the frequency with which sewage sludge was being spread on farmland. Gulls are attracted to this sludge and also frequent grassland areas. Viable eggs of *T. saginata* has been found in faeces from gulls, as not themselves get infected with this worm (Feachem *et al.*, 1983).

Mammals

Mammals, especially some species of small rodents, can also act as vectors. However, it is more common for wild animals to act as a reservoir for contagious diseases; i.e. the animals are infected with the pathogen but generally show no symptoms of disease. This is a well recognised problem in various parts of the world. For example, in France and northern Germany, wild boars have been found infected with CSF. The infection could have been transmitted to boars consuming infected household waste or coming into contact with sewage, etc. Such infections may be transmitted back to domestic swine, causing disease outbreaks. In Sweden, which has been free from CSF since 1944, it has been pointed out that the increasing and expanding population of wild boars poses a potential hazard of this kind.

Reducing the risk for vector transmission of diseases

Organic waste should be hygienically treated, matured and stabilised before vectors have the opportunity to come into contact with the material. This type of treatment will not only reduce the risk for disease transmission by vectors coming into contact with the material, but will also decrease the attractiveness of the material to them. During the storage, treatment and processing of organic waste, efforts should be made to fence out larger vector animals.

Hygienic treatment of organic waste

To minimise the risk for transmission of pathogens from organic waste to animals and humans the waste should be hygienically treated before being spread on farmland. For some serious or exotic pathogens a complete kill-off is desirable, whereas for many others a marked reduction is sufficient. Although it is technically possible to decontaminate the waste completely; i.e. to sterilise it, before using it on farmland, this is an unrealistic alternative for both practical and economic reasons. Exposure to heat is an effective way to inactivate pathogens. A heat treatment may be performed as a pasteurisation, or heat produced in connection with composting or anaerobic digestion can be used. Many of recommended treatment methods found in the literature involve exposure to temperatures in the interval 50-70°C for durations varying from one hour to several weeks. The higher the temperature, the shorter is the time that it must be maintained. In Denmark recommendations are based on the FS (faecal streptococci)-method: One hour at 70°C is generally recommended for various kinds of waste, but 4 hours of exposure to temperatures as low as 50°C is recommended under some conditions and in combination with anaerobic digestion (Bendixen & Ammendrup, 1992; Espensen, 1996). Extremely strict regulations will tend to discourage the reuse of organic waste on arable land, so a balance needs to be maintained between health risks and treatment costs. In drawing up guidelines the risk of exposure, dose and type of pathogen, etc., and the eventual spread of the pathogen from other sources must all be considered.

In addition to heat treatment a number of other physical techniques, such as irradiation and freezing/thawing, can be used to kill pathogens in organic waste. Chemical treatments usually involve a strong change in pH induced by the addition of alkaline compounds, such as lime, or acids, e.g. peracetic acid. Anaerobic digestion and aerobic composting are biological techniques that are becoming increasingly popular. Long-term storage is another frequently applied biological method. The various treatment methods available differ substantially in terms of the effectiveness with which they reduce pathogen numbers. A review of hygienisation techniques was presented by Inger *et al.*, (1997). In addition to well known factors as temperature, treatment duration, pH and oxygen availability, factors such as microbial antagonism and antibiotic production, exo-enzyme production, and toxic breakdown products also affect treatment efficacy.

Regardless of the kind of suppressive treatment used, there is always the risk for recontamination of the biological waste with pathogens. For example, in our laboratory salmonella has been detected in pasteurised (70°C for one hour) and digested residue from a commercial biogas plant. Under certain circumstances there may also be a risk for multiplication of these pathogens. It is therefore essential that sterilised waste be handled, stored and transported with great care.

To be able to evaluate the hygienic status of the end-product from a treatment process it is important to continually monitor the process with regard to treatment duration, temperature, pH, moisture content, etc. Often it is easier to specify a process rather than the end-product. Clear distinctions between process standards and product standards need to be drawn.

End-product analysis

Another way to verify the pathogen-suppressing effect of a treatment is to evaluate the end-product microbiologically. One could, for example, check directly for the presence of certain pathogens, or a more indirect analysis, based on indicator organisms, can be used. It is difficult to quantify pathogens in organic waste. Thus some workers have suggested a round-about strategy in which well-defined pathogens are added to the process and repeatedly monitored. In this way it is possible to find "weak points" that can be targeted in order to more effectively suppress the pathogens concerned. One example of this approach is described by Norin *et al.* (1996) who added "tea bags" of permeable polyamide containing Ascaris-eggs to a pilot-scale aerobic thermophilic reactor containing sludge. *Salmonella typhimurium* bacteriophage 28B, used as a viral indicator, was added directly to the sludge in the reactor. In the same paper, parallel laboratory-scale studies of treatments designed to suppress *Mycobacterium paratuberculosis* and *Salmonella spp* were described. Except for the bacteriophage, which required more than 24 hours of exposure to 60°C, the studied pathogens were reduced satisfactorily by 12-24 hours of treatment at 55°C. It should be kept in mind that in direct analyses, pathogen numbers may be underestimated in cases where the pathogen is hidden by organic material and/or fat or oil.

Analysis based on indicator organisms is the most frequent approach in cases where a hygienic standard must be met. Hygienic standards are often based on the results of tests with indicator organisms, i.e. organisms whose presence is often correlated with the possible presence of specific pathogens in a given material. Among the organisms used for this purpose are certain species of bacteria (e.g. faecal streptococci and coliformes), bacterial spores (e.g. clostidia spores) virus (e.g. parvovirus), bacteriophages (e.g. *Salmonella Typhimurium* 28B) and parasites (e.g. *Ascaris suum* eggs). An indicator organism should be easy to detect and common in the material or substrate of interest. It should not be able to vigorously multiply in the material, but should have the capability to stay alive for an extended period. The use of indicator organisms can get complicated since they seldom meet every one of these criteria in a given case. Another problem is that several pathogenic micro-organisms may survive longer than indicator organisms. Faecal streptococci (FS) appear to be useful for monitoring disinfection processes under certain conditions (Bendixen & Ammendrup, 1992; Espensen, 1996). *Enterobacteriacae*, including coliforms and *E.coli*, generally have a lower resistance to heat and extreme pH thus their presence can only be interpreted as indicating gross under-processing. Still, they may be of help when evaluating recontamination after hygienisation (Strauch & De Bertoldi, 1991). More research will have to be carried out in this area in order to determine which indicator organisms are most suitable for hygienic evaluation of organic waste.

As an extra safety precaution the agricultural uses of an end-product may be restricted. Restrictions are most likely to be placed on end-products of low hygienic standard, and as a consequence there is very little demand for them. However, it should be clearly understood that in every case where pathogens are spread in nature there is always the risk that they will be transported from a less sensitive crop/area to a more sensitive one. Also, as earlier mentioned, some pathogens are capable of multiplying in the environment. If the use of organic waste as fertiliser in agriculture should become generally accepted, the health risks must be controlled.

Legislation concerning hygienic treatment

In Sweden, there is no biosecurity-related legislation regulating the use of organic waste as fertilisers. By contrast, such legislation has been passed in several other countries, including Norway and Denmark. Sewage sludge in Sweden is usually spread without being hygienically treated. One exception concerns low-risk animal waste: Regulations established by the Swedish agricultural board (SJVFS 98:34, K14) specify that such material must be exposed to 70°C for one hour prior to use in a biogas plant or composting process.

Furthermore, all high-risk animal waste and low-risk material destined for uses other than those mentioned above has to be exposed to 133°C for 20 min under 3 bar pressure. This heat treatment is in accordance with EU legislation. The definitions of low- and high-risk material are rather complex but, in general, all material of animal origin is included in one of these categories with the exception of manure and ingesta from the stomach and gut. However, bacterial spores, some heat-resistant viruses and prions can persist in residues even after exposure to 70°C for one hour. Porcine parvovirus is an example of a heat-resistant virus that needs 1.5 h of 70°C for a $4\log_{10}$-reduction (Bendixen, 1995). Several species of spore-forming bacteria are pathogens that may cause problems if the current disease situation changes. Spores can survive pasteurisation and for kill-of the material needs to be sterilised (according to the legislation for Animal high risk material).

Hygienic aspects of different fractions of organic waste

Domestic animal manure

The traditional solid manure treatment system includes composting and thus some heat treatment of the product. If to small quantities of straw are used, related to the amount of manure produced, no heat will be developed (Plym Forshell, 1996). However, with the slurry systems popular today, which have a low dry matter content, natural heat treatment is not possible.

Traditionally, and to a certain extent even today, manure is mainly used locally, thereby minimising the risk of spreading diseases over extended distances. However, manures are often treated on a large-scale basis and may be transported long distances as well as over international borders. Transport is necessary owing to the concentration of animal production and crop production to within certain areas, which leads to a local overproduction of manure at animal production facilities and a demand in crop production areas. Such manure can be used simultaneously by many farmers over a large area. Also, animals and animal products as well as feedstuffs are transported over longer distances today compared with the traditional situation. As a result, the risks of introducing exotic infections and spreading domestic infections in connection with manure use are increasing.

Digested residues from biogas plants

In Sweden, there are presently eight large-scale biogas plants designed for digesting manure together with other fractions of organic waste. Several more plants are being planned. Biogas plants cannot be operated in a profitable way unless the digestion residues can be used as fertiliser to be spread on farmland. Such fertiliser can be especially valuable for ecological farmers and constitutes an important link in the recycling of organic waste between urban and rural areas. Most Swedish plants treat all waste according to the legislation for low-risk animal material; i.e. it is subjected to a separate hygienisation step (pasteurisation at 70°C for one hour). This treatment reduces numbers of most pathogens to acceptable levels. However, standard procedures for inspecting and certifying biogas plants have yet to be developed.

In Denmark, where large-scale biogas treatment plants have been in operation longer than in Sweden, extensive studies have been made concerning safeguards against pathogens in these plants (Bendixen & Ammendrup, 1992). Hence, the Danes frequently use the FS-method, a microbiological method based on the quantitative determination of faecal streptococci in biomass before and after treatment (Bendixen & Ammendrup, 1992; Espensen, 1996). However, as for all indicator organisms some limitations concerning use and evaluation of results exist. The FS-method alone is not considered to be sufficient for monitoring continuous processes or processes where temperatures exceed 60°C (Espensen, 1996).

Sewage sludge

From a veterinary sanitation point of view, sewage sludge is a complex material, the use of which poses a number of animal and human health hazards. It is well known that sludge regularly contains pathogens such as salmonella (Bendixen & Ammendrup, 1992; Danielsson, 1977; Feachem *et al.*, 1983). The presence of pathogens in sludge reflects the incidence of infectious diseases within the human population in the area served by the plants. Since micro-organisms adhere to organic material, pathogen concentrations in sewage sludge are about the same as those in raw wastewater. A large and increasing part of the sludge produced in Swedish sewage treatment plants (40% in 1997, Swedish Water and Wastewater Association, Stockholm) is spread on arable land. Regulations concerning the application of sludge to farmland have been developed (Swedish Environmental Protection Agency, 1995). However, no regulations concerning the hygienic standard of this sludge exist in Sweden. The conventional treatment of sludge, such as anaerobic digestion at 35°C or long-term storage, reduces the content of pathogenic micro-organisms to a certain content, but these treatments are nowhere near as effective as hygienic treatments. Also, some sludge is spread fresh without treating it at all. In the future, several new methods for hygienically treating sludge based, for example, on dehydration, composting, drying and pelleting, should become generally available for use.

Composted material

The composting process or aerobic digestion can be managed in several different ways. For example, the material can be treated in liquid or solid form, in a container or in a pile, and in large or small amounts. If the composting process is satisfactorily managed, the resulting alkaline pH and high temperature should inactivate most of the pathogens (Kron, 1997; Strauch & De Bertoldi, 1991). Thus it is important to periodically measure pH and temperature during the process period in order to be able to evaluate the hygienic status of the compost. However, such measurements are hard to make in compost piles since conditions differ between the central and outer parts. Hence, it is essential that the compost pile be optimally managed, e.g. turned over several times.

Conclusions

The increasing use of organic waste as fertiliser on arable land is a very positive development, reflecting a growing awareness of the need to recycle nutrients and thereby enhance the sustainability of food production while reducing society's waste burden. However, the potential health risks associated with this strategy cannot be ignored, and no reduction in the biosecurity of humans or animals should be tolerated, nor should any lowering of sanitary standards be allowed. The rapid pace of development in the field of recycling of organic waste has to be followed by an increasing understanding and respect for the risk of disease transmission. To deal effectively with this potential threat, we need additional scientific documentation regarding the hygienic quality of fertilisers based on organic waste. With such knowledge it should be possible to develop simple and effective systems for safely recycling organic waste. In other words, the organic waste will have to be treated and the end-products certified, based either on a treatment of known efficacy or on microbiological analysis.

Acknowledgement

The author wish to thank Dr. Anna Aspán and prof. Anders Engvall for critical review of the manuscript.

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J. Petersen & S.O. Petersen (eds.) 1999. Use of municipal organic waste
Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
Session 2
Danish Institute of Agricultural Sciences report no. 13

The function of soils after sewage sludge amendments: Soil microbial aspects

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Introduction

The maintenance of soil quality and fertility are important aspects of a sustainable farming system. An increased recirculation of organic residues may give a number of environmental benefits, such as decreased need for land filling and recirculation of nutrients. However, there is also a risk of an increased load of heavy metals and xenobiotics. If we are to increase the circulation of organic residues, such as sewage sludge, between the urban and rural areas, we must ensure that the quality and fertility of our soils are not negatively affected in a long-term perspective. Soil microorganisms are an important factor when determining the impact of anthropogenic activities on soil quality. Microorganisms live in intimate contact with their environment and respond quickly to changes. In addition, soil microorganisms are responsible for a great number of important processes involving the mineralisation and immobilisation of plant nutrients. Together with physical and chemical parameters this makes them suitable as indicators of changes in soil quality and fertility.

The main objective of the present study was to evaluate long term effects on soil microorganisms and soil chemistry after 16 years of sewage sludge amendments at moderate application rates. The soil microbial parameters to be studied were chosen to reflect general microbial processes (carbon and nitrogen mineralisation), and more specific processes (nitrogen fixation, ammonium oxidation and denitrification). Effects on soil microbial biomass were assayed by the substrate-induced respiration method. In addition, changes in activity of acid and alkaline phosphatases were tested. All these methods were adopted from the Swedish EPA programs "Soil test system – MATS" and "Integrated soil analysis – ISA".

Materials and methods

Experimental design

The field experiments were situated close to Lund (Igelösa) and Malmö (Petersborg), in the southern part of Sweden (56°N, 13°E). The sewage sludges used in the experiment were collected from the municipalities of Lund and Malmö for the sites of Igelösa and Petersborg, respectively. The addition of sludge was equivalent to 0, 1 and 3 ton ha^{-1} $year^{-1}$ (dry matter). However, the additions were performed every 4th year (1981, 1985, 1989 and 1993), resulting in additions of 0, 4 and 12 ton ha^{-1}. Both treatments with and without sludge were complemented with mineral fertiliser. The addition was 0, half and full rate of nitrogen in relation to recommended rates in the area. At half and full nitrogen rates, the recommended rates of phosphorous and potassium were also applied. Four replicates of all combinations of sludge and nitrogen were used, giving a total number of 36 parcels (with dimensions 6 x 20 metres) at each experimental site. In order to facilitate management of the blocks, the experimental design was not completely randomised.

Microbiological analyses

The basal respiration and substrate induced respiration (SIR) was determined in 30 g soil adjusted to 60 % of the water holding capacity and incubated in 20°C. The evolution of CO_2 was determined with a respirometer (Respicond III, Nordgren Innovations AB, Djäkneboda 99, S-918 93 Bygdeå, Sweden) using the method described by Nordgren et al. (1988; 1992). SIR was measured after 10 days, where each soil sample received a glucose mixture consisting of 600 mg glucose, 60 mg $(NH_4)_2SO_4$ and 1200 mg talcum evenly mixed into the soil. After substrate addition the initial disturbances caused by change in temperature, gas exchange and soil mixing was allowed to fade out (3-4 hours) and the average of three hours of measurements was used to calculate SIR. Basal respiration was calculated as the average of the last 40 h of measurement before SIR.

Heterotrophic nitrogen fixation was measured after amendement of glucose. After three days acetylene was added and the ethylene production rate was used as a measure of the nitrogen fixing capacity (Mårtensson and Witter, 1990).

The nitrogen fixing capacity of cyanobacteria was determined on four occasions over a three month long incubation time under constant illumination (Mårtensson and Witter, 1990).

The nitrogen mineralisation capacity was analysed with the anaerobic incubation method described by Waring and Bremner (1964) and modified by Stenberg et al (1998). Ammonium was extracted after seven days of incubation at +37°C.

The potential ammonium oxidation rate was assayed as accumulated nitrite according to the short incubation, chlorate inhibition technique described by Belser and Mays (1980) and modified by Torstensson et al. (1992).

The potential denitrification activity was assayed according to the short incubation, C_2H_2 inhibition method described by Pell et al (1996). The denitrification rates were calculated with a product formula that also takes growth of the bacteria into consideration (Stenström et al. 1991).

Phosphatase activity was analysed by the p-nitrophenyl phosphate method described by Tabatabai and Bremner (1969) and Sjökvist (1993). The rate by which p-nitrophenol is formed is related to the amount of active enzyme. Both acid and alkaline phosphatase was determined.

Statistical analysis

Data matrices consisting of the two different soils and treatments as objects and the different chemical and microbiological parameters as variables, were reduced by principal component analysis (PCA) to a few underlying components expressing the major variation in the data set. All variables were centred and scaled to equal variance by dividing them by their own standard deviation. Separate models were made for each soil and sampling occasion. Standard and stepwise discriminant function analyses (DFA), based on Mahalanobis distance, were performed in order to determine which variables discriminated between the *a priori* defined groups of sludge and nitrogen treatments.

Results and discussion

It was evident that the two experimental sites were different regarding both chemical and biological parameters. Igelösa had a higher content of carbon and nitrogen and generally also a higher biological activity. Generally, sludge application increased the organic carbon content. Although not significant, the trend was for increasing total nitrogen concentrations with increasing rates of sludge application in Igelösa, while no such effects were found in Petersborg.

The tests concerning nitrogen fixation did not perform well. This may be due to a high spatial variability, suppression of activity due to high amounts of available nitrogen, or to unsuitable methods. Variation within treatments were high and often the activity was zero. These tests were thus not included in the evaluation.

Despite differences in their original chemical and biological properties, the two experimental soils reacted in a similar way to the sludge amendment, as revealed by both PCA and DFA. The first principal component generally described organic matter-related influences and the second component mainly described pH-related effects. The DFA showed acid and alkaline phosphatases, potential ammonium oxidation, and total nitrogen to be the most influential parameters when trying to discriminate between *a priori* defined groups of sludge treatments.

The experimental sites described, together with the large amount of data collected, represents a unique opportunity to reveal effects on soil quality when applying sludge at moderate application rates. The study showed that amendment of sewage sludge affected several investigated parameters, chemical as well as biological. This effect was mostly positive, probably due to the increase in available organic material. This increase in activity is temporary. The true effect of added metals and xenobiotics is not to be seen until most of the extra carbon has been respired.

Acknowledgements

The financial support of SYSAV Utveckling AB is gratefully acknowledged.

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J. Petersen & S.O. Petersen (eds.) 1999. Use of municipal organic waste
Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
Session 2
Danish Institute of Agricultural Sciences report no. 13

Effects of organic wastes on microbiological aspects of soil quality

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Introduction

Organic waste products represent a source of energy and nutrients for soil microorganisms upon field application, but in recent years the presence of organic micro-pollutants in organic wastes has lead to concern about the potential for adverse effects on crop and soil quality (Helweg *et al.*, 1996). It is likely that one of the first aspects of soil quality—by Doran and Parkin (1994) defined as "the capacity of a soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health "—to be affected by the composition of organic wastes will be the functions of primary decomposers, i.e., the soil microflora. Soil microorganisms catalyze nutrient transformations in arable soils and thus constitute an important link between organic pools and the growing crop; changes in the functional capacity of a soil could interfere with the turnover and plant availability of nutrients. This paper describes previous and on-going work to study interactions between organic waste products and soil microorganisms.

Field application of sewage sludge

In the fall of 1995 an intense debate was initiated in Denmark over the risks associated with sewage sludge application to arable soil. This triggered a number of research activities, including a field study with application of sewage sludge and solid cattle manure (Krogh *et al.*, 1996; 1997). One experiment that will be emphasized here was carried out on a coarse sandy soil at Lundgaard, and with application of sludge from Herning municipal sewage treatment facility at rates of 0, 3.5, 7 and 21 ton DM sludge ha^{-1}. Table 1 shows concentrations in the sludge of four organic pollutants, i.e., the four parameters for which intervention (cut-off) values have been established in Denmark, along with the intervention values currently employed.

Table 1. Concentrations of four selected pollutants in the sludge used for the experiment, and currently used intervention values in Denmark.

Compound	Herning sludge	Intervention value[1)]
	mg kg^{-1} DM	
LAS	1900	2600 (1300)
Σ PAH	11	6 (3)
Nonylphenol (+ethoxylates)	100	50 (10)
DEHP	23	100 (50)

[1)] Shown in parentheses are intervention values as of 1 July 2000.

The experimental lay-out was a randomized block design with six replications, from which soil was sampled and pooled for microbiological analyses. Microbiological assays aimed at quantifying functional groups of organisms, as well as the overall size and composition of the microbial community. Soil for these analyses were collected 3, 21 and 145 days after sludge application. Data analysis indicated that sampling errors had occurred after 21 days, and results from this sampling are therefore not presented.

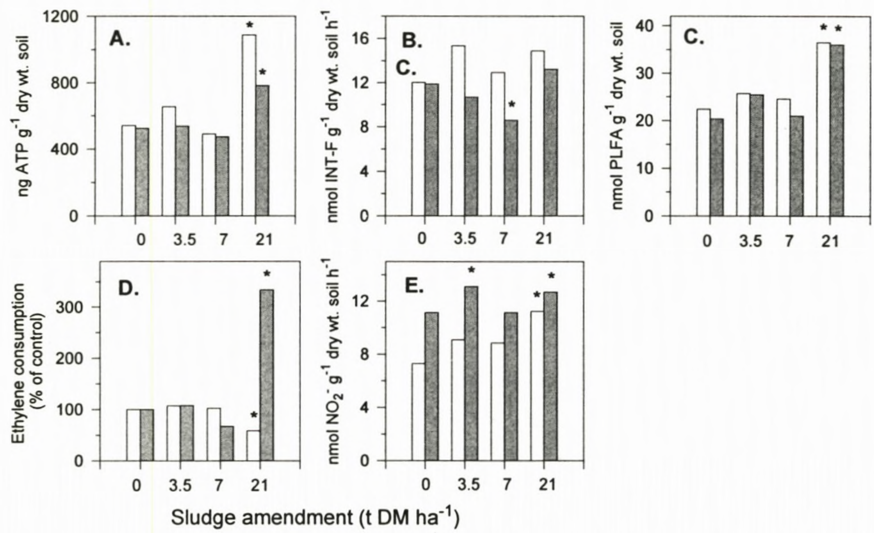

Figure 1. Concentrations of ATP (A), potential dehydrogenase activity (B), PLFA concentrations (C), ethylene degradation (D) and potential NH_4^+ oxidation in soil amended with 0-21 t DM sludge ha^{-1} 3 days (open bars) and 145 days (grey bars) after application. Sludge amendments were compared with the control using Dunnett's test; asterisks denote significance at $P<0.05$.

Between the first sampling in mid November 1995 and the last sampling in early April 1996 there were limited changes in most microbiological parameters. On both occasions broad parameters, like ATP, potential dehydrogenase activity and phospholipid fatty acid (PLFA) concentration, were generally increased at the highest sludge application rate (Fig. 1A-C). In fact the concentration of PLFA at the highest application rate (35-37 nmol g^{-1} dry wt. soil) corresponded closely to the concentration predicted from separate analyses of sludge and soil (37.6 nmol g^{-1} dry wt. soil), which suggests that the turnover of sludge microorganims in the soil may have been limited during the winter period covered in this study. This conclusion is corroborated by the PLFA composition of soil, sludge, and soil-sludge mixtures. Results from a principal component analysis of the mol percentage distribution are shown in Figure 2. The PLFA composition of sludge was very unique compared to soil samples, but the samples from the highest sludge application rate were distinctly closer to the pure sludge along the first principal component (PC1), explaining 77% of the total variation, than samples from the other amendment levels.

More specific assays, like ethylene degradation and potential NH_4^+ oxidation, initially showed varying response; ethylene degradation was inhibited at the highest sludge application rate, but NH_4^+ oxidation stimulated (Fig. 1D-E). After 145 days ethylene degradation was greatly stimulated at 21 t DM sludge ha^{-1}, indicating some microbial succession even though this was not apparent from the analysis of PLFA composition. A stimulation of NH_4^+ oxidation after 145 days had occurred in all treatments and was thus not closely associated with the sludge application.

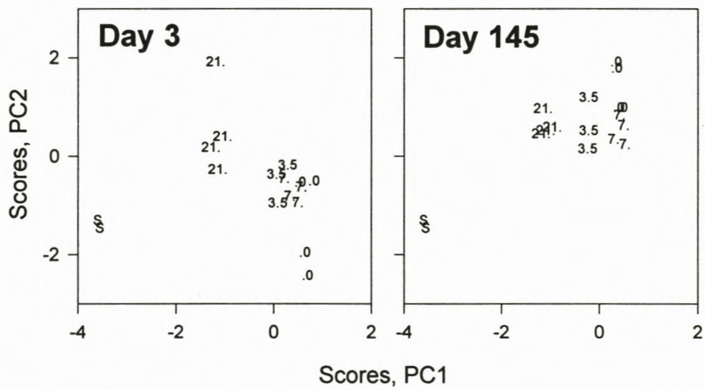

Figure 2. The mol percentage distribution of 35 long-chain phospholipid fatty acids in pure sludge (S) and in soil samples collected after 3 or 145 days from the different sludge amendment levels (0, 3.5, 7 and 21 t DM sludge ha^{-1}) were analyzed by a principal component analysis.

Direct effects of LAS

The Danish Institute of Agricultural Sciences (DIAS) is involved in an exotoxicological assessment of linear alkylbenzene sulfonate (LAS) effects on soil microorganisms. Dose-response curves have been established with soil from Lundgaard for a wide variety of parameters including ethylene degradation, Fe(III) reduction, potential dehydrogenase activity, β-glucosidase activity, NH_4^+ oxidation activity, cellulolytic organisms, and PLFA composition (Elsgaard *et al.*, in prep.). Results from this on-going study have not been released for presentation yet, but broad indices of microbial biomass and activity appear to respond differently to LAS amendment, i.e. be less sensitive, than assays targeting specific functional or taxonomic groups.

Activity and growth of microorganims in soil are generally limited by the availability of suitable energy and nutrient supplies. Starvation may influence their ability to adapt to environmental stresses (Petersen and Klug, 1994), and the toxicity of a given compound may therefore depend on the availability of energy. It follows that, for example, LAS entering the soil in a matrix of sludge may affect the soil microbial biomass differently than LAS alone. Also, the physical adsorption of LAS to sludge particles may contribute to such diverse effects. Current experimental work is being performed with amendment to soil of LAS or LAS+sludge, and preliminary results confirm that the presence of sludge dampens ecotoxicological effects of LAS.

The soil-plant-waste environment

In the field study described above, the highest sludge application rate corresponded to an addition of 18 mg LAS kg^{-1} soil (cf. Table 1) if completely dispersed in 0-15 cm depth. This value is in itself within a range that could give measurable reductions in several microbial parameters, at least in the absence of sludge. In practice the distribution of organic waste materials will be very heterogeneous, and so there will be soil volumes with much higher concentrations of waste and, accordingly, much higher concentrations of any pollutants in the waste. Microorganisms in and around such organic waste 'hot-spots', i.e. those organisms who are primarily involved in transformations of nutrients from the waste, may thus be exposed to a different concentration range of organic micro-pollutants than predicted from average values.

Sewage sludge is normally stored under anaerobic conditions prior to land application. Organic matter has a considerably higher water retention capacity than mineral soils, and soil moisture will therefore be elevated in soil volumes enriched with organic waste (Petersen and Andersen, 1996). Organic matter also stimulates biological acitivity, leading to an increased O_2 demand compared to the bulk soil. The combination of elevated moisture and O_2 demand may stabilize the anaerobic environment of sludge particles and result in a much higher persistence of compounds that are less degradable under anaerobic conditions—which includes DEHP, nonylphenol, LAS and PAH—than predicted if it is assumed that the soil environment is fully aerated at the time of application.

The kinetics of biodegradation may be crucial with respect to plant uptake of organic micro-pollutants, since the growth of plant roots can become stimulated in the nutrient rich environment around sludge particles (van Vuuren *et al.*, 1996). Therefore, delays in biodegradation of organic micro-pollutants due to anaerobic conditions will increase the potential for direct contact between roots and pollutants in the sludge.

Organic wastes and soil quality

The net effect of organic wastes on soil quality will consist of beneficial effects on, e.g., bulk density (Stone and Ekwue, 1996), C and N mineralization potential (Dar, 1997) and microbial activities (Hattori, 1988), and adverse effects due to factors such as high ionic strength and toxic levels of pollutants (Knight *et al.*, 1997; Krogh *et al.*, 1996). Not all parts of the soil microbial community may respond similarly to the input, as suggested by the studies of LAS sensitivity described above, and inhibition of one subset of the community may be compensated for by increased activity of another.

A study of organic waste effects on selected soil physical, chemical and microbiological properties was recently initiated within the Danish Centre for Sustainable Land Use and Management of Contaminants, Carbon and Nitrogen. Participants are DIAS and Research Centre Risø. One aim of this project is to quantify and contrast short-term (weeks) and long-term responses (years) of arable soils to organic waste amendment, and to climatic and soil type related variability. A second objective is to characterize organic waste products in terms of organic pools and, through isotopic labelling, trace the turnover of C and N in waste amended soil. The goal is to improve our understanding of the interactions between organic wastes and soil in order to allow some prediction of the effect of a given product on soil quality. As suggested in the introduction we believe that biological parameters will give an early response to positive and/or negative effects of organic amendments and thus may serve as early indicators of changes in soil quality.

Acknowledgements

The work presented in this paper was supported by grants from the Danish Environmental Research Programme 'Centre for Sustainable Land Use and Management of Contaminants, carbon and nitrogen', and from ERASM and a consortium of companies (CLER, Ecosol, Condea Augusta, Petresa, Wibarco, Procter & Gamble, Unilever, Henkel).

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J. Petersen & S.O. Petersen (eds.) 1999. Use of municipal organic waste
Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
Session 2
Danish Institute of Agricultural Sciences report no. 13

Prediction of mineral N dynamics following additions of organic waste to agricultural soil.

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Introduction

The mineral N dynamics during decomposition of organic matter in soil depends on the composition of the organic matter. A stable and N rich material may give a slow net N mineralisation, while a material consisting of easily decomposable organic C may stimulate a temporary net assimilation of N. The composition and pre treatment of organic waste may determine the mineral N dynamics connected to the decomposition of the matter. In order to minimise the risk of N leaching and maximise the agronomic value of organic waste applied to agricultural land, the mineral N dynamic should be evaluated in order to advice adjustments to the agricultural practise.

In order to predict mineral N dynamic following additions of organic waste to agricultural land, mathematical models of soil C and N dynamics may be a useful tool. In the present study the SOILN_NO model (Vold *et al.*, 1998) is applied to data from a laboratory incubation experiment where different kinds of organic material have been added to a soil. By appraising the estimated deviation of the C and N content of the different pools against a "common sense" opinion of the distribution and examining the prediction errors, the adequacy of the SOILN_NO model for predictive purposes is assessed.

Material and methods

The incubation experiment

A mineralisation experiment was carried out in the laboratory in order to measure transformations of soil NH_4^+ to NO_3^- and the production of CO_2 gas. The soil was sampled from an experimental field at the Agricultural University of Norway, Ås. The parent material is post-glacial clay. The soil was a loam, classified as a typic Haplaquept with 21% clay, 40% silt and 39% sand (Børresen, 1993). The soil water content during the incubation was adjusted to 55% of the water holding capacity of the soil, corresponding to a water content on 25.4%. One set of the soil beakers were kept as unamended controls (O). To the rest, different kinds of organic material were added. The organic wastes used were raw sludge (OA1), anaerobically stabilized sludge (OA2), primary source sorted organic waste (OA3), slightly decayed organic waste (OA4) and rotator treated sewage sludge compost (OA5). Composted wastes have been shown to be highly resistant to microbial degradation in soils (Castellanos and Pratt, 1981, Senesi, 1989, Bernal *et al.K*, 1994). Usually it is relatively easy to force a model to reproduce a stable data set by adjustment of parameters. In order to test the function of the model, organic materials which were expected to give significant fluctuations in soil N had to be included in the experiment The following organic materials were therefore included: bone meal (KJ), cow feed pellets (F97), and dried leguminous plant (TBV). The soil samples were amended with OA2 in an amount corresponding to 0.30 mg N/g dry soil, whereas the other organic materials were added in an amount corresponding to 0.15 mg N/g dry soil (Table 1). The beakers with soil/organic matter amendments were incubated at 15°C for 8 weeks in a dark chamber. Every week beakers were removed and stored frozen until the measurements of KCl-extractable NH_4^+ and NO_3^- by FIA (flow injection analysis). The CO_2 production was measured every third day in the beginning of the experiment and later on every week by a wide bore capillary gas chromatographic system described in detail by Sitaula *et al.* (1992). The number of replicates in incubation experiment 1 varied between two and four, while all the incubations in experiment 2 were duplicated.

Table 1. Total C, total N and C:N ratio in the soil and the different types organic matter used in the incubation experiment.

Material	Total C content [g C (kg dry soil)$^{-1}$]	Total N content [g N (kg dry soil)$^{-1}$]	C:N
Soil (Control)	35.81	3.58	10.00
Raw sludge (**OA1**)	1.56	0.12	13.00
Fully decayed sludge (**OA2**)	3.12	0.27	11.56
Primary source sorted organic waste (**OA3**)	2.07	0.13	15.92
Slightly decayed organic waste (**OA4**)	1.75	0.12	14.58
Rotator treated sewage sludge compost (**OA5**)	3.70	0.12	30.83
Bone meal (**KJ**)	0.57	0.13	4.38
Cow feed pellets (**F97**)	2.12	0.13	16.31
Dried leguminous plant (**TBV**)	3.04	0.13	23.38

The model

SOILN_NO (Vold., 1997) is a mechanistic model simulating soil C and N dynamics. It was developed to predict quantitatively the levels of N leaching from agricultural soils under different agronomic practices. The model is described in detail by Vold (1997).

Parameter and initial values setting

In the present study two different approaches were used for matching the observed C and N dynamics.

Parameter set 1

The parameters were set to the values obtained from optimization to data from a laboratory incubation experiment (Vold *et al.*, 1998). It is desirable to keep changes in the basic parameters to a minimum, but the microbial growth yield efficiency (f_e) and the half saturation constant for respiration in the denitrification expression (c_r) values were allowed to fluctuate within their area of definition (Vold *et al.*, 1998) (Table 2).

Since SOILN_NO is at a stage of parameterization, generating a universal parameter set simulating C and N dynamics satisfactorily may be more important than just attaining the parameters giving the optimal fit for a specific site or experiment:

Parameter set 2

An alternative parameter set which has shown to be applicable to some agricultural field experiments (Vold, 1997, Haugen *et al.*, 1998), were used as a basis for the simulation.

In addition, the initial amount of C and N in three different organic pools, litter 1, litter 2 and humus, were estimated in order to obtain the best fit between observed and simulated C and N dynamics. When optimizing to the control, the sum of C and N in biomass, litter 1, litter 2 and humus should not exceed the total amount measured in the soil. The basis content of C and N dispersed into the different organic pools by optimization to the control, were kept as a steady basis in the subsequent adjustments to include the application of organic waste. In the optimization of initial values for the different treatments, the lower limit of the actual value was set to the value obtained for the control, while the upper limit was set to the total C and N content in the organic materials added. The distribution of C and N proposed by the model may be interpreted as a characterisation of the organic material added. In order to evaluate the adequacy of the model, the proposed distribution may be appraised against a "common sense" opinion of how easily decomposable the different organic matters were.

Results and discussion

The incubation experiment

All the treatments had their highest respiration during the first week of incubation, with decreasing respiration thereafter.

The decomposition of sludge (raw sludge (OA1), fully decomposed sludge (OA2) and rotator treated sewage sludge compost (OA5)) showed a rapid disappearance of NH_4^+ (Fig. 1). The decay of OA2 and OA5 was characterised by an initial steep increase in the soil NO_3^- content, substituted by a slighter increase after about two weeks of incubation. For OA1 an initial disappearance of NO_3^- was observed. Microbial assimilation of mineral N during decomposition is mainly expected when the organic matter is poor in N. In the present study OA5 was poor in N (C:N=31, Table1), but that was not the case with OA1 (C:N=13, Table 1). However, the microbial assimilation seen for OA1 may be caused by specific constituents of the sludge. For example, in a study with decomposition of cattle and pig slurry in soil Kirchmann and Lundvall (1993) found microbial N immobilisation to be caused by the presence of volatile fatty acids in the decomposed organic matter. The fatty acids acted as an easily decomposable C source for micro organisms and caused immobilisation of N. Furthermore, hemicellulose and lignin concentrations of sludge have been reported to have negative effects on N mineralisation rates (Hattori and Mukai, 1986). No significant N_2O emission was observed during decomposition of the three different types of sludge (Bakken and Sogn, in prep.) The present period of net N assimilation in the OA1-treatment was short.

As illustrated by the relatively rapid decrease in respiration (Fig. 1), the period of net N assimilation was presumably connected to burn-out of the energy source.

Among the two types of solid organic wastes, primary source sorted (OA3) and slightly decayed (OA4), addition of the less composted material (OA3) gave the highest biological activity (respiration) (Fig. 1). Similar to the decomposition of OA1, described above, decomposition of the primary source sorted organic waste (OA3) (C:N=16, Table 1) was also characterised by an initial net disappearance of NO_3^-, and assimilation of NO_3^-. The N dynamics during the decomposition of OA4 were characterised by a constant and slight increase.

In a study of C and N mineralisation of fresh, aerobic and anaerobic animal manure during incubation in soil, Kirchmann (1991) found the microbial activity resulting from addition of non-decomposed organic materials to soil to be much higher than that obtained by adding composted materials. The results from the present study support those findings. Among the organic wastes addition of OA3 gave the highest biological activity. In addition, application of the KJ, F97 and TBV resulted in a considerable respiration (Fig. 1). The organic material included in order to secure significant fluctuations in the N dynamics, also gave an active N turnover during decomposition. In particular bone meal (KJ) released NH_4^+ during decomposition. The C:N ratio of KJ was low (C:N=4, Table 1), and microorganisms may release mineral N during degradation of such N rich materials. The decomposition of KJ gave a steeply increasing soil NO_3^- concentration, levelling off at a significantly higher N level than the other materials investigated. The decomposition of both F97 (C:N=16) and TBV (C:N=23) were characterised by net assimilation of mineral N. Whereas the net disappearance of N only lasted for about 1 week for F97, it continued for about 3 weeks for TBV.

Application of SOILN_NO

In the model application the NH_4^+ and NO_3^- values were summarised in order to represent the soil mineral N content.

Use of Parameter set 1

Calibration to control:

The SOILN_NO model was calibrated using data from the control soil, i.e. the soil incubated without addition of organic matter (Fig. 2). The model was relatively easily constrained to reproduce the observed data (Fig. 2). Except for the two parameters f_e (the microbial growth yield efficiency) and c_r (the half saturation constant for the respiration in the denitrification function), the model parameters were set to the figures estimated by Vold *et al.* (1998). When optimized to match the C and N dynamics during incubation of the control soil, the microbial growth yield efficiency (f_e) was forced to the maximum within its area of definition. The half saturation constant for the respiration in the denitrification function (c_r) was squeezed as low as possible.

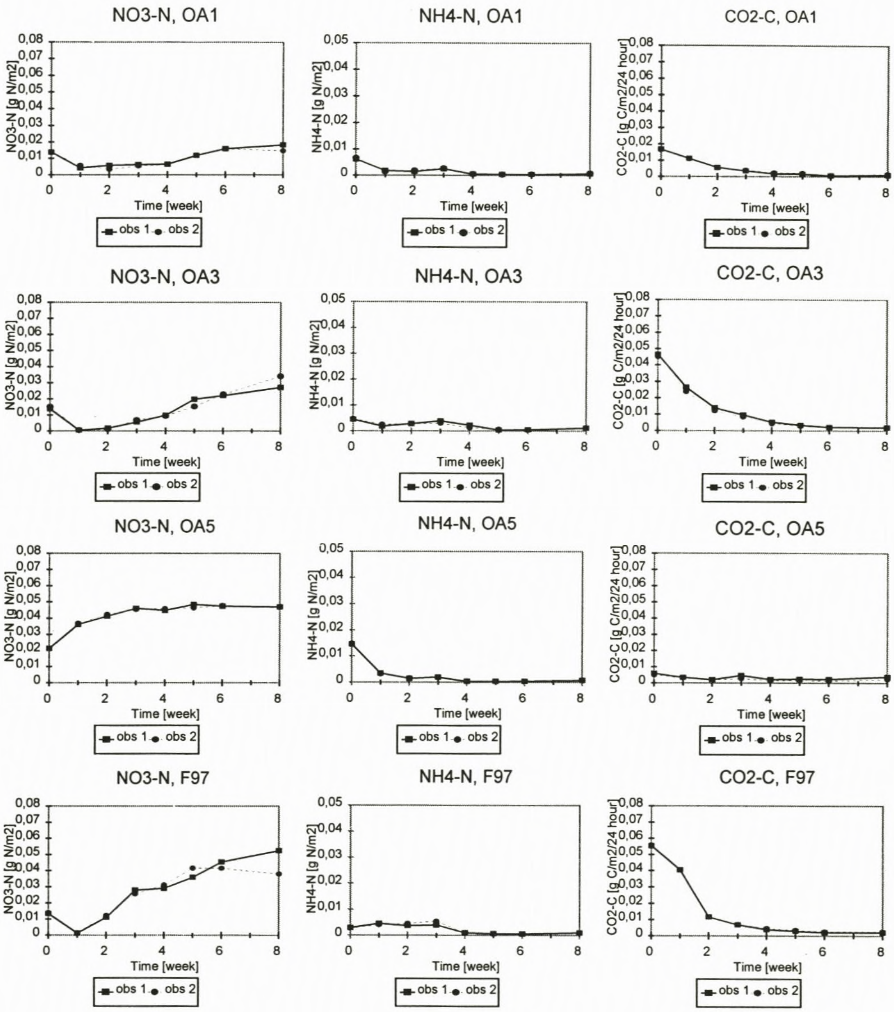

Figure 1. Observed changes in soil NO_3^-, NH_4^+ and CO_2 (g) during decomposition of some of the organic matters used.

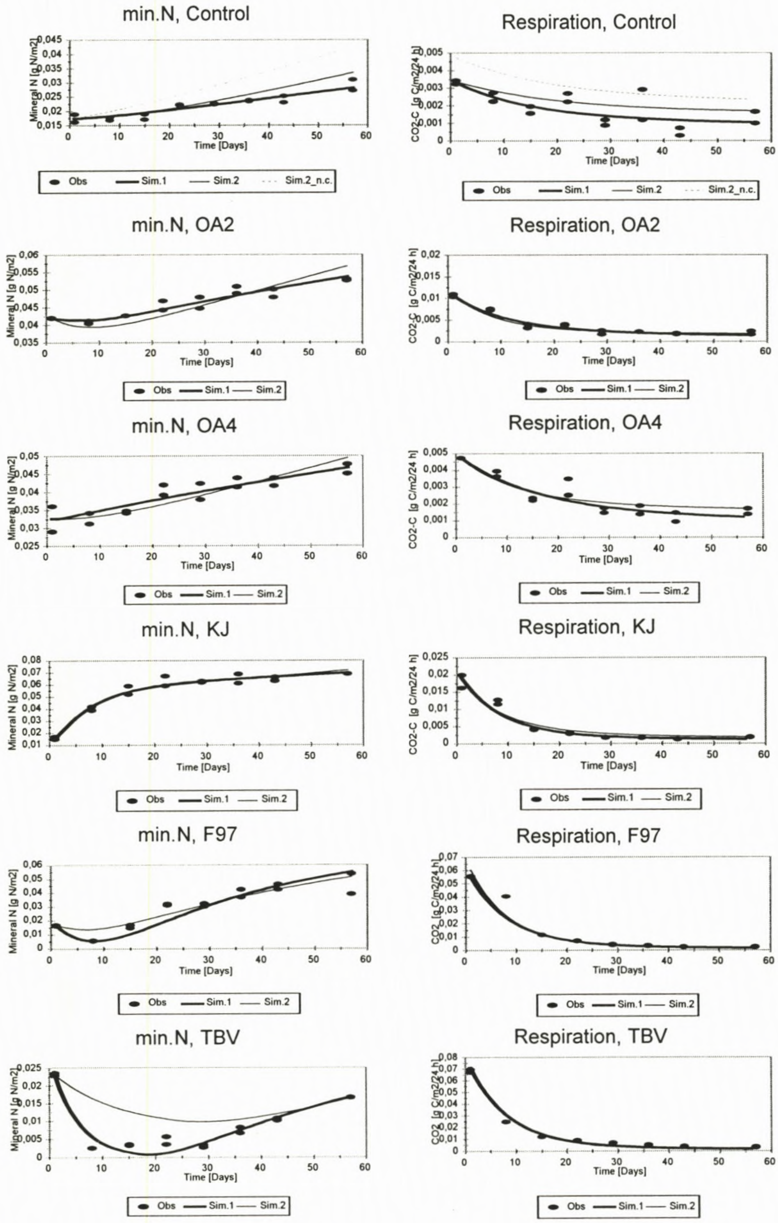

Figure 2. Simulated and observed mineral N ($NO_3^- + NH_4^+$) and CO_2 (g) dynamics during the incubation experiment for some of the organic matters used.

By optimization approximately 42% of the soil total C was considered by the model , and dispersed among the litter 1 (0.02%), litter 2 (0.2%) and humus. The majority were laid in the humus compartment. The rest 58%, not considered by SOILN_NO, must be interpreted as having a more refractory character than allowed for in the model. Concerning N, more than 99% of the soil total N content was put in the humus fraction, 0.2% in litter 2 and only 0.0002% in the litter 1 pool.. The distribution gave C:N ratios of litter 1 which are 100-fold higher than in litter 2. An easily decomposable organic material which mainly consists of C may not be unrealistic.

Simulation of organic waste decomposition:

The N transformations and CO_2 emission following addition of fully decayed sludge (OA2), easily decayed organic waste (OA4) and bone meal (KJ) were indeed satisfactory simulated by the SOILN_NO model (Fig.2). When the N dynamics were characterised by sudden shifts between a temporary delay in the net disappearance of NH_4^+ followed by NO_3^- accumulation, resulting from the addition of raw sludge (OA1), the primary source sorted organic waste (OA3), cow feed pellets (F97) and dried leguminous plant (TBV) the simulations were somewhat delayed relative to the observations (Fig.2). However, also the general patterns following these additions were well reproduced by the SOILN_NO model.

Table 2. Distribution of the C and N content of the different organic matters added to three organic pools within the SOILN_NO model. The figures given in the table were added to the amount of C and N put in the different compartments during the calibration of SOILN_NO to the control soil. All the figures are given as % for the total C and N content in the organic matter applied. The deficit to 100% for C and N each, represented the fraction proposed to be more stable than those allowed for in the SOILN_NO model. The fraction of C and N disregarded must be derived from the organic materials added; differences in availability of soil C and N should not be expected.

Org.mat.	% of total C considered	% of total N considered	%C in litter1	%C in litter 2	%C in humus	%N in litter 1	%N in litter 2	%N in humus
OA1	19	100	13	6	-	-	10	90
OA2	100	7	2	4	94	-	7	-
OA3	44	100	27	17	-	36	64	-
OA4	100	88	-	3	97	-	8	80
OA5	100	13	-	-	100	13	-	-
KJ	50	53	43	7	-	53	-	-
F97	45	92	37	8	-	-	92	-
TBV	100	51	35	-	-	9	42	-

The amounts of the applied C and N dispersed to the different organic pools were reasonably well estimated by the model when appraised against "common sense" (Table 2). The organic matters expected to be relatively easily decomposed (OA3, KJ, F97, and TBV) had the major part of C in the litter 1 pool according to the model (Table 2). For OA3 and KJ the major part of the N content was referred to the litter 1 compartment. The N content of F97 and TBV was mainly represented by the more slowly decomposing litter 2 fraction. As expected, OA4 was suggested by the model to be more stable than OA3. For OA4 only a minor part of C and N were put in the litter 2 pool, whereas the major part was transmitted to the humus compartment. Except for some minor fractions of labile C and N, the model suggested the different kinds of sludge used (OA1, OA2 and OA5) to be very stable.

Use of Parameter set 2

Calibration to control:

By using the parameter set 2 derived from previous field studies, no combinations of initial values making the model match the observed C and N dynamics were found (Fig. 2, control soil). However, by increasing the microbial growth yield efficiency from 0.37 to 0.5 (the same level as optimized for parameter set 1), the model approached the observed data (Fig. 2). The deviations of the soil total C and N content in the different organic compartments were higher than with parameter set 1. While about 100% of the soil C content was put in the humus compartment, almost all the N was kept in a more refractory fraction than allowed for in the model.

Simulation of organic waste decomposition:

Although not as good as parameter set 1, the parameter set 2 seemed to be applicable for simulating the C and N dynamics following the addition of the different kinds of organic waste (Fig. 2). The subsequent optimization of initial values to match the observed data again proposed a reasonable distribution among the different organic pools. Also in this attempt the model proposed OA3, KJ, F97 and TBV to be the most labile materials, with their major C and N content located in the litter A pool.

General assessment of model adequacy and parameterisation

The overall impression from the present application of the SOILN_NO model, was that the model is adequate for simulating the C and N dynamics following decay of quite different organic materials. By using parameter set 1, adjustments of only two parameters within their areas of definition resulted in low relative prediction errors. Also, the model suggestions of C and N dispersion between the three distinct litter pools were reasonable characterisations of the different organic materials added. Thus we feel that the relative success of the model in accounting for the treatment differences in soil N levels, respiration, and turnover rates lends support to the model formulations of the effect of quality of organic matter inputs on soil organic matter dynamics.

Parameter set 1, where two parameters were allowed to be optimized in order to match the observed data, resulted in significantly lower prediction errors than parameter set 2. However by adopting the value of the parameter controlling the microbial growth yield efficiency (f_e) from the optimization of parameter set 1 in parameter set 2, SOILN_NO made a relatively successful simulation of the observed C and N dynamic. In strict sense of model validation, the fact that it was necessary to alter some parameters is evidence that the model does not provide a fully generalized explanation of soil organic matter dynamics. In other words, there appears to be significant site or substrate dependent factors that are not currently explained by the model. However, the necessity of adjusting f_e might be connected to the nature of the experimental approach to simulate natural mineralisation by an incubation experiment in the laboratory, rather than to the nature of the true decomposition. Thus the parameter set 2, without the adjustment made here is not rejected from further use on field experiments.

In a forth-coming paper an approach to validate the SOILN_NO model will be carried out. The parameters and initial C and N distribution obtained in this study will be tested against data from a model lysimeter experiment where soil column with barley have been added OA4 and F97 during a 3 years period.

Conclusion

The general impression from the present application of SOILN_NO, is that the model seems to be adequate for simulating the C and N dynamics following decomposition of quite different organic materials. Also the model characterisation of how labile or refractory the organic matter was, seemed reasonable. Generally, the simulation results agreed reasonably well with measurements from the incubation experiments, which supports the main functional relationships postulated in the model.

Although minor adjustments of parameters had to be carried out in the present application of SOILN_NO, the possibility of finding a generalized parameter set is supported by the present study. Consequently, adequacy of the SOILN_NO model for predictive purposes is expected.

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J. Petersen & S.O. Petersen (eds.) 1999. Use of municipal organic waste
Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
Session 2
Danish Institute of Agricultural Sciences report no. 13

Biodegradation of DEHP in sludge-amended agricultural soil

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Introduction

Among the phthalate esters, DEHP (di(2-ethylhexyl)phtalate) is the most frequently used additive in the manufacture of flexible polyvinyl chloride (PVC). It is also to a lesser ex-tent used in the industrial production of lubricants, glues, insect repellents and dielectric fluids. The annual global production of this plasticizer constitutes approximately 10^6 tons (Nielsen & Larsen, 1996).

DEHP has been classified as a priority pollutant with a relatively low acute toxicity, but with suspected mutagenic and carcinogenic effects (Nielsen & Larsen, 1996; Beliles at al., 1989; Schulz, 1989).

A significant amount of DEHP is released annually to aquatic and terrestrial environments from waste water treatment plants. In Denmark, the annual release has been estimated to 5-8 tons for terrestrial and 13-18 tons for aquatic ecosystems (Hoffman, 1996). Mass balance estimates of DEHP degradation for 3 Danish sewage treatment plants showed large variations in DEHP degradation (24% to 73%) between the individual plants. The content of DEHP in the dewatered sewage sludge varied from 17 to 120 mg/kg dw (Kjølholt *et al.* 1995). Regulatory limit values have been set by the Danish EPA in 1997 for DEHP and 3 other organic contaminants in sewage sludge when used as organic fertiliser on arable land (Table 1). The limit values will be lowered further in the year 2000.

Table 1: Limit values for organic pollutants in Danish sewage sludge used on agricultural land.

Organic pollutant	Limit value (1.7.1997) mg/ kg dw sludge	Limit value (1.7. 2000) mg/ kg dw sludge
LAS	2600	1300
PAH (total)	6	3
NPE	50	10
DEHP	100	50

Experimental set-up

The experiments described in this paper were carried out with dewatered sewage sludge from two different treatment plants A: Aalborg West (17% dry matter) and B: Lundtofte (22% dry matter). Two agricultural top soils were used. A sandy loam soil from Foulum (2.5% organic matter) and a coarse sandy soil from Lundgaard (2.2% organic matter). The soils had never received sewage sludge prior to this study.

Microcosms were set up in the laboratory with sludge mixed homogeneously with sieved top soil (FA): Foulum soil + sludge A and (LB): Lundgaard soil + sludge B (ratio 55:1 dw). In addition, sludge in the form of water saturated aggregates were placed either on top of sterile quarts sand (sludge A) or sandwiched between two soil layers (sludge B + Lundgaard soil) packed to a volume density of 1.4 g/cm^3 dw.

The microbial mineralization of DEHP in the sludge and sludge-amended soil was measured as $^{14}CO_2$ after addition of [^{14}C]DEHP to the sludge fraction. The methodology is described in detail by Roslev *et al.* (1998).

Adsorption and bioavailability

DEHP is a non-polar compound with a high log K_{ow} (7-7.8) and low water solubility (<0.4 mg/l). It will adsorb strongly to the organic fraction in sludge and soil (Staples *et al.*, 1997). A linear adsorption isotherm for DEHP was measured in batch experiments with sludge enriched soil (FA) after 2 and 7 days (Fig.1).

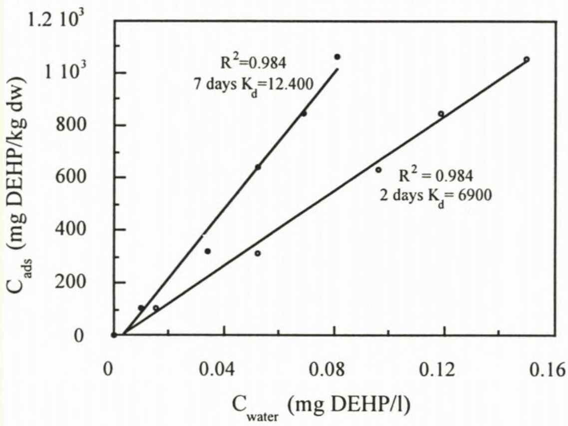

Figure 1. Adsorption isoterms for DEHP in sludge-amended soil after 2 and 7 days equilibration (data from Gadegaard *et al.*, 1997)

The adsorption of DEHP could be described by a linear adsorption isotherm ($C_{Ads} = K_d$ x C_{water}) and the estimated adsorption coefficient K_d increased from 6.9 x 10^3 by day 2 to 12.4 x 10^3 by day 7. This corresponds with adsorption kinetics observed for other non-polar compounds with high log K_{ow} values, and is interpreted as an initial rapid adsorption to outer surfaces of the organic matter followed by a slower diffusion and adsorption into the soil organic matrix.

The high K_d value indicates that only a small part of the total soil DEHP will be present in the soil pore water. At the limit value for DEHP (100 mg/kg dw sludge) only 0.14 μM DEHP will be available for microbial degradation in the soil water of the sludge-amended samples.

The desorption of DEHP from inner surfaces in the organic soil matrix is slow and may with time become the rate limiting step in microbial degradation. It also hampers extraction efficiency and may lead to errors in mass balance measurements of DEHP (Roslev *et al.*, 1998).

The effect of organic matter on the bioavailability of DEHP due to adsorption is illustrated in Fig. 2. Here, mineralization of radiolabelled DEHP was investigated in microcosms containing a pure culture of the DEHP degrading strain SDE-2 isolated from sewage sludge A . The mineralization of DEHP in liquid culture was attenuated significantly by addition of organic matter from either Foulum top soil or sewage sludge, and the attenuation was closely related to amount of organic matter added (Roslev *et al.* 1998).

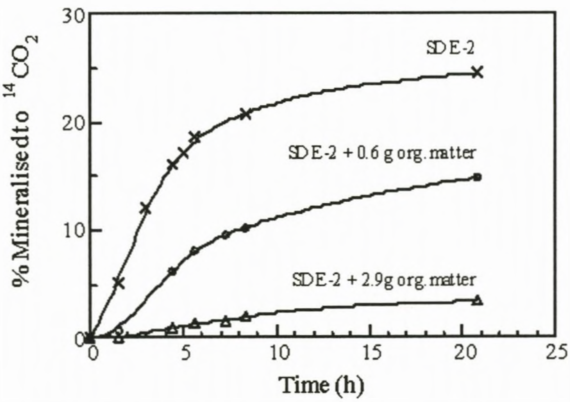

Figure 2. Mineralization of [^{14}C]DEHP by the DEHP-degrading strain SDE-2 in liquid culture (x) and with addition of organic matter from agricultural soil (o) (0.6g/l) and sewage sludge (Δ) (2.9g/l). Figure redrawn from Roslev *et al.*, 1998.

Kinetics of DEHP degradation in soil

Aerobic degradation rates were measured over time in microcosms with sludge-amended soil (FA) containing concentrations of DEHP in the range 62-2090 mg/kg dw sludge. The initial degradation rates showed a linear increase with increasing DEHP concentrations and indicates that the mineralization follows first order kinetics (Fig. 3). The upper end of the DEHP concentration range is well above regulatory limit values for DEHP, and it can be expected that DEHP degradation in the soil in general will be substrate limited with respect to DEHP. However, degradation rates may be influenced by other factors such as the size and specific activity of the DEHP degrading microbial population and restrictions in bioavailability due to slow desorption of "aged" DEHP from the organic matrix.

Long term incubations of mixed sludge-soil microcosms over 130 days revealed that first order degradation kinetics (Phase I) was followed by a slower degradation rate (Phase II) best described by a fractional power model. This shift in apparent kinetics for DEHP mineralization was evident after approximately 3 weeks (20°C, aerobic).

The slower Phase II degradation can be attributed to slower desorption of aged DEHP from inner sites of the soil organic matrix, but may also reflect a decreasing microbial population of DEHP degraders over time.

From the combination of Phase I and II degradation models, effective half-lives of the DEHP substrate could be estimated (Roslev *et al.* 1998, Madsen *et al.*, submitted).

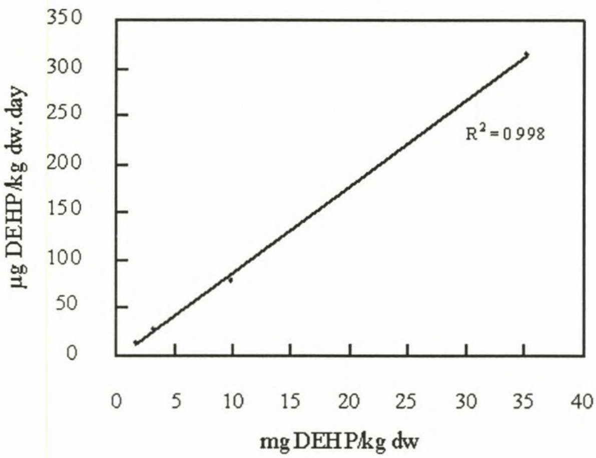

Figure 3. Initial mineralisation rates of [^{14}C]DEHP in sludge amended soil with DEHP concentrations in the range 1.55-35.1 mg DEHP/kg dw soil (equivalent to 0.14-2.8 μM DEHP/ l pore water; Kd = 12.400). Data from Madsen *et al.* (submitted).

DEHP degrading bacteria in sludge and soil

Both the dewatered sludge and the agricultural top soil had a capacity for DEHP degradation under aerobic conditions. The highest capacity was present in the sludge, and aerobic degradation of the sludge obtained by mixing with sterile quartz sand gave a half life ($t_{1/2}$) at 20°C of only 51 days, whereas $t_{1/2}$ for sludge mixed into soil was 151 days.

The lower degradation rates for sludge mixed with soil may be due to a number of factors such as dilution of DEHP in the soil pore water, change in soil physical and chemical conditions, and increased competition for substrates other than DEHP.

Characterisation of the micro-organisms actively metabolising [^{14}C]DEHP by use of radiolabelled biomarkers in the cell wall was performed on sludge and sludge-amended soil. These experiments revealed that DEHP metabolising bacteria from the sludge also seemed to be the most active degraders after mixing into the soil (Roslev *et al.* 1998).

Inoculation of sludge and soil with DEHP degrading bacteria (strain SDE-2 isolated from sludge A, 10^9/g dw) gave a significant (four-fold) stimulation of the degradation rate. This increased degradation potential was still fully present in the sludge after 30 days (Roslev *et al.* 1998). These results indicate that DEHP degrading bacteria present in the sludge survived better in the sludge environment than in the soil.

Incubation of sludge enriched soil with sludge from the two different sewage treatment plants (Aalborg West and Lundtofte) showed marked differences in the DEHP degra-dation potential. After 4 month of incubation, 51% of the ^{14}C label was still left in the soil amended with Aalborg sludge, whereas only 38% remained in soil amended with Lundtofte sludge. This variation in DEHP degrading capacity may relate to differences in sludge treatment at the two sewage plants. One half of the Aalborg West sludge consisted of sludge from the primary settling tank, which had been treated in a mesophilic biogas reactor before dewatering, whereas no anaerobically treated sludge was present in the Lundtofte sludge.

Effect of oxygen availability

Long term incubations of sludge enriched soil (FA) under aerobic and anaerobic conditions showed that DEHP can be degraded without oxygen, but at a much reduced rate. Initial rates decreased by a factor 5.5 and the half-life of the DEHP substrate increased from 151days to >365 days (Madsen *et al.*, submitted)

Table 2. Effect of oxygen availability on DEHP mineralization in sludge amended soil with initial DEHP concentration of 1.6µg/g dw (from Madsen *et al.*, submitted)

Sample	Incubation condition	% [^{14}C] label remaining in soil after 130 days
sludge (A) amended soil	20°C, arobic	51
sludge (A) amended soil	20°C, anaerobic	80

Effect of temperature

Temperature had a significant effect on aerobic DEHP mineralization measured in microcosms with sludge amended soil (FA). Initial rates decreased by a factor of 2.4 with a decrease in temperature from 20°C to 10°C. The strongest effect was seen at temperatures below 10°C, where a temperature decrease of only 5°C gave the same reduction of initial rates (factor 2.4). After 4 months of incubation only 23% of the initial [^{14}C] DEHP had been mineralized at 5°C, whereas almost 50% had been mineralized at 20°C (Table 3).

Table 3. Effect of temperature on DEHP mineralisation in sludge amended soil (FA). Aerobic incubations with an initial DEHP concentration of 1.6 µg /g.dw (from Madsen *et al.* , submitted)

Temperature	Initial DEHP mineralisation rate (ng/g dw·d)	[^{14}C] label remaining in soil after 130 days (%)
20°C	21.3	51
10°C	8.8	61
5°C	3.7	77

Effect of sludge aggregate size

Sludge applied as organic fertiliser in the field is spread and incorporated into the top soil layer in clumps of varying size, and these sludge aggregates will stay partly anaerobic for some time due to a combination of high water content and high oxygen consumption rate. Aerobic incubations of water saturated sludge aggregates placed on sterile quartz sand gave low DEHP degradation rates only slightly higher than DEHP degradation under fully anaerobic conditions (Tables 3 & 4). These results could lead to the assumption that aggregates would severely reduce DEHP degradation in the field situation. However, incubations of water saturated sludge aggregates of equal size in contact with soil (LB, sandwich) gave a quite different picture as illustrated in fig. 4.

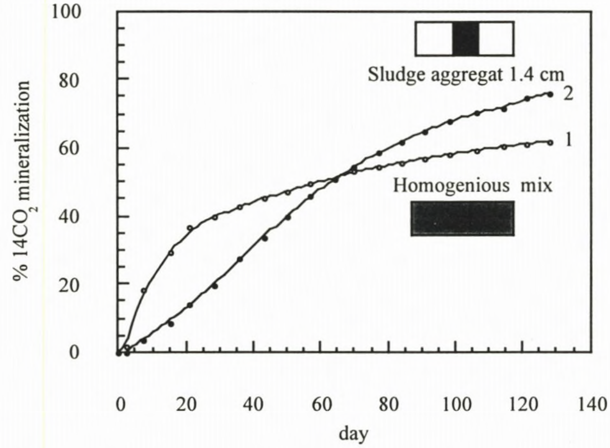

Figure 4. Mineralisation of [^{14}C]DEHP to $^{14}CO_2$ in sludge amended soil. Incubated at 20°C for 128 days. 1: homogeneously mixed 2: sludge aggregate (1.4 cm) sandwiched between soil layers.

Within the first two months, accumulated $^{14}CO_2$ was highest in the homogeneously mixed sludge-soil samples. The lower initial mineralization rate for the aggregates can be attributed to the anaerobic volume present in the aggregates, where DEHP degradation is reduced. After 2 months of incubation, however, highest accumulated $^{14}CO_2$ was measured for the aggregates (1.4 cm), and 86% of total DEHP content were mineralised here after 4 months compared to 61% in the mixed sludge-soil samples. For aggregates of the same size, but incubated on coarse sterile quartz sand, only 27 % of the DEHP was mineralized to $^{14}CO_2$ (Table 4).

This difference in DEHP degradation between initially water saturated sludge aggregates with and without soil contact is caused by convective water transport from the aggregate into the soil due to gradients in soil water potential over the sludge-soil interface.

This effect has been demonstrated for bands of cattle manure injected into soil (Olesen *et al.*, 1997). Air-filled pore space is created, first in the outer parts of the aggregate and with decreasing oxygen consumption over time, oxygen can slowly penetrate deeper into the aggregate through the air-filled pore space.

In the sludge aggregate without soil contact, the pore space will remain water filled, and since diffusion is approximately 10^4 times slower in water compared to air, anaerobic conditions can be maintained in large parts of that aggregate, even at moderate oxygen consumption rates.

The more complete mineralization of the DEHP content in the sandwiched aggregate after 4 months as compared to mixed samples could be interpreted as better conditions for DEHP degrading bacteria in the sludge environment, once oxygen with time penetrates to the inner parts of the aggregate.

Table 4. Effect of sludge aggregate size on [14C]DEHP mineralisation to 14CO2.

Sample	Incubation condition	% [^{14}C] label remaining in soil after 130 days
sludge (A) aggregate 1.5cm;	20°C, aerobic on quartz sand	73
sludge(A) homogenious mix	20°C, aerobic sterile quartz sand	35
sludge (B) aggregate 1.4 cm	20°C, aerobic sandwhich	24
sludge (B) amended soil homogenious mix	20°C, aerobic	38

Conclusions and perspectives

A direct comparison of the laboratory degradation patterns measured for DEHP at a time scale of a few months with the fate of DEHP under field conditions at time spans of several years is of course problematic. However, some general trends with respect to the key parameters regulating DEHP degradation can be outlined.

The laboratory experiments showed, that DEHP adsorbs strongly to the organic soil fraction. Microbial metabolism of the compound is substrate limited and degradation is, therefore, relatively slow with about 50% of the DEHP content mineralized after one year at a mean soil temperature of 10°C. The strong adsorption will reduce leaching to a minimum and a continuos application of sludge fertiliser over the years will, therefore, lead to an increased level of DEHP in the top soil. Long term field experiments combined with measurements of existing field sites with known long term application of sludge is needed to elucidate the resulting level of DEHP.

Aerobic degradation of DEHP proceeds much faster than degradation under anaerobic conditions. Spreading and soil incorporation of sludge creates aggregates of varying size in the top soil and temporary anoxic conditions in the sludge aggregates will reduce DEHP degradation during an initial degradation period. It appears, however, that for smaller aggregates mineralization is only hampered for a shorter period and may become more complete with time due to more favourable conditions for DEHP mineralizing microbes inside the sludge aggregate after oxygen penetration.

The effect of aggregate size will be further clarified in upcoming field experiments. There are strong indications from biomarker experiments, that the DEHP degrading microbial population present in the sludge also are the most active degraders after soil incorporation. A high degradation capacity in the sludge will, therefore, increase degradation rates after soil application. Sludge from sewage treatment plants with different pre-treatment of the sludge showed different degradation potentials. Better knowledge about the factors affecting growth and activity of DEHP degrading bacteria in the sewage treatment plant and the subsequent procedures for sludge treatment can contribute to higher DEHP degrading capacity in the sewage plant and in the field after soil application.

Acknowledgements
We thank Helle Blenstrup, Christina Mogensen and Kirsten Maagaard for excellent technical assistance and Peter L. Madsen for valuable input. This work was supported by the Danish Strategic Environmental Reserach Programme, Centre for Sustainable Land Use.

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Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
Session 2
Danish Institute of Agricultural Sciences report no. 13

New ecological treatment technique for sewage sludge

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Abstract

The amount of sewage sludge processed annually in Finland is around 1.5 million tons (Ministry of the Environment 1998). In the 1980's, nearly half of the sewage sludge was utilised as fertiliser in agriculture. In the 1990's, the agricultural use of sludge has decreased significantly, mainly because of noxious elements in sludge. At present, the most common sludge treatment technique in small and medium-sized sewage treatment plants is composting in piles. One-third of the composted sludge is used as soil amendment in landscaping, but this has not totally compensated for the reduced use of sludge in agriculture. Consequently, one third of the sludge is disposed of to landfills (Ministry of the Environment, 1992).

The increasing volume, limited utilisation options and low demand for composted sludge has led to storage problems in many sewage treatment plants. The establishment of a new, environmentally safe composting area is expensive, likewise the technical methods of sludge treatment (Flink & Leppälä, 1997). These problems have made it necessary to develop new techniques for sludge treatment, aiming at better quality and applicability of the end material and lower treatment costs.

In the Biofield technique the sewage sludge is amended to surface soil where it is decomposed through natural processes. The vegetation planted in the field is assumed to promote the treatment process. In addition, the crops produced in the field can be utilised as a source of energy or fibre. The leaching waters can be recycled to the field or to the sewage treatment plant. The Biofield technique aims at producing a cleaner end product than do conventional sludge treatment techniques, and thus suitable material for various fertilisation and soil amending purposes. It is also a cost-effective technique when compared with sludge composting, because of low material and labour needs.

The ecological advantages of the Biofield technique are mainly related to favourable effects of the vegetation in the sludge treatment process. Vegetation increases the organic matter content in the soil, and thereby improves the physical properties and the biological activity. These mechanisms, together with additional nutrients and organic matter in the sludge, enhance the overall decomposition process. Also, vegetation has a major role in the regulation of water balance in the Biofield. For instance, plants may transpire several hundred kilograms of water for every kilogram of dry plant-biomass during the growing period

(Heinonen *et al.*, 1992). Vegetation is thus assumed to decrease the leaching of waters and therefore reduce environmental nuisances as well as treatment costs. Vegetation is also assumed to decrease nutrient concentrations in leachate waters through two mechanisms. Firstly, vegetation binds nutrients into biomass. For instance, the nitrogen content in the yield can exceed one hundred kilograms per hectare and the phosphorus content up to twenty kilograms per hectare, depending on the plant species (Heinonen *et al.*, 1992). Secondly, vegetation diminishes erosion and hence leaching of nutrients. Perennial vegetation, especially, is effective in preventing nutrient losses (e.g. Heinonen *et al.*, 1992). In addition, the use of certain plants has proved to be a promising method for cleaning metal-contaminated soils through bioremediation (e.g. Raskin *et al.*, 1997), which is also the basic idea behind the Biofield technique.

The efficiency of the Biofield technique was tested in a lysimeter experiment in the summer of 1998. Germination and growth of two plant species (*Brassica nigra, Brassica rapa var. oleifera*) in strongly sludge-fertilised soil and in immature sludge compost, as well as the metal binding ability of plants and the quality of leachate waters, were studied. The results showed that the tested plant species produced a normal yield (compared to the average yield in southern Finland), despite the relatively high doses of sludge (180 m^3/ha) in the growth substratum. Phosphorus and nitrogen concentrations in the leachate waters were not significantly higher in sludge-treated lysimeters than in untreated reference lysimeters (phosphorus: p=0.958; nitrogen: p=0.786), which indicates that the vegetation efficiently bound the additional nutrients of the sludge.

In conclusion, the literature study and also the preliminary results of the field experiments indicate that the Biofield technique offers a promising alternative method for sludge composting. However, more research is needed, concerning for example the highest possible dose of sludge, the wintertime function of the Biofield, and the quality of the end material.

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J. Petersen & S.O. Petersen (eds.) 1999. Use of municipal organic waste
Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
Session 3 Poster abstract
Danish Institute of Agricultural Sciences report no. 13

Compost from large scale units in Sweden
– survey of producers and consumers

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The survey consists of two different questionnaires that have been sent to both companies/ administrative organizations that produce compost on a large scale, and to potential consumers of compost on the market. Since the emphasis of the survey was on compost quality, a more detailed study into quality assessment and quality declaration has been carried out with the help of telephone interviews.

Results producer survey

In total we recieved questionnaires from 48 producers which can be divided into the following categories:

– Municipal waste handling administrations	24
– Regional waste handling companies	16
– Waste handling company within municipal administrations	6
– Municipal park administrations	3
– Cemetary administrations	3
– Farm	1
– Unclear administration	3

The main stream of raw material composted in the plants was park and garden waste. In 37 plants this kind of material dominated. Of the total amount of about 200 000 tons of material which is treated by the plants in the survey, 45 % is park and garden waste. In 6 plants the main resource is household waste. 4 plants mentioned sludge from waste water treatment as the main resource.

The questions to the producers dealt with volume, technique, rutines for sampling, sale and marketing. 26 plants stated that the the main part of the compost produced is used within their own administration or company. In 8 cases all compost was used as cover material on their own waste depony. 14 plants stated that compost was mostly produced for commercial purposes.

All plants were asked to declare on a scale from 1 to 7 how easy or difficult it is to get rid of the compost produced. More than 30 % declared that "it is very easy", 20 % declared "it is very difficult".

The main obstacles defined, which make it hard for compost to compete against other soil improvers on the market, are: the product has no or little documentation concerning how it works in plant cultivation (9 answers); fear of spreading weed seeds (7 answers); compost generally has a bad reputation (6 answers); and bad marketing strategies (6 answers).

The results from the producer study indicated that the consumer´s experience of the function of compost is very limited, and that compost has an undeservedly bad reputation. In order to deal with this, the consumer should be able to evaluate the product under realistic conditions, which would be possible by carrying out more demonstration trials and scientific trials.

Results consumers survey

In total we recieved questionnaires from 63 potential consumers of compost which can be divided into the following categories:

– Park administrations	41
– Cemetary administrations	5
– Housing companies	5
– Landscape architecture companies	4
– Park management company within municipal administrations	3
– Garden construction companies	2
– Park and housing management within county councils	2
– Agricultural school	1

The consumers of compost clearly express a wish for appropriate norms to assess quality and the spreading of compost, but there is also a demand for process development. In order to increase the possibilities for use and market shares, users ask for improved guarantees from the sellers, reliable deliveries, a sustained level of quality, better guidelines from the seller, and also education for the users.

45 of the consumers declare that they have experiences with the use of soil improvement material based on compost. The effects of continuous use of compost most frequently mentioned in the consumer survey are: increased plant growth (32 answers), improved soil structure (36 answers), simplified soil treatment (36 answers), and better plant establishment (26 answers). Both increased and decreased amounts of weeds have been observed. It is difficult to define whether occurence of plant diseases in connection with use of compost has increased or decreased, since only a few mentioned such effects.

Within a AFR-project

This study was conducted within a project financed by Avfallsforskningsrådet (Swedish Waste Research Council). The project entitled *Compost Quality and Potential for Use* was carried out 1995-1997 by the Department of Agricultural Engineering, SLU Alnarp, in collaboration with Movium, SLU Alnarp and The Danish Forest and Landscape Research Institute, Hørsholm, Denmark.

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All-year composting of household waste in Finland

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In this study users' practical experiences with on composting and various types of composters designed for all-year use were evaluated. Data were collected by questionnaire in the autumn 1997. The questionnaire was directed to the residents of one-family and terraced houses where composting of household waste occurred. Their addresses were obtained from municipal waste management authorities. Altogether 1,000 questionnaires were sent to Kokkola, Jyväskylä, and the Helsinki Metropolitan area. The answer rate was 56 %.

According to the questionnaire the reasons for composting are to get mould, desire to take care of enviroment, and savings in garbage disposal payments. In the Helsinki Metropolitan area, respondents were more keen on taking care of their own wastes than in other areas. The answers revealed that composted household waste was used in the garden or in the yard as soil improvement material and as fertilizer.

Freezing of composters was a common problem. More than half of the respondents reported that their composters usually froze during the winter. The composters were, however, considered to function relatively well, and only 6% reported that their composters functioned inadequately. On the whole, the respondents were reasonably satisfied with their composters, although their durability was occasionally commented.

The amount of waste was important for the functioning of composters. As the amount of waste to be composted increased, the risk of freezing decreased considerably. Successful composting also depended on the tending of the compost and the user's skills. Well-functioning composts were usually mixed weekly or monthly, and the users' knowledge about composting was good.

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An air-cooled laboratory composting system with independent control of temperature and oxygen status

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The Danish Forest and Landscape Research Institute has in collaboration with Aalborg University, Esbjerg designed a new laboratory system for investigation of composting processes with special emphasis on the fate of xenobiotic compounds in processing of MSW and waste water sludge.

The prototype system (fig. 1) consists of a 5 l reactor insulated with 10 cm of polyurethane foam (the final system will include seven such reactor units). Two Pt-100 RTD´s measure the temperature inside the compost and in the process air, respectively. A small water pump transports percolating water from the bottom of the reactor back on top of the compost to avoid uneven drying-out of the composting material.

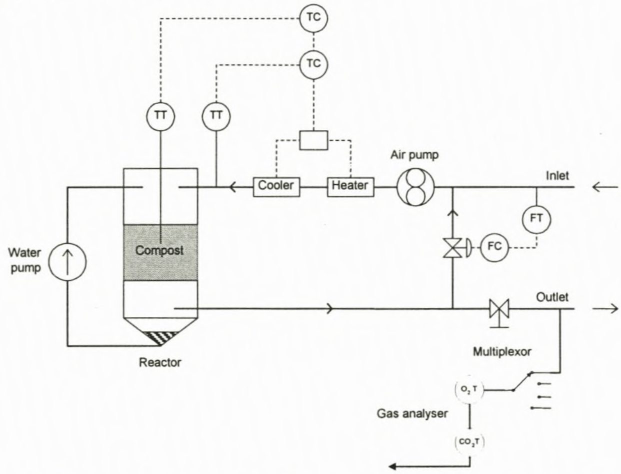

Figure 1. Process and instrumentation diagram. Temperature transmitter (TT), flow transmitter (FT), temperature controller (TC) and flow controller (FC).

Air for aeration and cooling/heating of the compost is supplied by a controllable membrane pump with a max. pumping capacity of 30 l min^{-1}. The pump is placed in series with an air-cooler, an air-heater and the reactor in a closed loop that can be used for recirculation of process air. The air flow in the loop is controlled by a solenoid valve. The loop is connected to the outside through an inlet and outlet pipe, respectively; the valve is placed on a piece of tubing connecting these two pipes. The airflow through the inlet is monitored by an electronic mass flow transmitter.

This set-up enables independent control of temperature and oxygen (and carbon dioxide) status inside the composting reactor. The oxygen status can be controlled as follows: by opening and closing the valve between the outlet and the inlet, the ratio between fresh air and recirculated process air can be changed. For example, if the valve is totally closed the pump draws fresh air through the system and no recirculation takes place. On the other hand, when the valve is fully open no air from outside passes through the compost and the recirculation is at 100%. (This situation requires a pressure drop over the inlet that is larger then the pressure drop over the entire recirculation loop). By turning the valve, any ratio between fresh air and recirculated air can be achieved. Thus, the amount of air that passes through the air-cooler and heater can be kept constant, e.g. at 30 l min^{-1}, which allows us to control the compost temperature even when no fresh air enters the system.

Oxygen and carbon dioxide concentrations of the outlet air are measured by a paramagnetic sensor and IR-spectrometry allowing for on-line calculations of oxygen consumption and carbon dioxide production, respectively.

The system is fully computerised and all parameters are under the control of the process-control and data acquisition software Genesis™ for Windows: The temperature inside the compost and the air temperature in the recirculation loop are continuously monitored and transmitted by analogue signals to the computer, as is the inlet air flow. Analogue signals to the cooler, heater and valve controls - through PID loops - the inlet air flow and the reactor temperature. The water pump is controlled by digital out-signals. Fig. 2 shows an example of temperature control during composting of simulated MSW (a mixture of dog food and sawdust).

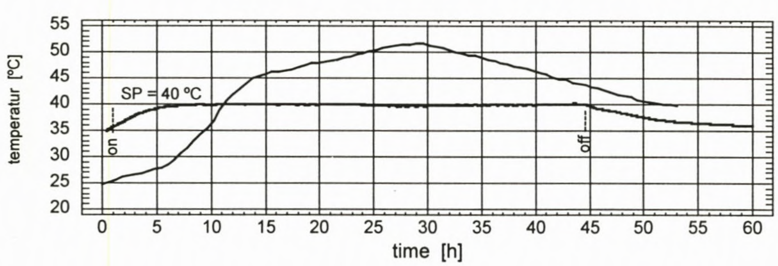

Figure 2. Example of two composting runs: one with temperature control (set-point = 40 °C) and one without.

Consequently, the system gives the user a high degree of control over important process parameters as temperature and oxygen/carbon dioxide status in the reactor. It is, therefore, possible to investigate in depth the influence of temperature on the composting process, e.g. by letting the compost follow a pre-defined temperature profile, and at the same time determine the influence of oxygen/carbon dioxide concentration on the process. Other topics to be studied in the seven parallel systems include effect of structure material on the decomposition of organic mater, elimination of pathogens and weed seeds, fate of nutrients, etc.

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Biowaste collection in ventilated vessels – field tests in Joutsa and Muurame

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Source separation is a central goal for future Finnish waste management systems. Separate treatment of biological waste is an ongoing trend in many Finnish communities. Because a lot of people, about 55%, are living in detached houses, a centralized weekly biowaste collection from all households is economically impossible. However, if source separation of biowaste cannot be demanded from all kinds of apartments, the positive downstream effects of separation on waste quality and the handling system cannot be guaranteed. Home composting is likely to be a solution for only part of the small facilities.

Ventilated biowaste vessels that can also be used as compost bins, make it possible to extend collection intervals which means that the biowaste handling costs for the housekeepers can be reduced. Field tests with such vessels have been carried out in Joutsa and Muurame since the begining of 1997.

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Compressed peat products solve the problems in biowaste collection

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Water seepage accumulate at the bottom of containers during collection of biowaste in the kitchen pail and in the biowaste bin in court-yards. Anaerobic decomposition processes may take place in the garbage due to the high water content and the lack of oxygen. The consequences are unpleasant smell, materials hard to handle, and dirty and smelling collecting bins. Biowaste collection intervals must be kept short, which cause high collection costs to the housekeepers. Unpleasant liquids even cause design problems for biowaste collection vehicles.

Central kitchens, hospitals and big apartment buildings are typical places having problems with wet biowastes. The amount of liquids squeezed out of the heavy biowastes may constitute from some litres (big apartment building) to more than 50 litres (central kitchen with no liquid separation). Some liquid problems are also caused by vapor condensated on the inner surface of the bin flowing down along the walls to the bottom of the bin. These problems can be observed also at places producing dryer biowastes. Even after normal emptying of a wet bin, about half a litre of liquids is left at the bottom. This increases hygienic problems.

Some of the problems caused by liquid biowastes can be solved by straining the biowastes in the kitchen. This method solves only partly the problems. In our project water and odour problems are solved by soaking biowaste humidity to compressed peat plates or briquets placed at the bottom of the bins, or by adding compressed peat to the biowaste collecting bag used in the kitchen. The liquid amount soaked by a compressed peat plate is normally 4-5 times its own weight. A 2 cm thick plate (weight about 1,4 kg) at the bottom of the bin soaks up to six litres of liquids. The bottom of the collection bin remains dry and clean and odour problems decrease considerably. The biowaste handling costs for the housekeepers can be reduced by allowing longer collection intervals and less washing.

An ongoing research and product development project is carried out by VTT Energy and Kekkilä Oyj. Different product alternatives have been tested under laboratory conditions and in field tests.

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Needs for more hygienic treatment of faecal matter in Finnish rural area

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Drinking water can transmit many diseases caused by micro-organisms. These problems also occur in Finland and other Nordic countries. If a drinking water epidemic happens, the reasons and sources of contamination should be identified so that the further accidents can be avoided. For that reason it is important to analyse the water source very soon during the epidemic, together with the examination of patients. Coliphages may be one interesting group to consider as an indicator for water hygiene, and this work was carried out to get more knowledge about coliphages in epidemic cases.

Materials and methods

Untreated water samples from three cases of water epidemics were examined. Conventional hygienic parameters (total coliforms, faecal coliforms, enterococci and faecal sulphite reducing clostridia) of drinking water were determined. Determinations were those accepted by the European Union according to the Finnish or European standard methods, except that clostridia were incubated in true anaerobic jars. In addition, both DNA- and RNA-coliphages (hosts *Escherichia coli* ATCC 13706 and 15597) were determined either from 100 ml and/or 10 l samples. These phages were selected, because they must originate from faeces of humans or warm-blooded animals.

The epidemic cases of the present article all happened in Southern or Central Finland. Case A happened in a sports and training centre. Soon after arrival guests often got diarrhoea, which was identified to be caused by calici virus in some cases. Drinking water was taken from a great lake through a pipe at the bottom, with a short filtration before the well. Waste water was lead to the same lake but to a seemingly downstream site (as seen from the surface).

Case B was a large epidemic in a parish, where almost all people (more than 2500 persons) who used the chlorinated tap water got diarrhoea caused by calici virus. Raw water was taken from 12 m depth of a lake; this depth and the depths 5, 9 and 14 m (almost at the bottom) were analysed for faecal indicators. Water from a brook was analysed one month later. This brook runs to the lake 100-150 m upstream from water pipe, very close to many houses in this parish.

In case C a drilled well served as drinking water source for an educational centre of a church. Young people to be prepared for their confirmation got gastro-intestinal illness identified in some cases to be caused by *Clostridium perfringens*.

Results

Raw water in case A contained total coliforms (11/100 ml), faecal coliforms (5/100 ml), DNA- and RNA-coliphages 1/100 ml and 1/100 ml or 11/10 l and 2/10 l, respectively. The levels of enterococci and faecal clostridia were under the detection levels.

The lake water of case B contained enterococci (1/100 ml) at depths of 9 m and 14 m but no other indicator bacteria. Raw water at the depths 12 m and 5 m did not show any indicators for 100 ml sample size. DNA-coliphages were found (1/10 l and 6/10 l) at the depths of 12 and 14 m, respectively, but none at other depths. No RNA-coliphages could be shown in any of the samples. Only coliphages were observed, and the results are presented in the Table 1.

Table 1. The numbers of DNA- and RNA-coliphages/100 ml in brook water at parish B.

Sites downwards to the lake	DNA-coliphages	RNA-coliphages
Tyyne (highest)	<1	<1
Eino	5	<1
Ojansuu	1	23
Suomala	2	33
Pumping site (lowest)	<1	35

In the case C no indicator bacteria were found in any of the 100 ml samples. On the contrary the number of RNA-phages was so high (much more than 1 000) in both 100 ml and 10 l samples that no accurate results can be presented. No DNA-phages could be shown.

Discussion

The faecal contamination was very clear in all three cases studied. In case B the faecal contamination of brook water was much heavier than of the lake. That was in spite of the fact that the spring sun might already have destroyed a part of the micro-organisms and dilution due to melting snow may have been another important factor reducing the concentration of micro-organisms. The faecal pollution probably appeared near Eino and Ojansuu and the increase of RNA-phages can be seen as an indication of this.

The faecal contamination in case C was clear, and the number or sources may be small, because the spectrum of micro-organisms found was narrow. It may have been only one person who emitted the RNA-phages and the pathogenic agents. In contrast, the spectrum of indicators was wide in case A, suggesting that more than one person served as sources. The sanitary waters in the case A could have moved in upstream direction at the bottom of the lake. Anyhow, the pollution may have originated from the houses themselves, because the houses affected by the epidemic in the cases A and C were situated far from other inhabitants.

A better sanitation system might have protected these water plants so that no accident had happened.

In case B faecal contamination of untreated water, originating from the parish itself, can be seen as a primary source. This contamination was concentrated at the bottom of the lake because in winter time the most contaminated waste water has the highest specific weight. A secondary reason was the insufficient chlorination due to organic compounds in the water (and maybe biofilms on the inner walls of the pipes), since organic matter is known to use up chlorine.

In these cases coliphages seemed to serve as the most sensitive indicators.

Conclusions

An inadequate treatment or no treatment of human sanitary waters can not be accepted. More research should be done to make *nice, easy but safe* toilet systems available in rural areas, as well as for training centres where people come only for short periods. Visitors can not learn how to use a complicated latrine system during a short time they like to concentrate to their mental or physical development.

Coliphage system seems to be a powerful tool for water hygiene. It should be developed so that analysis of 10 l samples would be easy and more data found in different raw waters.

If surface waters are used as water source for drinking water, the bottom may not always be the best place to take raw water. The efficiency of disinfection should be studied regularly by sampling pipe waters. For chlorine, for instance, there are cheap and easy test methods.

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A process for combined treatment of solid municipal wastes and sludges for producing biogas

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At present there is a problem with processing of municipal wastes and sludge from the town purification plants. Of special importance is processing and neutralization of excessive sludge from the Leningrad region because they contain up to 10% of heavy metals and a considerable amount of pathogenic microorganisms presenting a danger to vital activity.

An anaerobic technology for combined processing of municipal wastes and sludge sediment is proposed, which makes it possible not only to cut the volume of the wastes by 30-40% but also to produce a considerable amount of biogas which can be used for obtaining energy. The process parameters - moisture content of the fermented mass, the temperature - and their influence on the gas formation effectiveness are discussed. The microbiological plating and evaluation of the number of aerobic and anaerobic microorganisms showed that the fermented mass was dominated by an anaerobic microflora, 1 g dry matter contained up 10^7 cells.

The economic parameters of this process are considered and the prospects for its implementation in practice are discussed using the field composting plot at an experimental plant for mechanized processing of the municipal waste as example.

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Ecological assessment of sewage sludge application

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The present long term field studies were started in 1995 and 1996 and are planned to continue until the end of 1999. The aim is to assess the effects of sewage sludge on soil fauna and microbial processes in fields annual sown with cereal (winter-wheat or barley) and fields once sown with barley and clover-grass. Additionally the effects of sewage sludge and contaminants found in sewage sludge have been studied in the laboratory.

Methods

Three levels of sludge and cow dung (3.5, 7, 21 t d.w. ha^{-1}) are compared to control plots without fertilization. Sludge from two different waste water treatment plants (WWTP 1 and WWTP 2) have been applied to each crop, and two different soil types have been used, a sandy soil at Lundgaard and a sandy loam at Askov. A number of soil fauna and microbial processes were measured in the first year. In subsequent years only the number of soil microarthopods and potential nitrification activity have been monitored. In 1998 the cereal fields were re-amended with sludge 3 months prior to sampling. The concentration of selected xenobiotics were generally in accordance with the Danish ordinance on sewage sludge (Table 1). Cut-off values for four chemicals are included in the latest regulation, i.e. PAH's, LAS, nonylphenol (+ 1-, 2-ethoxylates) and the phthalate DEHP.

Table 1. Concentration of selected chemicals in sludge from two waste water treatment plants (WWTP)

Compound	WWTP1 (mg kg^{-1})	WWTP2 (mg kg^{-1})	Danish cut-off values (mg kg^{-1})
PAHs	3.9	9.5	6
DEHP	14.0	31.0	100
NP	35.0	140.0	50
LAS	1,700	1,100	2,600

Results

Laboratory experiments

A comparison between a direct exposure of springtails in soil and exposure through spiked sludge mixed with the soil, showed that both detergent LAS and nonylphenol (NP) had lower toxicity when applied to the soil through spiked sludge (Table 2). Reproduction, growth and survival of earthworms was reduced by 50% at 13.7, 23.9 and >40 mg NP kg^{-1}.

Table 2. Reproduction effects (number of offspring) of LAS and NP to Folsomia fimetaria in a sandy soil. The chemicals were either mixed into the soil directly or via spiked sludge.

	EC_{10} (mg kg^{-1})	EC_{50} (mg kg^{-1})
LAS	208	444
LAS (spiked)	757	1,102
NP	27.2	44.2
NP (spiked)	48.7	58.4

The post management of sewage sludge may play a role in the toxicity towards springtails. Acute toxicity tests and food preference experiments (sludge, cow dung, yeast and nothing) with *F. fimetaria* have shown that fresh sludge is less toxic than anaerobically digested sludge.

Field studies

The field studies running for 3 years in the cereal field (Table 3) and for 2½ years in the clover-grass (not shown) did not show any negative effects of sewage sludge or cow dung on the microarthropod populations. Earthworm numbers were monitored during the first year. A significant stimulation in number and biomass was observed at the low dose of sewage sludge, whereas no difference to the control was found at the two higher concentrations. For the nitrification process only stimulation was observed.

Table 3. Effects of sludge and cow dung on soil microarthropods, 1000 indv. pr. m^2, in a cereal crop. Asterisk, *, indicates a significant difference from the control (LSD, $P \leq 5\%$). MA: total microarthropods, Coll: Collembola, Ffim: *F. fimetaria*.

Year	Dosage t DM/ha		MA	COLL	ACARI	FFIM
1995	Control	0	6	3	3	0
	Sludge	3.5	8	4	4	1
	-	7	6	2	4	0
	-	21	8	3	5	1
	Dung	3.5	7	4	3	1
	-	7	10	5	5	2
	-	21	*12	5	7	1
1996	Control	0	83	16	67	1
	Sludge	3.5	89	30	59	2
	-	7	*82	*21	61	1
	-	21	94	32	62	4
	Dung	3.5	84	18	66	*0
	-	7	*134	*56	*79	*4
	-	21	*150	*86	64	*42
1997	Control	0	176	39	138	2
	Sludge	3.5	186	45	141	7
	-	7	191	68	*123	2
	-	21	*274	*104	169	6
	Dung	3.5	201	61	140	5
	-	7	203	61	142	4
	-	21	*289	76	*214	2
1998	Control	0	82	18	64	2
	Sludge	3.5	67	22	45	4
	-	7	63	22	41	4
	-	21	113	*46	67	*15
	Dung	3.5	79	18	60	3
	-	7	74	21	53	7
	-	21	*145	61	*84	22

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Soil solution chemistry and soil microbiology following sewage sludge application to agricultural soils

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To evaluate the short and long term ecological concequences of sludge application, a comprehensive understanding of the interactions between sludge and soil solution chemistry, as well as soil microbiology is needed. In this long term project (started in 1996) a meso-scale experiment was designed. Sewage sludge, stabilised in three different ways, was added to three soils: loam, sandy loam and loamy sand. Sludge was applied to 30 kilograms of soil at rates of 20 tons/ha or 200 tons/ha (dry weight), which generated 21 sample treatments, including control. Twenty tons/ha equals the maximum amount that may be added within a ten-year period in Norway .

Soil solution samplers collect soil solution during precipitation events. Nitrification potential, respiration activity, and the activity of the enzymes acid phosphatase, urease and dehydrogenase has so far been measured in the soil, and macronutrients and heavy metals have been determined in soil solutions in the different treatments.

The amount of cadmium, zinc and copper in the soil increased by 50-350% with application of 200 tons/ha. In some of these treatments, however, the soil solution concentration of cadmium and zinc increased to about 100 times the level in the reference soils.

Results show that the soil respiration rate and enzymatic activities increased 2-3 times in the soil amended with 200 tons/ha, and that only minor changes occurred in the 20 ton/ha treatments during the first 7-9 months after application. The nitrification potential was reduced markedly the first weeks after 200 tons/ha application, but within 9 months the potential is about twice the level in the reference soils.

Both the amount of sludge, sludge stabilisation and soil type are important both for soil solution chemistry and the microbiological response of sludge application. Integration of soil solution chemistry and microbiology may improve risk assessment of sewage sludge when evaluating the beneficial and detrimental concequences of sludge application to soil.

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Assessment of compost maturity

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Abstract
The purpose of the Nordtest project 1363/97 was to investigate the status on quality testing of compost among Nordic compost producers, and to review available test methods for compost maturity, ecotoxicology and organic pollutants. The study included a survey of the current status of the quality testing at composting facilities in the Nordic countries, as well as literature studies on suitable methods for assessing compost maturity and ecotoxicity and on organic pollutants in compost. The focus of this paper is on the results from the questionnaire and literature review on compost maturity tests.

Compost maturity can be defined as a measure of the process-related readiness for use. The maturity critically affects the successful utilisation of compost in agriculture. Especially the maturity is of great importance when the compost is used prior to sowing or in growth substrate. Immature compost may have severe effects on soil environment and plant growth.
A questionnaire was distributed to known Nordic composting facilities. Answers received revealed that the majority of the Nordic compost producers mainly control compost quality using simple process parameters and chemical analysis of the compost. About 30% controlled the maturity using plant assays, less than 10% of the respondents reported use of self-heating tests.

Today there are no national regulations or standards on compost maturity test methods in the Nordic countries. In the literature more than 30 test methods have been suggested to predict the maturity of compost. The suggested methods can be divided into chemical methods, physical methods, microbiological methods and plant assays.

Among the suggested maturity test methods, most of the available standards are based on chemical methods. The best documentation on suitability for compost maturity testing exists for the methods that have been implemented in regulations, standards and guidelines around the world. These are especially the plant assays, the self-heating test, as well as some chemical test methods.

Although plant assays provide valuable information, the methods involved are time consuming and expensive. Promising physical and chemical methods show deficiencies especially in cross sensitivity and relevance. The paper concludes with the recommendation

that, unless plant assays are conducted, a set of different maturity tests should be used to predict compost maturity.

Preface

This paper is based on the findings of the Nordtest-Project 1363/97 "Assessment of Compost Maturity and Ecotoxicity". The purpose of this project was to identify problems associated with quality evaluation and utilisation of compost in the Nordic countries. The project report includes the results from a questionnaire that was distributed to compost treatment plants in the Nordic countries and a literature review on methods for assessing the quality of compost and on organic pollutants in compost. The project was coordinated by Dr. Merja Itavaara at VTT, Finland.

Introduction

Compost production from organic waste is rapidly increasing in the Nordic Countries. The total annual compost production from organic waste in Norway is given as an example. From a mean production of approximately 7.000 t/a 1995 the production increased to 60.000 t/a in 1997 and is expected to reach 100-150.000 t/a by year 2000. (Vethe & Lystad, 1997)

National regulations on compost use include specific limitations on content of pollutants like heavy metals. These limitations are only to a small extent related to the quality of the compost. Concerning maturity and stability, only general demands are given (e.g. in the Norwegian Fertiliser Act, saying: "The compost must to stabilised to an extent where no malodour or other environmental problems occur.") (Landbruksdepartementet, 1996)

The terms " compost maturity" and "stability" are often confused. In this paper "compost maturity" is addressed in a wide sense, as a measure of the process-related readiness for use. This readiness depends upon several factors, e.g. the degree of decomposition and levels of phytotoxic compounds like ammonia and volatile organic acids, and thereby, also includes the concept of "compost stability".

The degree of maturity is of major importance for the successful use of compost. Immature compost can have harmful effects on plants and soil ecosystems, especially if the compost is applied before sowing or if the compost is used as a growth substrate (Inbar *et al.*, 1991).

Among the typical effects of using immature compost are oxygen depletion in the root system, loss of nitrogen, increasing solubility of heavy metals, formation of phytotoxic substances, regrowth of pathogens and harmful microbes, as well as nutrient leaching (Zucconi, 1981; Cottenie, 1981; Zucconi *et al.*, 1985; Jimenez & Garcia, 1989; Inbar *et al.*, 1990).

The aim of maturity testing is to predict any negative impact of the compost when applied to soil systems or stored. A maturity test should not be applied to provide information about effects related to the content of nutrients and toxic compounds introduced to the compost prior to composting.

Owing to the lack of suitable analytical methods, the composting industry has for decades relied on plant assays to determine plant growth response. Even though efficient plant assays have been developed recently, these are in general both time consuming and expensive.

Meanwhile there has been a constant search for physical, chemical and microbiological methods that can provide quick and reliable information about maturity. Some examine the chemical changes and the changes in biological activity occuring during the decomposition of organic matter in order to predict the degree of degradation. Recent methods use physical or chemical characterisation (dry and wet chemistry) to characterise the compost based on the presence or absence of certain substances characteristic for the individual stages of the composting process (Epstein, 1997).

One problem encountered in the search for an overall method for compost maturity testing is the variety of composts. The composition of the compost depends mainly on the raw materials (sludge, garden waste, municipal waste, industrial waste, and amendments) and the composting technique. This means that a method that has given reliable results for one type of compost is not necessarily reliable with another.

Use of quality testing among Nordic compost producers

Results from the questionnaire

Purpose and methods

The aim of the questionnaire was to survey the current situation on quality testing of compost among Nordic compost producers. For that purpose the questionnaire was translated into each language and distributed to the known compost producers in Finland, Iceland, Norway and Sweden December 1997. For Denmark, information on 133 compost facilities registered in the Danish Compost Statistics 1996, compiled on behalf of the Danish Environmental Agency, was used. A copy of the complete questionnaire used, can be made available from the author.

Results

In total 82 answers were received: 43 from Finland, 31 from Norway, 7 from Sweden and one from the only facility in Iceland.

General data

Overall 86% of the reported facilities are in public ownership, 14% are private. In Sweden and Norway, the part of private owned facilities reported is larger with 57 and 29%, respectively.

Among the answers received, 73% of the compost facilities produced less than 5.000 tons compost per year. Average production was 3373 t/a. The Finnish composting plants were, on average, larger with an average production of 8.398 t/a. The results indicate that a majority of the Nordic compost production today is taking place in Denmark and Finland.

The data on the reported treatment time show variations. Some of the facilities have reported only the time for the intensive treatment, whereas the majority has reported the time including maturation. The calculated average compost treatment time of 8.7 months should therefore be seen as a reported minimum.

A majority of 87% of Nordic compost facilities is based upon open systems like windrow or 'mattress' composting. Reactor systems contribute only 8%. Anaerobic systems were only reported for Denmark and Finland. Regarding the composted materials, the questionnaire show national differences. In Denmark the majority are green waste composting facilities, whereas in Finland the majority of the facilities compost sewage sludge. In Norway, the majority are biowaste facilities.

Table 1. Ownership and production of Nordic composting facilities

Country	Denmark	Finland	Iceland*)	Norway	Sweden	Total
Public (municipality/-ies)	125	35	1	22	3	186
in percent	*94%*	*80%*	*100%*	*7 %*	*43%*	*86%*
Private	8	9		9	4	30
in percent	*6%*	*2 %*	*0%*	*29%*	*57%*	*14%*
Total amount of compost produced (t/a)	286,418	335,901	2,250	80,563	3,185	708,317
Mean compost production per facility (t/a)	2,154	8,398	2,250	2,685	531	3,373
Mean treatment time	**)	10,9	2,5	6,0	7,6	8,7

*) Data from one facility **) No data available

Table 2. Composting system and raw materials in Nordic composting facilities

Composting system	DK	FIN	IS	N	S	Total (in %)
Windrow composting	117	31	1	21	3	83%
Madrass composting	7			1	1	4%
Reactor composting	2	9		5	1	8%
Wet composting				4		2%
Anaerobic treatment	3	1				2%
Other		1			1	1%
Composted materials (excl. amendment)	DK	FIN	IS	N	S	Total
Garden waste	127	2	1	5		135
Sewage sludge	8	37		12		57
Source separated bio waste	16	16		15	3	50
Manure	15	6		10	3	34
Industrial waste & sludge		12		6	1	19
Other material		4		11		15

Process and quality testing

Regarding compost quality, the compost facilities were asked: *"What methods are used to control the compost process and quality at your plant?"*

The answers received (Danish facilities not included) showed a need for careful interpretation, as some facilities probably have not reported all their efforts to control the compost process and quality. There is reason to believe that at least temperature measurements and chemical analyses are used more widely than actually reported. (results given in Table 3).

The process and quality control methods reported show that most facilities use temperature and treatment time to control the process (78% and 85% respectively). Measuring pH and O_2/CO_2 content is reported to be less common (49% and 39% respectively).

Among the methods more closely related to product quality, a total average of 50% reported the use of chemical analysis. The extent of use of chemical analysis for maturity testing is not known. The more specific maturity tests, self-heating tests and plant assays are reported for a total of 9% and 28% of the facilities, respectively.

Microbiological analyses that were reported to be used, were hygiene control analysis (such as Salmonella).

Comparing the answers on a national level, the specific maturity tests were reported to be more widely used at the Norwegian facilities than at the Swedish and Finnish facilities. Furthermore, the process control with pH and O_2-/CO_2-measurements was more often used in Norway than in the other Nordic countries. Facilities in Finland and Sweden seem, to a greater extent, to rely on chemical analysis and temperature, treatment time and odour to control process and product quality.

Table 3. Reported quality control methods (in % of total answers)

Methods for process and product control	Finland	Iceland*	Norway	Sweden	Total
Temperature measurements	67	100	90	86	78
Treatment time	93		77	86	85
Odour	51		39	86	49
pH	26		58	43	39
O_2-/CO_2-measurements	9		52	29	27
Self heating test (Rottegrad)	5		16	0	9
Plant assays	23		39	14	28
Chemical analysis	33	100	68	71	50
Microbiological analysis	7		42	43	23

* One answer

Regulations on maturity and maturity testing

Nordic countries

In the Nordic countries existing regulations for the trade and utilisation of compost are based on national legislation. The relevant legislation contains specifications regarding the content of heavy metals, and to some extent the content of foreign matter, salinity, organic compounds and pathogens. Some regulations state requirements to the composting process. On compost maturity there are few specifications.

The Norwegian Fertiliser Act states that products originating from organic waste are to be stabilised to a point where it does not cause odour or other environmental problems through storage and use. Documentation of compost stability or maturity may be requested by the Norwegian Agricultural Inspection Service, but no requirements to the method of documentation are given (Landbruksdepartementet, 1996).

In Sweden, Denmark, Finland and Iceland there are no regulations on the maturity of compost. A national quality assurance system is in preparation by the Swedish Association of Waste Management, and will probably recommend the use of one respiratory method and one chemical method for the prediction of maturity (Lundeberg, 1998).

Other countries

Some countries outside Scandinavia have detailed regulations on compost maturity or stability. A short review of such standards and regulations is given below.

In Germany the "Information sheet LAGA M10" gives instructions and information concerning the quality of compost. The LAGA M10 defines the method of self-heating in dewar-flasks as a suitable method for predicting the "Rottegrad" or degree of degradation, and a cress (*Lepidium sativum*) growth test to predict the impacts on plants.

The Dutch KIWA certificate for compost states a requirement for a germination test (Nieuwsbrief De Compost|, 1997)

In Canada, three standards on compost define a maturity indicator test. According to these standards, compost can be declared mature if it meets two of the following requirements:

- C/N ratio <=25,
- Oxygen uptake rate <= 150 mg O_2/kg volatile solids per hour, and
- Germination of cress seeds and of radish (*Raphanus sativus*) seeds. (The Compost Council of Canada, 1998)

The Australian Standard for Composts, Soil Conditioners and Mulches AS 4454, published in 1997, defines compost as mature when:

- Nitrate-N/ammonium-N ratio is equal to or above 0,14
- Ammonium-N content in extract is below 300 mg/L
- Toxicity index (cress germination) is equal to or above 60 (only required for compost as soil conditioners) (Standards Australia, 1997)

Literature review on maturity test methods

Criteria for a suitable compost maturity test

For compost maturity tests to be used for prediction and documentation of compost maturity, a set of criteria must be met. These are outlined below:

A test method for compost maturity should first of all be universally applicable on different compost materials. The complex structure of compost and compost maturing makes this a difficult task. The method should give relevant information for all kinds of compost. The relevance of a compost maturity test is closely related to the aim of the method.

Secondly, the test method must be well documented. For recently suggested test methods this criterion may be difficult to meet, but this is normally only a matter of testing.
The method must furthermore meet certain standards on reproducibility, accuracy and sensitivity. The demands for accuracy must, however, not exceed the given uncertainties in testing of biologically active material.

A recommended method should be secure against attempts to influence the test result. In this sense, chemical test methods that include the presence of a certain compound are less secure against unwanted influence than methods including the absence of chemical compounds. The same problem is encountered with the tests utilising microbiological activity, where the absence of - or small extent of - activity is evaluated as maturity (even though cytotoxic compounds may be the reason for the low activity).

Finally a maturity test method must be practical and not too expensive in use. It must be considered advantageous if the same method that is performed by an authorised laboratory can also be performed at the compost facility.

Available compost maturity tests

More than 30 compost maturity tests have been suggested in the literature during the last decades. Some of these tests have been implemented into national standards and guidelines (see page 108). In Table 4 a summary of the methods suggested or currently in use are listed. The methods can be divided into:

- Physical methods
- Chemical methods
- Microbiological methods
- Plant assays

Table 4. Methods suggested or currently in use to assess compost maturity (adapted from Epstein 1997)

Physical methods	Chemical methods	Microbiological tests and activity	Plant assays
Temperature	pH	Respiration-O_2 consumption	Germination tests
Colour, odour, structure	Cation exchange capacity	Respiration-CO_2 production	Root elongation tests
Self-heating test	Reduction of organic matter	The Solvita test	Growth tests
Spectroscopy	Carbon/nitrogen ratio	Enzyme activity tests	
Optical methods	Nitrogen species		
	Organic chemical constituents		
	Phytotoxic agents		
	Humification parameters		

Another way of dividing the suggested maturity tests is to divide the methods according to the aim of the method. Whereas the plant assays aim at predicting the direct effects of the compost on plants, the microbiological methods and some physical and chemical methods aim at predicting the degree of degradation as a measure of maturity (e.g. the self-heating test and methods utilising reduction in organic matter).

Other chemical and physical tests aim at characterising the compost based on the presence and absence of certain substances or substance groups typical for individual stages of the composting process. Typical for these methods are the spectroscopy test and the method analysing the NO_3/NH_4-ratio.

Evaluation of compost maturity tests

Most of the available standards among the suggested methods are chemical. The best documentation of the suitability for compost maturity testing exists for the methods that have been implemented in standards and guidelines around the world. These are especially the plant assays, the self-heating test as well as some chemical test methods (see above).

Among the test methods, plant assays provide perhaps the most valuable information on compost maturity. The germination or growth of seedlings or plants in compost is in many ways a direct measurement of the process-related readiness for use. But the results from plant assays can also contain information not related to compost maturity, such as nutrition or content of toxic compounds that are not related to the composting process and compost maturity. This information may be wanted, but in some cases these findings are not relevant. Furthermore, as previously mentioned, these methods are relatively time consuming and expensive.

Among promising physical, chemical and microbiological methods, the aim of the methods is often limited to determining the degree of degradation. This applies to the respiration methods, the self-heating test and several chemical test methods. The degree of degradation is, however, only one of the factors in compost maturity. The presence of process related cytotoxic compounds would reduce the cross sensitivity of these methods. Among the methods predicting the degree of degradation, the self-heating test and the highly practical Solvita test have advantages in easy performance and documented accuracy.

The methods based on characterisation of the compost range from simple chemical analysis of C/N-ratio to more sophisticated analysis (e.g. spectroscopy). Recently proposed methods, like C/N-ratio from water extract, NO_3/NH_4-ratio and some humification parameters, have shown an improved sensitivity and accuracy. The general deficiency of these methods remains, however, that different types of compost can lead to different results depending on initial ingredients or even final amending (e.g. by adding nitrate at the end).

Thus, it is the conclusion of this report that, unless plant assays are performed, a set of different maturity tests should be performed. Only through use of a carefully considered set of maturity test methods, the deficiencies of the different methods can be reduced. When conducting plant assays one should keep in mind that these tests may give more information than only about compost maturity.

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J. Petersen & S.O. Petersen (eds.) 1999. Use of municipal organic waste
Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
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Danish Institute of Agricultural Sciences report no. 13

Biological activity of municipal composts as an index of their quality.

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Abstract

It is shown that for evaluating the compost quality, along with traditional agrochemical parameters, it is expedient to apply an all-round analysis of their activity, which comprises such indices as an enzymatic activity, composition of the microbial cenosis, ability to inhibit the phytopathogenic microflora, the content of physiologically active substances - plant growth promoters.

An all-round method for evaluating compost quality by biological parameters was applied for a comparative study of the composts produced from municipal wastes of Sweden and Czechia. The recommended method will make it possible to expand the spectrum of the biocompost application and optimize the technology of its application in sustainable agriculture.

Introduction

At present there are technologies which allow effective processing of municipal wastes into biocomposts. Introduction of such composts into agricultural production practice requires an agroecological monitoring of their composition and quality. Control of the quality and fertilizing value of these composts is traditionally conducted by the content of NPK and heavy metals and often without the analysis of physiological, biochemical and microbiological peculiarities of such substrates.

At the same time, an all-round evaluation of the biological activity of the composts will make it possible to not only evaluate their quality objectively, but to expand the prospects of their application in agriculture. For this purpose we have developed an all-round method for evaluating the quality of the composts which includes such indices as an enzymatic activity, the composition of the microbial cenosis, the ability to inhibit the phytopathogenic microflora and the content of physiologically active plant growth promoting substances.

Materials and methods of the study

In this work there were used the samples of the composts produced after thermal fermentation of municipal wastes. The samples were offered by Company "Rondeco" (Sweden) and Factory of town Nova Raka (Czechia). The biocompost of Company "Rondeco" was produced after thermal fermentation of municipal wastes in a biofermenter at 60°C during 3 days. After fermentation inorganic constituents like glass, metal, plastic atc. were removed from compost. Then the compost is kept in the air for 5 weeks and pelleted.

The Czech compost was produced from municipal wastes which underwent a preliminary sorting. Waste paper, cloth, glass and metals were removed from them. After a sieve and a crusher the organic mass goes into a biofermenter with aeration. Biocomposting proceeded at 75°C for 10-14 days, then it matures in the air for 6 weeks. During maturing the compost is turned over 2 times and aerated. The compost composition was: 70% biowastes, 20% sawdust, chalk, 10% tree bark.

Isolation of clean cultures of microorganisms from the composts was carried out using standard methods on a tryptone soya agar (Alef & Nannipieri, 1995). After selection and isolation of dominating cultures they were identified using a method of cellular fat acid determination. For this the isolated cultures were preliminary grown on the tryptone soya agar at 28°C for 24 hours. The cellular fat acids were extracted then analysed using a gas chromotography method (Hewlett-Packard 5890A) and identified with Microbial Identification System using Aerobic Library version 3.9 (MIDI Inc., Newark, DE, USA).

Evaluation of agrochemical parameters and enzymatic activity of the composts was done using standard methods [Alef & Nannipieri, 1995) with some modifications. For evaluating the antibacterial activity of the composts, the test cultures of pathogenic fungi from the collection of All-Russia Research Institute of Agricultural Microbiology were used. Cultures of the following pathogens from the class of imperfect fungi were taken as test objects:

1. *Bipolaris sorokiniana* - agent of common leaf spot, black germ of grain and root rot of grain crops;
2. *Verticillium dahliae* - agent of verticillium wilt of tomatoes and other agricultural crops;
3. *Fusarium oxysporum* - agent of fusarial wilt of tomato crops;
4. *Fusarium culmorum* - agent of root and stem rot of grain crops, fusariosis of spike;
5. *Rhizoctonia solani* - agent of root rot of many agricultural crops;
6. *Sclerotinia sp.* - agent of sclerotiniose of agricultural crops.

Determination of antifungal activity of biocomposts was carried out in experiments *in vitro*, the method of agar blocks (Bilai, 1982) was used in our modification.

Table 1. Agrochemical parameters of biocomposts

Variant	Moisture content, %	N total	P total	K total	C total
		in % to absolutely dry matter			
Swedish compost	36	2,0	0,5	0,3	27,6
Czech compost	37	0,7	0,4	0,2	23

Results

As it follows from the data presented in Table 1, the agrochemical parameters of the composts under study have some differences. The Czech compost has three times less nitrogen than the Swedish one and has a little less carbon, phosphorus and potassium. Analysis of the microflora predominating in the biocomposts showed that the isolated cultures differed significantly and in each substrate a specific biocenosis develops (Table 2).

Table 2. Predominating microflora isolated from municipal composts

Swedish compost	Predominating %	Czech compost	Predominating %
Cellulomonas flavigena	30	*Arthrobacter ilicis*	18
Bacillus laterosporus	10	*Micrococcus varians*	22
Gordona bronchialis	10	*Bacillus mycoides*	11
Micrococcus kristinae	10		
Bacillus pumilus	29		
Micrococcus lylae	10		

In the Swedish compost the diversity of microorganisms was wider than in the Czech one (6 and 3 cultures respectively), it should be noted that the Swedish compost contained more bacteria possessing growth promoting activity (g. *Bacillus* - 39%; g. *Gordona* - 10%). Apart from that, it is known that the bacteria of g. *Bacillus* are capable to produce antibiotic substances and suppress growth of phytopathogenic fungi and microorganisms (Junge *et al.*, 1990 and Sarhan, 1989).

The results of the study of antifungal activity of municipal composts against 6 phytopathogenic fungi in the experiments *in vitro* are presented in Table 3.

Table 3. Antifungal activity of composts in experiments *in vitro*, concentration of substances in agarized medium - 2.5%.

Variant of experiment	Suppression of the growth of mycelium of fungi, %											
	B. sorokiniana		*V. dahliae*		*Sclerotinia sp.*		*R. solani*		*F.oxysporum*		*F.culmorum*	
	days of cultivation											
	3-rd	7-th	3-rd	7-th	3-rd	7-th	3-rd	7-th	3-rd	7-th	3-rd	7-th
Swedish compost	68	100	23	74	66	69	61	62	74	72	57	75
Czech compost	68	75	15	100	62	72	60	64	76	100	57	67

The data testify to a high inhibiting activity of biocomposts against mycelial growth of the test pathogens. Antagonistic activity of biocomposts on the seventh day of cultivation reaches 61.6-100%. The studied test-pathogens differ in their sensitivity to the composts. Thus, the Swedish compost completely suppresses the growth of fungus *Bipolaris sorokiniana* mycelium (by 100%) on the seventh day of cultivation, whereas the Czech compost suppresses the mycelium growth of this fungus by 75%. A reverse picture was observed with respect to fungus *Fusarium oxysporum*. On the 7-th day of cultivation the fungus completely lyses on the medium containing the Czech compost, and the Swedish compost suppresses the growth of the mycelium of this fungus by 72.0%. Different sensitivity of the test-fungi to the studied composts can be observed due to the fact that within 3-7 days of cultivation the antagonistic activity of the Swedish compost with respect to *Bipolaris sorokiniana* and *Fusarium culmorum* sharply increases respectively from 68.2 to 100%; from 57.0 to 75.0%; with respect to *Verticillium dahliae* from 23.1 to 74%. The antagonistic activity of this compost-with respect to fungi *Sclerotinia sp.*, *Rhizoctonia solani*, *Fusarium oxysporum* is high initially and changes slightly with time. A similar regularity is also characteristic of the Czech compost.

Thus, on the basis of the obtained results a conclusion may be drawn that the biocomposts from municipal wastes possessed a sufficiently high antifungal activity and can suppress the growth of phytopathogenic fungi. Obviously, the observed differences in the spectrum of antagonistic activity of the Czech and Swedish composts are associated with the difference in the composition of microflora of these composts.

A number of methods for evaluating the enzymes responsible for biotransformation of the organic substrate was developed for the evaluating the biochemical activity of the composts. The following enzymes were chosen: Catalase - an index of oxidation-reduction processes, the enzyme correlating with a number of aerobic microorganisms; protease, urease, amylase, cellulase - indices of the intensity of the process of degradation of organic substances - hydrolytic enzymes.

As it follows from the data presented in Table 4, both studied composts are characterized by a sufficiently high level of proteolytic activity which is apparently associated with a high activity of microorganisms performing the proteolysis of the protein substrates.

Table 4 . Enzymatic characteristics of the municipal wastes.

Variant of experiment	Protease	Amylase	Urease	Cellulase	Catalase
Swedish compost	972	337	6	270	51
Czech compost	1020	63	2	170	2

Protease - activity is expressed in mg of hydrolysed casein per 100 of dry matter for 24 hours.

Amylase - activity of amylolitic enzymes is expressed in mg of hydrolyzed starch per 1 g of dry matter per 1 hour.

Urease - activity is expressed in mg NH_4^+ per 1 g of dry matter per 1 hour.

Cellulase - activity is expressed in mg of glucose per 100 g of dry matter per 24 hours.

Catalase - activity is expressed in mg 0.1N $KMnO_4$ per 1 g of dry matter per 1 hour.

As distinct from the Czech compost, the Swedish one is characterized by a higher activity of the hydrolytic enzymes: urease, cellulase and catalase which testifies to a higher functional activity of microflora of this composts. It is possibly connected with the a large variety of microorganisms in the Swedish compost revealed by us.

Thus, a comparative evaluation by a biological activity of the two composts showed that the Swedish compost differs from the Czech one by a greater variety of microflora and a higher enzymatic activity which characterizes it as a compost with an increased biological activity. On the basis of the conducted studies, application of the biological parameters for evaluating the compost quality can be recommended. This will make it possible not only to expand the spectrum of its application, but also to optimize the technology of application in sustainable agriculture.

The work has been carried out under the program of the international project "Microbial communities" (1997-2000) financed by The Ministry of Science of Russia, and under the program "Integration of science and high school", the project "North-Western educational and scientific centre "Biotechnology".

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Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
Session 4
Danish Institute of Agricultural Sciences report no. 13

C and N turnover during composting of straw

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Introduction

Peat is the dominant growth medium for potted plants produced in greenhouses. It is a slow renewable resource that might become restricted due to conservation, and compost could be an alternative to peat as a growing medium in horticulture. Compost can be made from selected crops to obtain uniform products, or from household, park and garden waste. In research on composting of biological wastes some attention has been given to the plant nutrients of the composted products (Senesi, 1989).

The aim of this project was to improve the general understanding of the natural chemical-physical conditions and microbiological processes going on during composting of biological materials. A further aim was to develop a well defined and uniform compost based on raw materials suitable as growth media for horticulture. This paper focuses on changes in the chemical properties and degradation of the fibre fractions (hemicellulose, cellulose and lignin) during composting based on frequent sampling, especially during the heating and cooling phases. The results obtained will be used for selection of compost mixtures and procedures most suitable for cultivation purposes.

Materials and methods

Two different systems were used for the composting experiment, an 800 L wooden box and a 500 L cylindrical reactor, both with insulated walls. The box had only natural air exchange caused by heat convection, while the reactor had ability for forced aeration (250 L h^{-1}). Both systems were equipped with sensors for continuous registration of temperature. As basis for the composting, one type of plant material (*Miscanthus*) was selected. The shredded straw was mixed with pig slurry. Due to different mixing procedures, the starting material of the two systems were different. In the box the mixing was done by loosely packing of the material resulting in an initial *Miscanthus*/slurry ratio of 1:2 while in the reactor system the mixing was done using excess slurry resulting in a ratio of 1:7. This difference was also reflected in the C/N-ratio and the ash content. The box system had a relatively low ash content and an initial C/N ratio of 25 while the mixing in the reactor system resulted in a very high ash content and a C/N ratio of 16. Initial water content in both systems was 70% w.w. (wet weight). In Table 1 properties of the raw materials are listed. The water content in the box was adjusted 1-2 times every week to keep the water content between 50 and 70 % and the compost was turned twice a month to maintain aerobic conditions. In the reactor the water content was adjusted by a combination of draining and recirculation of excessive water and aeration.

Table 1. Chemical characterization of *Miscanthus* straw and pig slurry used for composting.

	Dry matter %	Total C % of d.w.	Total N % of d.w.	NH_4^+-N % of d.w.	C/N	Hemi-cellulose % of d.w.	Cellulose % of d.w.	Lignin % of d.w.
Miscanthus	77.2	44.8	0.57	< 0.01	79	23.9	45.4	13.8
Pig slurry	5.3	41.7	12.0	9.0	3.5	n.d.	n.d.	n.d.

n.d. = not determined

Sampling was initiated as soon as the mixed material was placed in the two systems. Then sampling took place according to changes in temperature (every 5°C until day 10). Sampling intervals were then once a day for the following 5 days and weekly during the remaining experimental period (total 190 days for the box system and 160 days for the reactor system). At each sampling event five subsamples were randomly taken, pooled and mixed into one sample and stored at 2°C until analysis.

Results and discussion

Temperature and water content for the two systems are shown in Fig. 1. The heating period (initial mesophilic phase) took place at the beginning of the composting and temperature maxima were reached within 2 days in the box (70°C) and after 3 days in the reactor (65°C). The duration of the heating phase was from day 1-10 in the box and day 3-11 in the reactor. The water content was different in the two systems. Initial values were similar (69 and 73%, respectively). In the box the water content decreased to 43% by day 23, while a constant level of about 75% was seen in the reactor until day 26. Rewetting the material in the box to a water content of 53% and turning of the material, resulted in a moderate reheating (temperature rise to 33°C). A corresponding reheating was seen in the reactor after an increase in water content to 83% as a result of recirculation of drainage. Both systems ended with similar water contents of about 80 % which is fairly close to the limit of a well aerated composting process (Golueke, 1972).

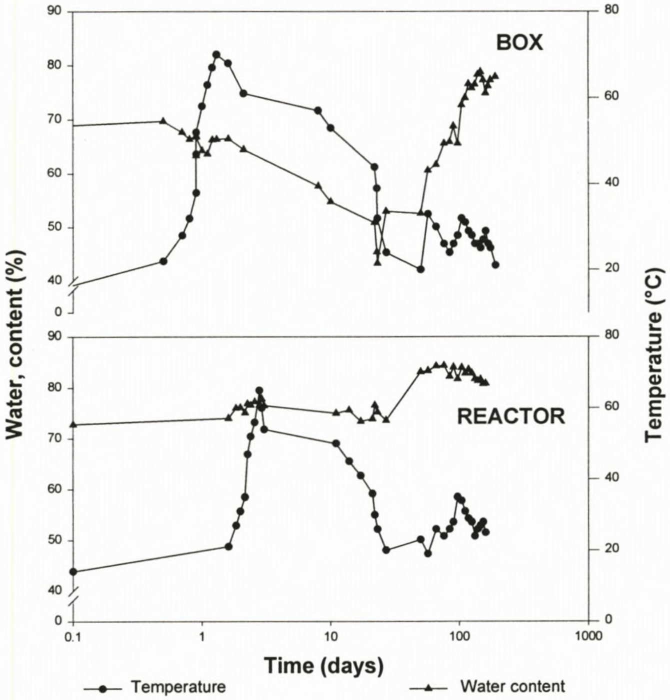

Figure 1. Changes in temperature and water content during composting of *Miscanthus* straw and pig slurry (Note: time axis is logarithmic).

Nitrogen turnover

The changes in total N and inorganic N (NH_4^+ and NO_3^-) during composting are shown in Fig. 2. Total nitrogen remained constant until the temperature had passed maximum and decreased to 50°C. The content of total N then decreased in the box from 125 to 75 mg N g^{-1} ash and in the reactor from 110 to 80 mg N g^{-1} ash. The lower level was maintained in the two systems throughout the remaining experimental period.

The content of ammonium decreased gradually in both systems from about 50 to 2 mg N g^{-1} ash as the temperature decreased to 20°C. A very low content (0.9-1.8 mg N g^{-1} ash) was found in the two systems throughout the remaining compost period.

In the beginning of any composting process, ammonia volatilization takes places due to initial heating and increasing pH (Jakobsen, 1992). Ammonia volatilization would especially be expected to proceed in a mixture of straw and pig slurry, since the content of available ammonium is high (75% of total N, Table 1). The pH values were highest around temperature maximum (8.8-9.0) in both systems. Petersen *et al.* (1998) found that the main volatilization of ammonia occurred during the first 2-3 weeks of composting in a manure heap, which had a lower C/N ratio than in the present study.

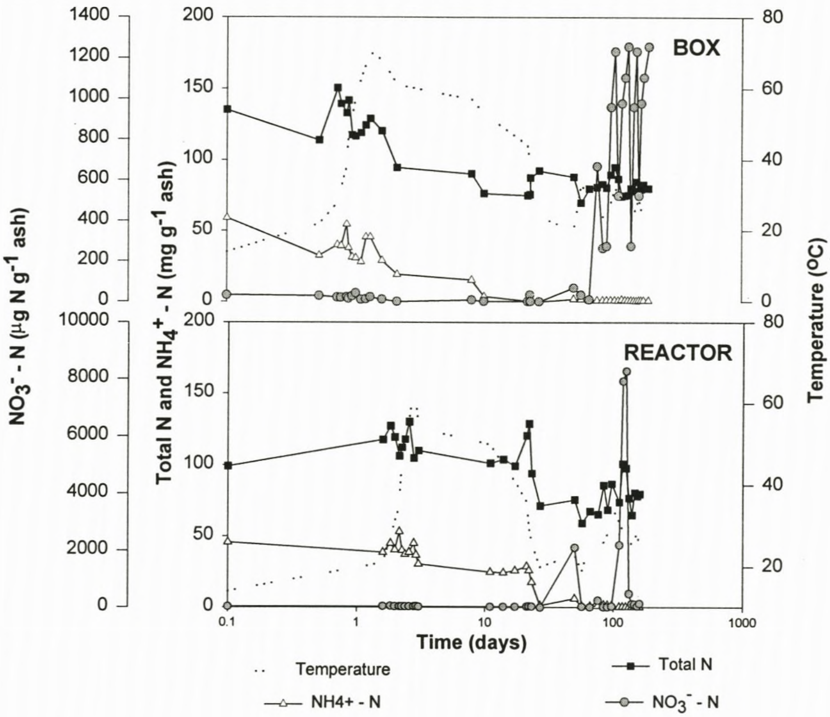

Figure 2. Changes in total N, NH_4^+-N and NO_3^--N during composting of *Miscanthus* straw and pig slurry.

Initial values of nitrate were low (30-50 µg N g^{-1} ash) and decreased during the heating phase and first part of the cooling phase (27 and 50 days in the box and the reactor, respectively). Later in the composting an accumulation of nitrate was found. In the box this accumulation (from day 120) resulted in maximum values of 1250 µg N g^{-1} ash. The high level was maintained till the end of the experiment. In the reactor similar fluctuations was found. Although an accumulation was measured, NO_3^--N did not contribute to more than 2-4 % of the total N.

The accumulation of nitrate found in both composting systems indicated favourable conditions (high oxygen tension) for nitrification. It has been suggested that the nitrate/ammonia ratio might be used as a stabilization parameter (Mathur *et al.*, 1993; Forster *et al.*, 1993).

Carbon turnover

The changes in carbon content and C/N ratio are shown in Fig. 3. The total carbon at the beginning of the compost phase was derived in part from the high molecular weight carbohydrates (the fibres) in the *Miscanthus* straw, and partly from the pig slurry added (Table 1).

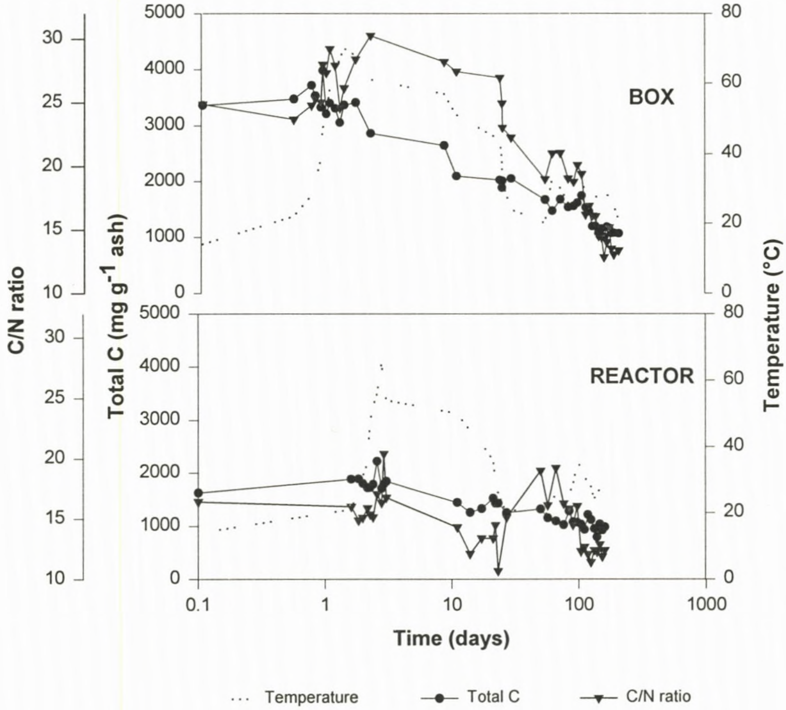

Figure 3. Changes in total C and C/N ratio during composting of *Miscanthus* straw and pig slurry.

The content of carbon was constant (box 3300 and reactor 1700 mg g^{-1} ash, average) during the first active phase (initial mesophilic phase) until the temperature maximum. When the decreasing temperature passed 50°C in the two systems, a decreased carbon content was found. Carbon content decreased gradually during the remaining composting period. An intermediary stable level was established between day 10 and 50 for the box and day 26 and 100 for the reactor. The subsequent small reheating (Fig. 1) resulted in a further decrease in the content of total C.

The differences in C and N dynamics were partly reflected in the C/N ratio. The initial C/N ratio was very different in the two systems (box 25 and reactor 16). In the box, two peaks with increasing C/N ratio (reaching a value of 30) was seen just before and after temperature maximum. In the reactor one peak (reaching 20) was seen just after temperature maximum. This reflects the very different volatilization rates for carbon (slow) and nitrogen (fast) at this stage. In the box the C/N ratio was fairly constant until day 22. During this period temperature decreased from 44 to 20°C. In the weak reheating phase from 20 to 33°C, a

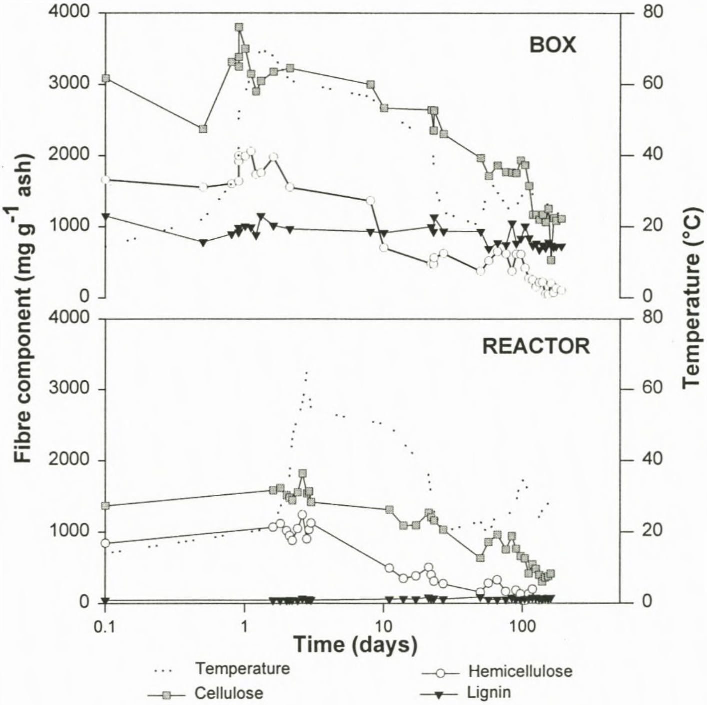

Figure 4. Changes in fibre composition during composting of *Miscanthus* straw and pig slurry.

marked decrease in C/N ratio occurred. In the reactor a temporary increase was seen just after the temperature minimum (26 days). Although the two systems had very different initial C/N values, the C/N ratios at the end of the composting were at the same level (C/N = 13).

The very different initial C/N ratio of the two systems reflected differences in the mixture of straw and slurry. During the entire composting period, 69 % of the total C and 41 % of total N were decomposed in the box system and the corresponding values for the reactor were 40 % total C and 21 % total N. These differences can be ascribed to different aeration status during the composting (highest water content in the reactor, Fig. 1). The importance of aeration has been investigated by Jeris and Regan (1973). They examined the free air space, oxygen consumption and decomposition in relation to water content. It was found that a free air space of 30-35% was optimal. This value for free air space corresponded to 60-65% water content in garbage and 50-55% water content in a mixture of garbage and sludge.

Degradation of fibres

The degree of decomposition of the high molecular weight fibre components, hemicellulose, cellulose and lignin (lignocelluloses), during the composting is important for the structure and physical properties of the end product. The changes in fibre composition are shown in

Fig. 4. In the box the content of hemicellulose remained constant during initial heating and temperature maximum. When the temperature decreased, hemicellulose also decreased (from day 2 to 8, temperature 60-55°C), and this tendency continued gradually during the remaining period of composting, resulting in only 6% of the initial content left at the end of the composting period. The same pattern was observed in the reactor.

The content of cellulose in both systems was constant until day 8 (temperature 55-60°C) and then decreased to a constant level between day 8 and 50. A further decrease in cellulose followed as the temperature decreased. The final levels of cellulose were 36% and 30% of the initial amount in the box and the reactor system, respectively.

A decomposition of lignin could only be observed in the box at the end of the process following the temperature decrease. The final content of lignin was 60% of the initial amount. In the reactor the content of lignin was constant at a very low level. Half life times for decomposition of hemicellulose were 8 days in the box and 21 days in the reactor. For cellulose, the corresponding values for the two systems were around 100 days.

Eklind (1998) referred the following half life times for different materials by composting mixtures of household waste and litter: *Straw:* Hemicellulose, 29 days and cellulose, 26 days, *leaf litter:* hemicellulose, 26 days and cellulose, 153 days. Compared to other raw materials, *Miscanthus* has a very high carbon content and a higher content of cellulose than for instance wheat straw, pine and birch wood (Table 2). Therefore other ratios for mixing straw and slurry might influence the physical properties of the compost product in relation to degradation of cellulose and lignin.

Table 2. Composition of fibres from different plant materials.

Plant material	% hemicellulose	% cellulose	% lignin
Wheat straw	44	36	14
Barley straw	27	39	10
Corn Cobs	42	39	5
Sugar cane	30	34	19
Pine wood	44	26	28
Birch wood	40	39	20
Sphagnum	4	18	28

After Lynch, 1993; Theander & Åman, 1984

Conclusions

1. The mixture of shredded *Miscanthus* straw and pig slurry was shown to give very favourable conditions for composting, achieving high temperature maxima.
2. Composting in the box system resulted in a higher degradation of nitrogen as well as carbon compared to a reactor system, and the final C/N ratios were similar (C/N=13) at the end of the composting period. In the box system 69 % of initial total C and 41 % of initial total N was decomposed. The corresponding values for the reactor were 40 % total C and 21 % total N.

3. The major loss of carbon occurred after the heating phase, and mainly originated from decomposition of hemicellulose and, to a lesser extent, cellulose. The degradation of hemicellulose was similar in the two systems (only 6 % of initial left) while significantly more cellulose was decomposed in the box (40 % left) as compared to the reactor (64 % left).
4. The initial C/N ratio is, together with water content and aeration during the composting process, a key parameter determining the physical structure of the final product.

Acknowledgements

H.E. Kresten Jensen, Danish Institute of Agricultural Sciences, Department of Ornamentals, Årslev, has kindly provided facilities for the composting experiments. We thank Erling Hyldig for skillful technical assistance with the compost handling. The project is part of a co-operative research program entitled " Biological Materials and Products" (1996-99) financed by The Danish National Research Council.

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J. Petersen & S.O. Petersen (eds.) 1999. Use of municipal organic waste
Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
Session 4
Danish Institute of Agricultural Sciences report no. 13

Composting of sewage sludge and biowaste in a tunnel composting system

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Sewage sludge has been co-composted with biowaste (source separated household biowaste) and wood chips since 1996 using the Vapo Biotech tunnel system in the composting plant of the city Kitee. The plant processes about 900 wet tonnes of sewage sludge, 400 wet tonnes biowaste and 800 tonnes wood chips annually. Since May 1998 wood chips were partly substituted by recycled compost (sieved, over 20 mm recycled).

The composting process is divided into three phases: a) about 14-21 days active tunnel composting with forced aeration and temperature control, b) a curing phase (two to six months) using non-aerated piles and c) screening (under 20 mm), and final curing for several months. The total mass reduction during the active tunnel composting was on average 24% in 1997.

The composting was studied during summer of 1998 in an attempt to better understand the controls of the composting process and to produce an end product with higher quality. In the first experimental series the process control based on temperature development was done as previously, but during the second and the third series the process control and the temperature limits used for the control were varied.The experiments were done in duplicate and the process was monitored for temperature, airflow, moisture content, weight loss, ash content, pH, EC, C/N ratio, microbial activity, hygienization, soluble nitrogen forms, dissolved organic carbon (DOC), and during curing also for phytotoxity.

Results are being analysed and will be presented at the seminar. They will be used to discuss parameters for compost process control, and to discuss problems in tunnel composting of biosolids.

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Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
Session 5
Danish Institute of Agricultural Sciences report no. 13

Plant nutrients in human urine and food refuse

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Summary

Recirculation of nutrients from urban areas to agricultural land is one of the big challenges of our time. In this paper human urine and effluents from digestion of food refuse are discussed and results from field trials with these fertilisers are presented.

Results from 1997 in ongoing field trials in the vicinity of Stockholm investigating the fertiliser effect of stored human urine, effluents from digestion of food refuse and mineral fertilisers applied to barley are reported in this paper. Nitrogen efficiency and influence on grain yield was studied. The risk of nitrate leaching was estimated from nitrogen balances and measurements of soil nitrate nitrogen content in late autumn.

An application of human urine containing 100 kg per hectare of total nitrogen (N) yielded 68 % of the harvest in plots fertilised with the same amount of N in mineral fertilisers. A spring application of effluents from digestion of food refuse yielded 82% of the harvest in plots fertilised with the same amount of plant available N in mineral fertilisers. An application of effluents from digestion of food refuse in the beginning of July yielded 52 % of the harvest in the plots fertilised with mineral fertiliser at the same levels, but applied at the time of sowing.

The risk of nitrate leakage was equally high from organic and mineral fertilisers in these trials. A nutrient balance of N added and N removed from the fields by the crop verifies that large inputs increase residual N in the soil.

These are results from ongoing field trials. The trials are repeated during several years. Thereafter, final conclusions will be drawn that are valid for different weather conditions.

Introduction

Swedish society strives towards sustainable living with integrated solutions of plant nutrient flows between consumers and producers of food. Clean food refuse from restaurants and households are potential sources of energy and plant nutrients. The new challenge is to find hygienic and environmentally sound solutions for recycling plant nutrients in food back to agricultural land. At the same time the strategy is to improve air and water quality, as well as recycling nutrient resources back to agricultural land as organic manure.

Swedish agriculture annually produces animals, milk, eggs, vegetables and grain containing 65 000 tonnes of N and 11 000 tonnes of phosphorus, out of which 20 % is lost in the food processing industry (Claesson & Steineck 1996; Löfgren *et al.*, 1998; Jakobsson *et al.*, 1998). Plant nutrients in the food that is consumed pass through the human body and end up in the toilet system, since very little of these substances are used to build new biomass in the body of a grown person. The plant nutrients are collected in waste water treatment plants and a large part pollutes the environment, depending on the system used.

Today many private households lack waste water purification systems or have deficient systems. Aaltonen & Andersson (1995) showed that sand filter beds used in single households in Sweden have deficient purification efficiency. After 13 years the sand filter has become a source, instead of a sink, for nutrients.

Separating urine from faeces directly in the toilet is one of many solutions for recycling nutrients without unwanted pollutants. There is an interest from organic farmers in Sweden to use human urine as liquid fertiliser because of the content of plant available N, phosphorus and the low heavy metal content (Lindén, 1997). Effluents from digestion of food refuse is another example of an organic fertiliser rich in plant nutrients. During the digestion, energy is expelled in methane gas. Residues are suitable as fertilisers for crops, since plant nutrients are retained in the residual effluents in forms easily accessible to the plant.

There is a need for research on efficiency and environmental impact of these new organic fertilisers. In a comparison between humans and pigs, humans excrete a larger proportion of their nitrogen and phosphorus intake in the urine. Most of the N in human urine is in a plant available form as ammonia N (Kirchmann & Pettersson, 1995; Claesson & Steineck, 1996).

Stored human urine normally has a higher pH, 8.6-9.2, than animal urine with a pH value of 8.4-8.8. The high pH in human urine may have a positive effect in killing infectious bacteria and virus (Höglund *et al.*, 1997). However, a high pH increases the risk for ammonia losses during storage and after spreading. Ammonia emissions are both a resource problem and an environmental problem (Löfgren *et al.*, 1998).

Effluents from digestion has a high pH, 8-9, as well. The material for digestion is pasteurised at 70°C for one hour before treatment in a digestion chamber. The content of nutrients in effluents from digestion depends on the material and the process. In most cases, the nutrient concentrations are similar to those of animal slurry.

Objectives

The objective of the field trial was to determine the effect of application rate, application time and application technique on grain yield and nitrogen utilisation after spreading human urine and effluents from digestion of food refuse. The organic fertilisers studied were compared to mineral fertiliser. Another objective was to estimate the risk of nitrate leakage based on N in the soil at harvest.

Materials and methods

Fertiliser value of human urine and effluents from digestion of food refuse was studied in a field trial with barley in 1997. The field was located south of Stockholm. The soil is a clay loam with 2.8 % soil organic matter (SOM) and pH 6.6. The experimental design was randomised blocks with three replicates. Human urine and effluents from digestion of food refuse were applied before sowing with a plot spreader with trailing hoses, 0.25 m apart. Incorporation into the soil with a light harrow was performed four hours after spreading. Human urine was band spread to barley at four different rates corresponding to 10, 20, 30 and 60 tonnes per hectare. A control treatment with no fertiliser and treatments with mineral fertiliser, NPK, were included at rates of 0, 30, 60, 90 and 120 kg N per hectare. Applications of effluents from digestion of food refuse were performed at 22 tonnes per hectare in the spring and 28 tonnes per hectare in the summer when the barley was 10 cm high.

Barley was harvested at maturation, and the weight was registered. Grain was analysed for volume weight, content of dry matter and total N. Sampling and analysis of the soil was made in the spring before sowing and late in the autumn to determine N mineralisation and estimate the risk of N leakage.

The stored human urine originated from people living in "ecological villages" in Stockholm where toilets have been installed separating urine from faeces (Jönsson *et al.*, 1997). The effluents from digestion of food refuse came from a project "From the dinner table to the soil", recycling food refuse in Stockholm. Samples of urine and effluents from digestion were taken at the time of spreading for analyses of dry matter (DM), pH, NH_4-N, total N, phosphorus, potassium and ash content (see Table 1a).

Table 1. Content of dry matter (DM), pH, NH_4-N, total N, phosphorus and potassium in kg per tonne in human urine and effluents from digestion of food refuse in 1997.

	DM	pH	Nutrient content, kg/tonne			
	%		NH_4-N	Tot-N	P	K
Human urine	0,74	8,9	3,4	3,7	0,3	1,0
Effluents from digestion, spring appl.	1,8	8,6	2,2	3,6	0,19	0,99
Effluents from digestion, summer appl.	2,3	8,0	2,3	3,5	0,20	0,8

Results and discussion

An application of human urine in 1997 corresponding to 100 kg per hectare of total N yielded 2 600 kg barley per hectare, see figure 1. The barley harvest was 32 % less than when the same amount of N in mineral fertilisers was applied. The yield response curves with the amount of N as independent variable were significantly different for human urine compared to mineral fertiliser in 1997. A spring application of effluents from digestion of food refuse containing 48 kg NH_4-N per hectare yielded 2 210 kg barley. Compared to the same amount of N in mineral fertilisers, the yield was 18 % lower. In the beginning of July, when the crop was 10 cm high, an application of effluents from digestion of food refuse with 66 kg NH_4-N per hectare yielded 1 660 kg barley per hectare. This yield was 48 % lower than for the same amount of N applied in mineral fertilisers, applied at sowing.

An application of 100 kg N per hectare in mineral fertilisers 1997 yielded 3 780 kg grain per hectare. This is lower than normal due to the weather and also to the fact that the field was sown two weeks after normal sowing time. Excessive rain in April and May resulted in a slow drying process of the soil. June was wet and July was extremely dry and warm (Elmquist *et al.*, 1998). The normal reference yield of barley for this area is 4 510 kg per hectare (SCB, 1997a). Although barley yields in the trial of 1997 were lower than normal due to the delayed sowing time, differences in nitrogen response of the yields were still quite evident. No toxic effects from human urine were visible in the crops in the field trial in 1997 as well as in pot experiments using the same human urine (Kvarnmo, 1998).

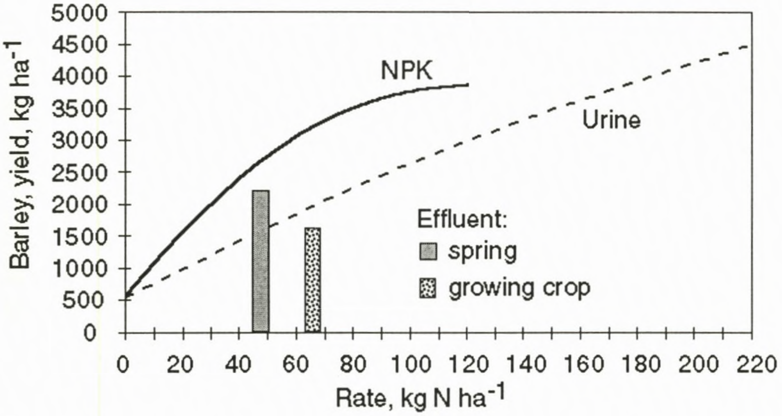

Figure 1. Yield of barley with 15% water content after application of human urine, mineral fertiliser and effluents from digestion of food refuse. The bars show yield response for effluents applied at two different times in 1997.

A nitrogen balance is a useful tool to evaluate the efficiency of N when using different kinds of fertilisers and amounts of N (SCB, 1997b). The net mineralisation of soil N during the season of 1997 was calculated from the control treatment to 35 kg N per hectare (see Table 2). This was calculated from N in the grain, root and straw of barley just before harvest, plus mineral N in the soil at spring, minus mineral N in the soil before harvest.

A nutrient balance of added N and N removed from the field by the crop verifies that large inputs increase residual N. This nitrogen was not found in the soil as mineral N in the autumn. According to measurements of the content of mineral N in the soil (0-90 cm) before freezing in the autumn (Table 2), the risk of nitrate leaching after application of stored human urine and effluents from digestion is just as high as from mineral fertilisers.

Results from one year of field trials are insufficient to draw final conclusions from. The trials are repeated during at least two years. Thereafter final conclusions concerning plant nutrient efficiency and N losses will be drawn.

Table 2. Nitrogen balance indicating input and output of N in the field trial in 1997 (total N in kg per hectare).

	Application rates of mineral N kg/ha	Application rates of total N kg/ha	Net mineralisation of soil N kg/ha	Import of N via deposition (SJV, 1997) kg/ha	Export of N in barley yield kg/ha	Calculated rest of N in soil kg/ha	Measured rest of N in the soil 0-90 cm (autumn) kg/ha
Control		0	35	10	12	33	35
Human urine	26	28	35	10	13	60	34
Human urine	69	74	35	10	24	95	35
Human urine	98	105	35	10	40	110	35
Human urine	212	230	35	10	71	204	37
NPK	30	30	35	10	29	46	33
NPK	60	60	35	10	39	66	34
NPK	90	90	35	10	55	80	31
NPK	120	120	35	10	61	104	33
Effluents spring appl.	48	79	35	10	30	94	37
Effluents summer appl.	66	101	35	10	26	120	45

Conclusions

- Stored human urine and effluents from digestion of food refuse can be used as fertilisers in grain production.
- An application of human urine containing 100 kg per hectare of total N yielded 2 600 kg grain of barley per hectare in 1997, amounting to 68 % of the yield from the plot with equal amounts of N in mineral fertiliser.
- An application of 48 kg NH_4-N in effluents from digestion of food refuse, applied in the spring of 1997 yielded 82 % of the yield at the same rate of N in mineral fertilisers. A summer application of 66 kg NH_4-N in effluents with no spring dressing yielded 52 % of the harvest obtained with the same rate of N in mineral fertilisers applied in spring.
- Application of effluents from digestion of food refuse at sowing time in spring yielded a higher harvest than application later in the summer in 1997.
- The nutrient balance of added N and N removed from the field by the crop verifies that large inputs increased residual N in 1997. This N was not found in soil samples at harvest

Acknowledgements

The field trials are financed by Stockholm Vatten AB, HSB Riksförbund, AB Stockholmshem, Stockholms läns landstingsfond, SKAFAB/SRV and Stiftelsen Lantbruksforskning.

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Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
Session 5
Danish Institute of Agricultural Sciences report no. 13

Perennial reed canary grass fertilized with sewage sludge

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Introduction

During the 1970's about 50% of the sewage sludge produced during waste water treatment in Finland was used in agriculture (Levinen, 1990). The use of sludge in agriculture subsequently decreased, with the increased awareness of the risk of heavy metals to health. The quality of sludge has improved in recent years through the elimination of a large part of the harmful chemicals. About 39% of sludge was used in agriculture in 1997 (Syke, 1998).

Sludge contains in organic form several minerals essential to plants. Moreover, both chemical and physical characteristics of the soil are improved by the addition of sludge. To some extent, nevertheless, the health risk remains, making the application of sludge to non-food crops an attractive alternative. High yielding crops including hemp, miscanthus and reed canary grass are highly promising sources of biomass for paper, fibre and energy use in Europe. At the same time there is a great need to reduce greenhouse gas emissions, and agriculture could make a contribution in this area through increased production of bioenergy crops to replace nonrenewable fuels.

A promising biomass crop for industrial use being investigated in both Finland and Sweden is reed canary grass (Pahkala *et al.*, 1994, Landström *et al.*, 1996). Reed canary grass is of interest as a perennial grass that can be cultivated for as long as 10 years without reestablishment. With its short fibre quality, reed canary grass could replace birch in the production of fine paper (Paavilainen *et al.*, 1997). Combustion properties, in turn, are adequate for its mixed combustion with wood chips or peat in power plants (Leinonen *et al.*, 1998).

Large-scale biomass production can only become competitive if costs are reduced in all steps of the production chain: cultivation, harvesting, transport and end use. One way to reduce cultivation costs, and at the same time dispose of a problem waste, would be to fertilize crops with sewage sludge. Before sludge could be applied in this way, however, we would need to know about its value as a nutrient source and the effect on the environment. The primary aims of this study were to determine the availability of nutrients (N, P) of sewage sludge to perennial reed canary grass, and to determine the fate of the heavy metals cadmium (Cd) and lead (Pb).

Materials and methods

Plant and soils

Reed canary grass (*Phalaris arundinacea* L. cv. Palaton) was used in both pot and field experiments.

The soil for the pot experiment was taken from the field at Jokioinen in spring 1995. The moist soil (13.8%) was homogenized before the experiment. The soil type was fine sand (15.9% clay, 5.1 silt, 79% sand, 1.8% org. C, pH(H_2O) 5.88, Ca 1072, K 214, P 12.3, Mg 164, Cd 0.06). Pb 1.78. The soil type at the field experiment was fine sand (18% clay, 9.9% silt, 72.1% sand, 2.6% org. C, pH(H_2O) 6.33, Ca 2027, K 499, P 100, Mg 201, Cd 0.16, Pb 2.40). The methods of soil determination have been described previously (Partala *et al.*, 1997). Extractable contents of nutrients particularly the extractable P content, were higher in the soil used for the field experiment than soil used in the pot experiment. According to fertility tests, the soil for the pot experiment was classified mainly as satisfactory, and the soil for the field experiment mainly as good. P was classified as possibly excessive. Concentrations of Cd and Pb were lower in the soil used for the pot experiment, but in both soils they were within a range normal for Finnish agricultural soil.

Quality of sewage sludge

The sewage sludge used for the pot experiment (SS 1) was from the Helsinki municipal treatment plant. Sludges used for the field experiment were from the Forssa treatment plant, with different batches applied in 1995 (SS 2) and in 1996 (SS 3). Dry matter contents were 26.6%, 15.5% and 15.4%, and organic C content 24.8%, 32.1% and 30.0% in sludges SS 1, SS 2 and SS 3, respectively.

Table 1. Concentrations of total and soluble nutrients and heavy metals in sewage sludge used for the pot experiment (SS 1) and the field experiment (SS 2 and SS 3).

Element	Total			soluble mg kg^{-1} DM			Limit value
	SS 1	SS 2	SS 3	SS 1	SS 2	SS 3	mg kg^{-1} DM
N %	2.79	3.9	3.1	6230	4270	4960	
P %	3.39	1.6	1.5	136	54	68	
K mg kg^{-1} DM	3920	3730	4500	484	1090	1094	
Ca mg kg^{-1} DM	22200	14900	14500	12258	8669	8584	
Cu mg kg^{-1} DM	472	276	31	36	75	600*	
Zn mg kg^{-1} DM	860	610	560	256	240	235	1500*
Cd mg kg^{-1} DM	1.7	0.44	0.69	0.15	0.02	0.08	3.0
Pb mg kg^{-1} DM	103.1	29.2	13.9	2.06	1.23	5.14	150

* Limit value can be doubled if the soil is poor in that element.

Concentrations of total Cd in sludge used for the pot experiment was two to four times those in sludge used for the field experiment (Table 1). Despite this, all concentrations were below the Finnish regulation limits for heavy metals in sewage sludges to be used in agriculture (Council of State decision, 1994).

Pot experiment

Plants of reed canary grass were grown in eight-litre pots (Kick-Braumann), each filled with six litres of moist fine sandy soil. Pots were placed outside, under a transparent cover without walls. The treatments were:

1. Control, no fertilization
2. Mineral fertilizer (N, K, 1000 mg pot^{-1}; P, 400 mg pot^{-1}; Mg, 200 mg pot^{-1}; Fe 100 mg pot^{-1})
3. Fresh sewage sludge 600 ml pot^{-1} = 47 t DM ha^{-1} (SS 1 +)
4. Fresh sewage sludge 1800 ml pot^{-1}= 141 t DM ha^{-1} (SS1 +++)

For each treatment there were four replicates. Mineral fertilizers were given as NH_4NO_3, KNO_3, $KHPO_4$, $MgSO_4$ and $FeSO_4$. During the experiment plants were irrigated with deionized water. The first harvest was in November 1995 after the growing season. The dead shoot and root material was harvested and weighed, and the content of nutrients and heavy metals was determined in above and below ground plant material. Soil samples were taken and extractable nutrient and heavy metal contents were determined. A second harvest was made in April of the following year, before the growing season. Measurements on shoot and root material were the same as those made in November.

Field experiment

The sowing date was 3 July 1995. The first sludge amendment took place four days before sowing, and the second amendment on 15 May 1996. Mineral fertilizer was applied in May 1996, after the harvest. The treatments were:

1. Control, no fertilization
2. Mineral fertilizer (in 1995, 70 N-14 P-28 K; from 1996 to 1998, 100 N- 20 P-40 K)
3. Sewage sludge 8.7 t DM ha^{-1} applied before sowing (SS 2 +)
4. Sewage sludge 26.2 t DM ha^{-1} applied before sowing (SS 2+++)
5. Sewage sludge 8.4 t DM ha^{-1} applied following spring (SS 3 +)
6. Sewage sludge 25.2 t DM ha^{-1} applied following spring (SS 3+++)

Plot size was 15 m^2 (1.5m x 10m) and there were four replicates for every treatment. Shoot and root samples were taken each year in July, October and May. The plots were harvested in May. Soil samples were taken in October and in May for NH_4 and NO_3 measurements, and in May for measurements of extractable P, K, Ca and Mg.

Sewage sludge amendment

In both pot and field experiments the higher rate of sludge amendment was three times the lower rate. The rate of sludge amendment was five to six times greater in the pot than in the field experiment. Moreover, the higher P, Cd and Pb concentrations in the sludge used for the pot experiment meant that about 10 times greater amounts of those elements (calculated per hectare) were added in the pot experiment than in the field experiment (Table 2). In the field experiment, the limit values for concentrations of heavy metals were slightly exceeded at the higher rate of sludge amendment, while the limit value for P was exceeded even at the lower rate of addition. In the pot experiment, even at the lower rate of addition, a single application of sludge added P, Pb and Cd to the soil in amounts equivalent to limit values over 80, 32 and 27 years, respectively.

Table 2. Total nutrient (N; P kg ha^{-1}) and heavy metal (Cd; Pb g ha^{-1}) additions in sewage sludge and limit values (g ha^{-1} y^{-1}).

	Sewage sludge addition						Value
	Pot experiment		Field experiment				
Element	SS1+	SS1+++	SS2+	SS2+++	SS3+	SS3+++	g ha^{-1}
N	1313	3939	342	1025	256	769	
P	1595	4786	140	420	126	378	15*
Cd	80	240	3.8	11.5	5.8	17.4	3.0
Pb	4850	14550	260	770	120	360	150

* Finnish agro-environmental programme, 1997.

Results

Yield

In the field experiment the reed canary grass was harvested in spring (delayed harvesting). Yields were high for all treatments, and even without fertilization the yield was high, about 8800 kg DM ha^{-1} in 1998 (Figure 1). The yield was only 14% higher with mineral fertilizer than without fertilization.

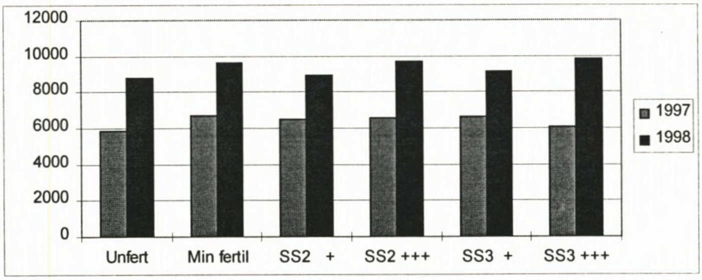

Figure 1. Dry matter yields (kg DM ha^{-1}) in field experiment harvested in May.

In the pot experiment, mineral fertilizer and sewage sludge addition increased the yield in proportion to the rate of nutrient application (Table 3). Root biomass decreased at the same time as the sludge amendment increased, however.

Table 3. Dry matter yields of shoots and roots in pot experiment.

Treatment	Autumn 1995, yield g pot^{-1}		Ratio (%)	Spring 1996, yield g pot^{-1}		Ratio (%)
	Shoot	Root	shoot/root	Shoot	Root	shoot/root
Unfertilized	4.3	6.8	39	4.0	5.1	44
Min fertilizer	55.6	38.6	59	59.2	33.3	64
SS1+++	77.3	44.4	64	90.2	36.6	71
SS1+	58.8	53.8	52	68.6	40.6	63

Yield quality

Sludge amendment clearly increased the N concentration of reed canary grass and, at the highest rate of addition, also the P concentration was increased (Tables 4 and 5). While the concentration of N was at the same level in the two experiments in both autumn and spring (0.8-1.7%), the concentration of P was higher in the field experiment in autumn. Compared to the pot experiment. The field experiment had lower N and P concentrations in the harvested shoots in spring. This partly due to mineral leaching and partly to the decreased ratio of leaves to straw. Leaf-straw ratio effects because leaves contains more minerals than straw (Pahkala *et al.*, 1999). In the pot experiment the roof prevented rain and snow from falling on the pots and leaching of minerals from the shoots.

Concentrations of Cd were about twice as high in the pot experiment as it was in the field experiment (150 μg kg^{-1} DM), but Pb concentration (180 μg kg^{-1} DM) was only about half that in the field experiment. In both experiments Cd concentration increased with the rate of sludge application, but only in the pot experiment did the Pb concentration increase with application rate. The concentrations of Cd and Pb were two- to six-fold higher in roots than in shoots.

Table 4. Mineral concentrations in dry matter of shoots of reed canary grass in field experiment.

Treatment	July-96				Oct-96				May-97		
	N	P	Cd	Pb	N	P	Cd	Pb	N	P	Pb
	%	g kg^{-1}	μg kg^{-1}		%	g kg^{-1}	μg kg^{-1}		%	g kg^{-1}	μg kg^{-1}
Unfertil	1.25	2.7	54	468	0.86	1.9	49	1054	0.80	0.9	1293
Min fer	1.97	3.3	56	481	1.16	2.3	72	1108	1.00	1.0	1285
SS2+	1.69	2.9	53	577	1.09	1.8	68	681	1.03	1.0	1150
SS2+++	2.14	3.2	76	600	1.46	2.2	46	732	1.24	1.1	1101
SS3+	2.16	3.6	51	507	1.21	2.0	49	914	1.15	1.0	1512
SS3+++	2.76	4.0	73	551	1.64	2.4	45	817	1.52	1.3	1265

Table 5. Mineral concentrations in dry matter of shoots and roots of reed canary grass in pot experiment.

Treatment	Autumn 1995				Spring 1996			
	N	P	Cd	Pb	N	P	Cd	Pb
	%	g kg^{-1}	µg kg^{-1}		%	g kg^{-1}	µg kg^{-1}	
Shoots								
Unfertilized	0.8	3.1	30	500	1.1	3.4	120	1175
Min fer	0.7	1.0	134	175	0.8	1.4	131	672
SS1+	0.8	1.2	120	185	0.9	1.3	176	703
SS1+++	1.6	1.6	187	188	1.7	1.8	199	690
Roots								
Unfertilized	0.8	1.6	1280	1340	0.9	1.3	255	2052
Min fer	0.8	1.9	1020	620	0.9	0.8	348	1343
SS1+	0.9	1.6	950	700	1.2	1.4	475	1257
SS1+++	1.9	2.0	810	840	1.7	1.6	751	1723

Uptake of N, P, Cd and Pb

In the field experiment the harvested yield contained only a small part of the N, P and Pb added sewage sludge (Table 6). If the yields and concentrations of elements were to constant, it would take 50 years before the amounts of P and Pb added in sewage sludge at the higher rate of sludge application would be removed. Similarly, the N application at the higher rate corresponded to the crop uptake in a 10-year period. In the pot experiment, the time required for Pb uptake to equal the amount applied would be 700 years, and for Cd uptake the time would be 40 years.

Table 6. Mineral by the crop in the field experiment.

	N	P	Pb	N	P	Pb
Treatment	kg ha^{-1}		g ha^{-1}	% of added		
Unfertil	47.6	5.5	7.6			
Min fer	67.3	6.7	8.6	67.3	33.4	
SS2+	67.1	6.6	7.4	19.6	4.7	2.89
SS2+++	81.5	7.2	7.2	8.0	1.7	0.94
SS3+	76.2	6.7	10.0	29.8	5.3	8.53
SS3+++	91.8	7.9	7.6	11.9	2.1	2.17

Soil mineral nitrogen and soluble phosphorus, cadmium and lead

Soil mineral nitrogen (NH_4-N and NO_3-N) was measured in the field experiment in spring and autumn. In spring 1996 the mineral N in the top soil (0-20 cm) was as high as 123 kg ha^{-1} with the higher rate of sewage sludge amendment (SS2+++). At the same time, mineral N in plots treated with mineral fertilizer was only 24 kg ha^{-1}. The higher amendment of SS (SS3+++) in spring 1996 increased the mineral N in autumn 1996 to 105 kg ha^{-1}. With the mineral fertilizer, soil mineral N at the same time was only 13 kg ha^{-1}.

In the pot experiment the mineral N concentrations in soil without fertilization, with liquid fertiliser, in SS1+ and in SS1+++ were 1.4, 1.5, 3.7 and 12.6 mg kg^{-1} dry soil, respectively.

In the field experiment the soluble P in soil was as high as 100 mg l^{-1} at the beginning of the experiment in 1995. After three years cultivation of reed canary grass the level had decreased to 71-77 mg P l^{-1}. There were no differences in extractable P concentrations between sludge application, mineral fertilizer application and the control.

In the field experiment, extractable Cd concentrations did not differ between sludge and mineral fertilized plots in 1998. Extractable Pb was slightly lower (7%) with the high rate of sludge addition than with other treatments. In the pot experiment, application of sludge had no effect on the soil Cd concentration, but increased the Pb concentration dramatically: at the lower rate of application the Pb increased by 46% and at the higher rate by 83%.

Conclusions

The high fertility of the field soil suppressed differences in the yields obtained with sludge, mineral fertilizer and without fertilization. The economic value of the sludge added to the field was low: an increase in yield of 1 tonne DM per ha increased the value of the yield by 200-400 Finnmarks (about $ 40-80). This value was calculated from to the price industry is supposed to pay for biomass. Since levels of soluble nutrients are lower in normal agricultural soil, the net gain in yield due to sludge application could much higher in a normal situation than in this experiment, as indicated by the results from the pot experiment.

While the contents of Cd and Pb in the sludge applied to the field were within the legal limit, the P content was too high. The Finnish limits for Cd and Pb would seem to be quite safe; for the plant Cd and Pb and extractable soil Cd and Pb concentrations did not increase remarkably. Because of the high concentration of Pb, the application in the pot experiment would be legal only if averaged over 97 years. Despite this, the Pb concentration in the plants was not higher with sludge than with mineral fertilization. The Cd content in harvested shoots, in contrast, with the rate of sludge application. Evidently Cd content increased in the crop, although an increase extractable Cd in the soil could not be measured. Despite the higher extractable Cd content in the field soil, the harvested yield contained less Cd than that in the pot experiment. The difference may depend on the higher soil pH in the field experiment.

The low levels of heavy metals in Finnish sludge are a real advantage in the utilization of sludges for non-food crops. In fact, the content of heavy metals is not the restricting factor for higher addition of sludge to agricultural land. Usually it is the high content of P that restricts the addition to low levels. According to Finnish legislation, for environmental reasons the maximum addition of P in sludge is limited to 80 kg ha^{-1} every fourth year. This value would be exceeded with as little as 30 t fresh sewage sludge addition (3% P) per ha.

The experiments demonstrate a good ability of reed canary grass to utilize nutrients in sewage sludge for growth. At legal levels of amendment, sludge did not increase the concentration of Cd or Pb in neither the harvested material nor the soil. The results apply to a single addition of sewage sludge; the situation after long-term application was not studied.

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Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
Session 6
Danish Institute of Agricultural Sciences report no. 13

Phosphorus recycling from waste water by filter media used as fertilisers

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Abstract

Phosphorus (P) is the most important resource in sewage water, but is only to a limited part recycled. Often it is not even separated, like in wastewater from many Swedish countryside households. Hence, large amounts of P reach different water bodies causing eutrophication. Ecologically engineered wastewater treatment facilities, such as constructed wetlands and infiltration plants are attractive for removing pollutants also in scattered settling areas. These facilities can be designed to further improve the P retention by using reactive media which possess high capacity to retain P. In order to recycle the P in plant production, the sorbed P must be plant available. Reactive media of natural as well as of anthropogenic origin, were tested with regard to their P retention capacity in batch experiments, as well as in column experiments in the laboratory. Physical and chemical properties of the materials were analyzed to estimate the usage of the reactive media in wastewater treatment plants. Chemical extractions were performed with a method used for estimating plant available P in order to obtain an indication of the plant availability. The plant uptake was then confirmed in a pot experiment with barley.

Crystalline blast furnace slag in two particle sizes, amorphous slag, opoka (natural and burned), limestone, burned lime, the soil from a spodic B horizon, and Leca (light expanded clay aggregates) were compared. In the P saturation experiment, the materials got exposed to P solutions of different P concentrations; 2, 10 and 20 mg $P*L^{-1}$, respectively. In the following P extraction experiment, ammonium lactate was used as an extractant. The results showed that the crystalline slag and burned opoka sorbed P most efficiently, the sorption exceeded 95% independent of P concentration. In the extraction experiment, nearly all sorbed P in the burned opoka was extracted with an ammonium lactate solution, while only small amounts were extracted from crystalline slag. However, a pot experiment with barley indicated superior P uptake from the crystalline slag. The differences might partly depend on the Mg content in the slag and Mn deficiency after application of opoka due to increased pH. It was concluded that the burned opoka and crystalline slag were the most suitable of the investigated filter materials from an agricultural point of view, since they possessed high P sorption capacities

and the sorbed P was highly plant available. Further experiments are needed to investigate the large divergence in plant availability between chemical analyses and pot experiments.

Introduction

Phosphorus in the wastewater from households in the countryside is an important source of pollution in many countries (Swedish Environmental Protection Agency and Ministry of Foreign Affairs, 1998; Vymazal *et al.*, 1998). If at all treated, the wastewater generally passes an infiltration bed of sand with a poor P-sorbing capacity (Swedish Environmental Protection Agency, 1995). Investment in technologically advanced wastewater treatment plants of the same type as in municipalities are not economically feasible (Swedish Environmental Protection Agency, 1995).

An infiltration bed can efficiently trap P, if material with a high P sorption capacity is used (Jenssen *et al.*, 1991; Johansson *et al.*, 1996). The material should be easy to replace when it is saturated with P. Phosphorus sorbed from wastewater should be recycled in agriculture, since P is an important plant nutrient with globally limited resources. Hence, P must be plant available when the P saturated sorption materials are applied to the fields.

The objective of this study was to determine the plant availability of P sorbed to eight different materials compared to P added as potassium phosphate. The plant availability was evaluated from yield response and P content of barley and compared with a P fraction (P-AL) used as an indicator of plant available P.

Materials and methods

The experiment consisted of two parts: I) estimation of plant available P by chemical extraction, and II) determination of P available to barley plants. Estimation of plant available P was done by extracting the studied sorption materials with an acid ammonium-lactate (AL) solution (pH 3.75, 0.40 M acetic acid, 0.10 M ammonium lactate) according to the Swedish Standard Procedure (Egnér *et al.*, 1960; SIS, 1993). Fifty ml AL solution was added to 2.5 g dry material and continually shaken in an orbital shaker for 90 minutes. If necessary, the samples were centrifuged and then filtered through a 0.45 μm Satorius filter. The supernatant was analysed for PO_4^{3-}-P according to the Swedish Standard Procedure (Murphy & Riley, 1962; SIS, 1984) by Flow Injection Analysis (autoanalyser Aquatec-Tecator). Prior to the AL-extraction the sorption materials (2.5 g in triplicate) had been equilibrated with 50 mL of P solutions containing different amounts of P (2, 10 and 20 mg/l) at room temperature for 24 h. After centrifugation and filtration, the solid residues were dried entirely before the extraction.

Determination of P available to barley plants was done in a pot experiment. Mitschellich pots were filled (4.25 kg DM pot^{-1}) with the A-horizon (5-20 cm) of a P depleted agricultural soil under permanent lay from Bjärröd, Scania, Sweden (55° 42' N, 13° 43' O, alt. 105 m). The soil was a sandy moraine with 14% clay, pH_{H2O} 6.3, and 6 and 50 mg kg^{-1} DM soil of AL extractable P and K, respectively (Egnér *et al.*, 1960).

Each pot received 1.5 g K_2SO_4, 0.5 g $MgSO_4$, and 1.0 g N as NH_4NO_3 as basic fertilisation. The soil was mixed with P in the form of K_2HPO_4, either separately or sorbed to different materials in quantities corresponding to 0.03 or 0.3 g P pot^{-1}, each treatment in triplicate. Three of the sorption materials were taken from a column experiment slag materials, which had received applications of 10 ppm P-solution every second hour for 13 months. The other materials were soaked in 1000 mg P L^{-1} solution and intermittently shaken for 24 h (material:solution 1:4 for CaO, other materials 1:2), dried (<35°C), rinsed with distilled water (material:solution 1:2) and dried again. The materials were (Table 1): crystalline slag 0-0.125 mm (CSF); crystalline slag 0.25-4 mm (CSC; CSC2=fresher and coarser slag, <7 mm, than CSC); amorphous slag 0.25-4 mm (ASC); limestone; burned lime; opoka mineral (Op; B = burned, N = not burned); the B horizon of a podzolised forest soil; and Leca (light expanded clay aggregates). Total P after saturation was determined in the different materials by extraction with 7 $\underline{M}$ HNO_3 in an autoclave (120°C for 30 minutes; SIS, 1986), followed by determination of P with ICP-AES (Perkin-Elmer, 1993).

Table 1. Amount of sorbent with 0.3 g P after saturation, P extracted by 7 $\underline{M}$ HNO_3, content of Al, Fe, Mg, Mn, and physical properties of the sorbents.
ASC=amorphous slag 0.25-4 mm; CSC=crystalline slag 0.25-4 mm; CSF= crystalline slag 0-0.125 mm; OpokaN=natural opoka; OpokaB=burned opoka; PodsolB=B horizon of a podzolised forest soil; +P=saturation with 1000 ppm P-solution for 24 h; col=daily applications of 10 ppm P for 13 months in columns (col); col+P=first "col" then "+P".

	Sorbent with	P-HNO_3	Al_2O_3	Fe_2O_3	Mg	Mn	Density	Hydr. Cond.
Material	0.3 g P (g)	(mg kg^{-1})	(g kg^{-1})	(g kg^{-1})	(g kg^{-1})	(mg kg^{-1})	(g cm^{-3})	(m/day)
$CaCO_3$+P	384	781	-	-	-	-	n.a.	n.a.
CaO+P	131	2290	-	-	-	-	n.a.	n.a.
OpokaN+P	236	1383	31	14	2	0.1	0.84	n.a.
OpokaB+P	235	1322	54	18	4	0.1	0.69	n.a.
PodsolB+P	196	1568	130	34	5	0.5	1.08	n.a.
Leca+P	615	498	182	62	15	1	0.45	648
CSC+P	572	524	25	2	43	3	1.57	16
CSF+P	484	620	27	2	49	3	1.61	1
CSCcol+P	238	1268	25	2	43	3	1.57	16
ASCcol+P	172	1750	26	2	30	3	1.47	41
CSCcol	349	961	25	2	43	3	1.57	16
ASCcol	323	990	26	2	30	3	1.47	41
CSFcol[*)]	830	394	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.

[*)] mixed with sand in relation 1:1
n.a. = not analysed

Barley, *Hordeum vulgare* L. c.v. Pernilla, was sown on June 6, 1998. The plants were thinned (June 22) to 40 plants pot^{-1}, watered daily when sunny weather, and rotated weekly to avoid skewness. Manganese solution (2.4 g*L^{-1}) was sprayed on the leaves twice (July 7 and 10) during the growing season. The plants were cut 10 mm above the soil surface on July 23, 1998, dried at 55°C, weighed, dried, digested in concentrated nitric acid (20 mL HNO_3 to 1 g sample) and boiled in a block digestor set at 125°C for 5.5 h. Double-distilled water was then added to obtain a total volume of 50 mL. A wetting agent (0.03 mL Triton X-100) was added on the following day. The Al, B, Ca, Cu, Fe, K, Mg, Mn, P, S, and Zn contents in the digests were subsequently determined using an ICP-AES (ICP-OES Perkin Elmer Optima 3000 DV; Perkin Elmer, 1993). Nitrogen was determined by Kjeldahl digestion using a Kjeltec-Auto 1030 Analyser (NMKL, 1976).

Data were subjected to analysis of variance and to multiple linear regression (MLR, Statistix, 1996). A significance level of 0.05 was used, unless otherwise indicated.

Results and discussion

The results from the AL-experiment showed that the most P was extracted from OpB (Fig. 1). Hence OpB apparently had the highest value as a P-fertiliser. Large amounts of P was also extracted from CSC2 and lime. Extractable P was substantially lower from the other two crystalline slag materials (CSF and CSC), while hardly any P was extracted from the amorphous materials (ASF and ASC). The coarser structure of the slag materials resulted in somewhat more P extracted. Limited amounts of P were extracted from OpN and even less from Leca. Also P sorbed by the Al- and Fe-rich B horizon was only to a small extent extracted by the AL-extractant. This is in agreement with the knowledge that plants primarily use soil-P in the form of Ca-P, while P in Al- and Fe-compounds is less plant available (Marschner, 1995). The high P-sorption capacity of Al- and Fe-oxide-hydroxides is difficult to exploit in the proposed waste water treatment systems, since it will severely reduce the possibility to recycle the sorbed P in agriculture.

The dry matter yield increased significantly at increased rate of added P, both in the form of a pure chemical (K_2HPO_4) and as P bound to sorption materials (Figs. 2, 3). Application of crystalline slag saturated with P gave even higher yield than application of P alone (Fig. 3). The reason might partly be ascribed to the Mg content of the slags (Table 1), since the soil was poor in Mg and possibly too little Mg was applied before sowing. The plant concentration of Mg was about 2 mg kg^{-1} DM in plants from all pots with slag application at the high P level and also in pots receiving fine textured slag at the low P level. This was significantly higher than in plants from other treatments, which had less than 1.5 mg Mg kg^{-1} DM, which is at the lower marginal for barley (Reuter and Robinson, 1986).

Contrary to what was expected from the P-AL values, application of opoka decreased the yield considerably (Figs. 1, 2). The treatments with podsol or Leca resulted in higher yield than the treatments with lime. The podsol or Leca treatments also gave higher yields than the opoka treatments at the higher P application rate. The latter case can be explained by the high P fertilisation level, resulting in that P loosely bound to Fe and Al in the podsol and Leca (Johansson and Hylander, 1998; Hylander *et al.*, 1999) resulted in considerable quantities of P available to the plants. A comparably short time elapsed between P addition to the materials and plant uptake, resulting in only limited crystallisation and other sequential reactions, which result in P containing compounds of low availability to plants, e.g. by transforming labile monodentate P complexes to irreversible bidentate complexes (Brady and Weil, 1996).

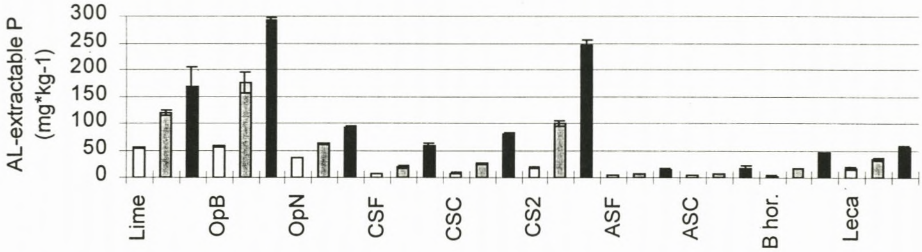

Figure 1. Ammonium lactate extractable P from various sorbents after equilibrium with P-solutions containing 2 (white), 10 (dots), and 20 (stripes) mg P L^{-1}.

Verticle bars indicate +/- SD. Op = opoka, B = burned, N = not burned, CSF= crystalline slag 0-0.125 mm, CSC=crystalline slag 0.25-4 mm, CS2=fresher and coarser slag (<7 mm) than CSC, ASF= amorphous slag 0-0.125 mm, ASC=amorphous slag 0.25-4 mm, B hor. = B horizon of a podzolised forest soil.

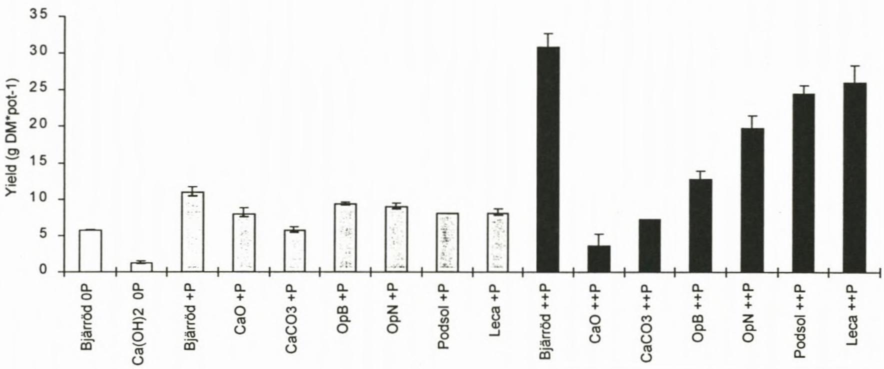

Figure 2. Yield (average and 95 % confidence interval) of barley plants (stem and leaves) grown in a soil fertilised with different materials saturated with P.

0P = no P-fertilisation, +P = 0.03, and ++P = 0.3 g P pot^{-1}. Bjärröd is original soil without sorbing material, Op = Opoka, B = burned, N = not burned, Podsol = B horizon of a podzolised forest soil.

The low yields after application of limestone and burned lime, and to a smaller degree after application of opoka (Fig. 2), was partly caused by the pH-increasing effect of the materials, resulting in Mn deficiency. All pots were sprayed with Mn after the first symptoms of grey speck disease was observed. This resulted in optimal tissue levels of Mn at the end of the growth period. The pots with slags had less or no symptoms of grey speck disease, although the slag is basic and used as a liming material, in agriculture. The reason is unknown, but the Mn content (~0.5%; Johansson and Hylander, 1998) of the slag might be one explanation. Another possible explanation is slower reaction in the soil, due to a coarser structure of the slag and a high Si content. Observations of treatments with grey speck disease during the growth period and the differences in yield at the high P application rate between treatments with natural and burned opoka and lime, respectively, (Fig. 2) support the influence of rapidly reacting liming material for worsening the grey speck disease.

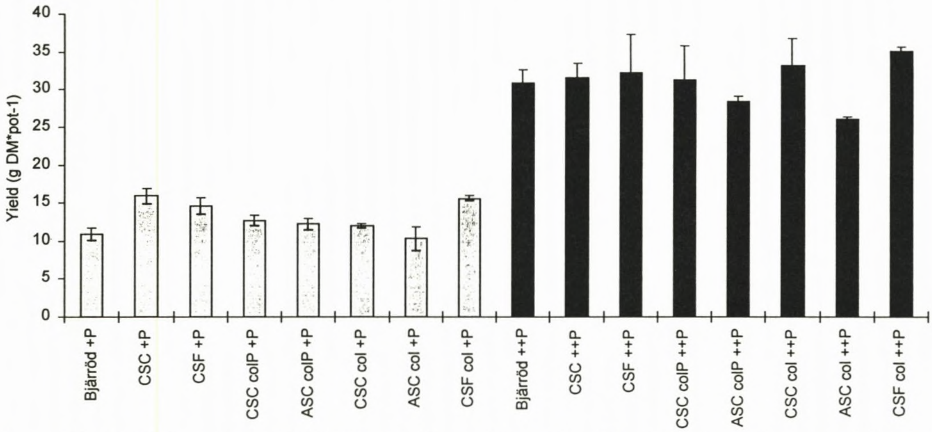

Figure 3. Yield (average and 95 % confidence interval) of barley plants (stem and leaves) grown in a soil fertilised with P sorbed to different materials.

0P = no P-fertilisation, +P = 0.03, and ++P = 0.3 g P pot^{-1}. Bjärröd is original soil without sorbing material, ASC=amorphous slag 0.25-4 mm, CSC=crystalline slag 0.25-4 mm, CSF= crystalline slag 0-0.125 mm, (mixed with sand in relation 1:1 in CSF col), col = the sorbent has received 13 months daily application of P-solution (10 mg/l) in a colonne, colP = col with additional P applicaton until saturation.

Plant concentrations of P generally followed the yield pattern, so it was highest in pots with highest yield (Fig. 4). Hence the P uptake pot^{-1} was highly variable between different treatments, contrary to N (Fig. 5) and K concentrations, which were generally lower in plants giving the highest yield. This was an effect of dilution of the applied amounts of N and K, which never dropped below adequate concentrations (Reuter and Robinson, 1986). An important exception from the low K concentrations at high yields, was the plants receiving CaO. Those plants had the lowest K concentrations in spite of the low yield and in addition the highest Ca concentrations, which were too high according to Reuter and Robinson (1986). These facts indicated that Ca ions from the powdery and hence rather reactive CaO had competitively reduced plant uptake of K. The influence of fineness to reduce the K uptake was also observed in pots receiving coarse or fine slag, though the differences were not significant there.

No plants had S deficiency, and most plants had high S levels at the low P application rate (Reuter and Robinson, 1986). The concentrations of B, Cu and Fe were optimal in all plants. The Zn concentrations in plants receiving CaO and fine slag were at the marginal, and optimal in the others.

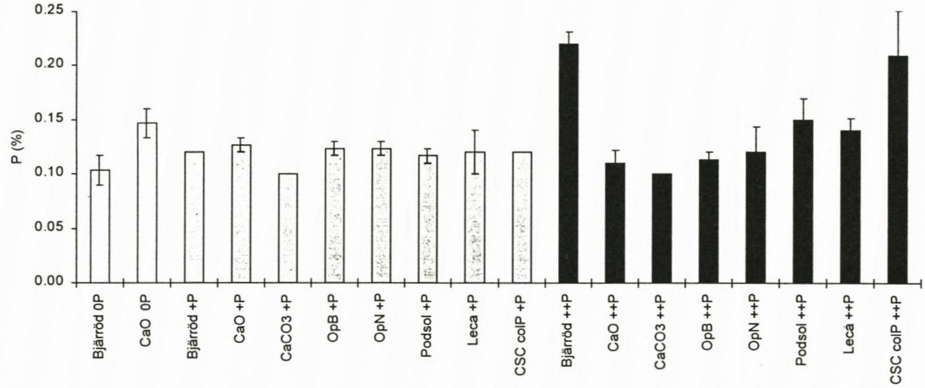

Figure 4. Phosphorus concentration (average and 95% conf. interval) in barley plants (stem and leaves) grown in a soil fertilised with different materials saturated with P.

0P = no P-fertilisation, +P = 0.03, and ++P = 0.3 g P pot^{-1}. Bjärröd is original soil without sorbing material, Op = Opoka, B = burned, N = not burned, Podsol = B horizon of a podzolised forest soil. CSC colP=crystalline slag 0.25-4 mm, which has received 13 months daily applications of P-solution (10 mg/l) in a colonne, and additional P applicaton until saturation.

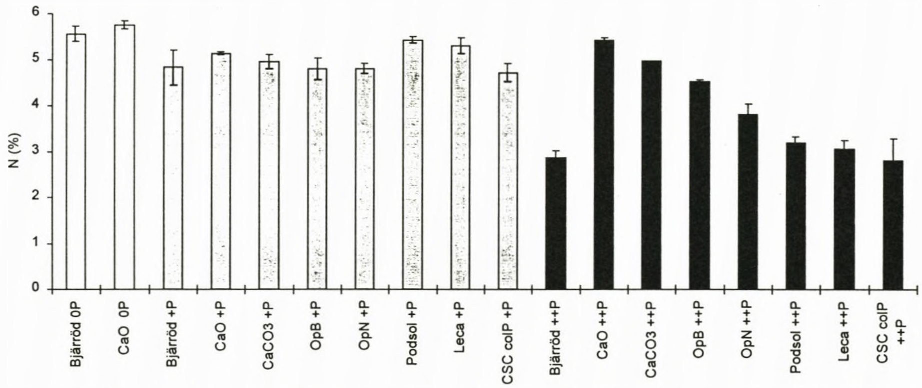

Figure 5. Nitrogen concentration (average and 95% conf. interval) in barley plants (stem and leaves) grown in a soil fertilised with different materials saturated with P.
Treatments are explained in Fig. 4.

Conclusions

Materials like opoka and slags containing Ca have a good sorption capacity for P and can be used for removing P in robust wastewater treatment plants. Considering heavy metal contents and hygienic aspects are acceptable, they can be used as P fertilisers and soil conditioners when their P sorption decrease. The present study indicated that the sorbed P is readily plant available. With many soils the amounts applied must be based on the liming effect of the sorbent rather than on optimal P amount. Further research is needed regarding the plant availability of sorbed P in different soil types under field conditions and after ageing of the filter substrates. This to avoid breaks on the path towards a sustainable society by reducing the P flow towards the oceans via recycling.

Acknowledgements

The study was jointly financed by the Swedish Council for Building Research, the Swedish Farmer's Foundation for Agricultural Research, and VA- Forsk. The blast furnace slags were supplied by SSAB Merox AB, Oxelösund, Sweden.

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Session 6
Danish Institute of Agricultural Sciences report no. 13

Composting toilets in permanent houses

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Abstract

The TTS Institute, in co-operation with the University of Kuopio, made a study on the functioning and typical problems with the composting toilets in continuous use, and how to prevent such disturbances. The study was carried out by interviewing 23 users of composting toilets in Finland and Sweden, and by on-site visits.

Users of composting toilets were fairly satisfied with their devices, although functional shortcomings and wrong constructions were noticed. Environmental friendliness and reduced use of water gave significant advantages. Many users found it important to have the composting toilet inside the house instead of an outside dry sanitation. On the other hand, flies and incomplete composting in the container were negative aspects in using this kind of toilets. The odour problems were sometimes mentioned. Due to these problems some people have given up their composting toilets.

Composting toilets require more care than conventional water closets, and a positive attitude from users in order to operate well. Many problems could be avoided if the devices were properly located and composting conditions for faeces in waste container optimised. More attention than before should be paid to subsequent composting and productive use of toilet waste to promote the recycling of nutrients.

Introduction

In Finland, approximately 800 composting toilets for round-the-year use are sold annually. In addition, numerous home-made composting toilets are installed, and their number is difficult to estimate. Composting toilets have become increasingly popular since the mid 90's, but replacing a water closet with a composting toilet is still rare, and the decision requires a lot of knowledge from the user (Malkki *et al.*, 1997).

The water closet is the most significant single pollutant in households. If it is replaced with a composting toilet, the nitrogen emissions to waters are reduced by 90%, and the release of organic waste and phosphorus is cut to less than half. Furthermore, overall water consumption decreases by a third, and wastewater load reduces notably.

Human faeces is considered a valuable nutrient source in many countries. The annual amount of toilet waste is about 520 kg/person. This amount includes alltogether 7.5 kg of nitrogen, phosphorus, and potassium, and some micronutrients in a form useful for plants. If the nutrients in the faeces of one person were used for grain cultivation, it would enable the production of the annual amount of grain consumed by one person (250 kg) (Wolgast, 1993). In Finland, faeces typically ends up in wastewater treatment plants where they blend with other wastewater. At the same time, their use value decreases.

The popularity of composting toilets is restricted by biased attitudes and the lack of information. Their reliability of operation is also uncertain, which is why people are not eager to buy them for their homes for all-year use. However, composting toilets could be very useful particularly in areas of scattered settlement, where wastewater containing faeces is absorbed into the ground and forms a pollution risk to the groundwater and nearby water bodies. The lack of fresh water for households outside public water distribution systems also provides grounds for interest in composting toilets.

TTS Institute and University of Kuopio studied the functionality of continuously used composting toilets, their typical malfunctions and ways to prevent these. Based on the users' experience and observations, estimations were made of how currently used toilet models could be technically improved for better functionality. Studies were also made of whether it would be possible to avoid problems simply by changing the ways people use their composting toilets.

Materials and methods

The research was conducted during 1996 and 1997 by interviewing 20 Finnish and 3 Swedish composting toilet users and visiting them in their homes. Data on the toilets were received from manufacturers, retailers, and experts in the field. The composting toilets included in the study were the following: *Aquatron* (1 toilet), *Clivus Multrum* (2), *Ekotuoli Biolet* (2), *Husqvarna* (1), *Naturum* (3), *Toa Thron*e (5), *Vera* (3), and *home-made composting toilets* (6). Drawings of the toilets and short descriptions of their operating principles are shown in Appendix 1.

The oldest composting toilets in the study had been in use since 1971, and most of the others since the early 1980s. The home-made toilets had large containers and resembled the *Clivus Multrum* except for one toilet, which had been constructed according to the *Asila* model (Lilja and Hyttinen-Lilja, 1991).

Most of the families interviewed were small. There were 15 households of one or two persons (65%), six households of three to four persons (26%), and two households of five to six persons (9%). In some households some members of the family did not use the composting toilet, but preferred the water closet or a privy. Twelve households had joined the municipal water distribution system, others had their own wells. None of the families belonged to the municipal sewage system, instead they managed their own wastewater treatment.

In the non-separating models (*Clivus Multrum*, *Toa Throne*, *Vera*, *Ekotuoli Biolet*, and *home-made composting toilets*), faeces composts with the litter in the waste container under partly or completely aerobic conditions, and in the presence of microbes. In the separating models (*Aquatron*, *Husqvarna*, *Naturum*), urine and faeces are separated either in the seat compartment or immediately beyond it. Each substance is treated separately; urine is led into a container in the ground, and solid waste is usually composted in a container placed under the seat.

Results

Why to use a composting toilet?

Nearly half of 23 families with composting toilets were "green", ecology oriented people. They wanted to contribute to reduce the pressure on the environment. It was important for them to return the nutrients in faeces to the cycle of nature by composting toilet waste and using it as soil improvement material on their own plots of land. The chance to reduce the consumption of drinking water was also considered important. This was particularly emphasized in those four families who lacked fresh water. In three households, constructing a closed well for toilet wastewater would have been almost impossible, or too expensive due to rocky plots. In some families, the composting toilet was situated in the house, which made the home more convenient, and the privy could be demolished. The lack of odour from the composting toilet was regarded as an advantage; the families were also content with the smaller water and sewage fees which resulted from the decreased water consumption.

Nearly 70% of the families were satisfied or very satisfied with their composting toilets. Even though some toilets failed to function from time to time, the families still did not want to replace the composting toilet with a water closet. The composting toilet was a conscious ecological choice, and people were committed to doing extra work and allowing time for this.

Only two families were completely discontent with their composting toilets and would rather have them replaced with water closets. The function of their toilets was exceptionally poor, and the faeces mass failed to compost in the waste container. The rest of the families were more or less satisfied with their composting toilets.

Bad structural solutions and malfunctioning

The waste containers of the composting toilets with large containers (*Clivus Multrum*, *Toa Throne*, and most of *the home-made models*) was typically situated in poorly accessible places. Mixing the faeces mass, which stimulates the composting process in the waste container, was almost impossible in the small space provided. Emptying the toilet was also difficult and dirty (unhygienic), particularly in places where the mass had to be taken out through living rooms.

In models with large containers and several compartments, the waste containers should be located in a place where there is direct access from outside the building. The container should be accessible, for example with a wheelbarrow into which the half-ready compost could be emptied. We saw only a few such ideal solutions, which indicates that not enough attention had been paid to the usability of composting toilets. In Finland, construction rules should be specified in this area.

Unpleasant odours used to be among the most typical problems in composting toilets (Malkki and Vanhala, 1994, Malkki *et al.*, 1997). The composting toilets that we studied were usually odourless, thanks to the electric fan installed in the ventilation pipe. In several households, the fan was on continuously and at full capacity. With respect to energy consumption, a low-power (15W) fan is not a problem, but excessive ventilation dries the waste mass and slows down the composting process.

In all the composting toilets there had been flies and other insects which could spread faeces-related diseases. The usual way to control insects was to spray pyrethrin-based or chemical pesticides in the waste container and the toilet room. Some people had also tried flypaper and a bacterial product *Bacillus thuringiensis*. In one case, there had been predatory insects in the waste container which had eliminated the flies.

Most of the toilet models were of the kind where faeces composts only partly. The uncomposted waste mass is unpleasant to handle, and it is also a potential source of infections. The composting process can be stimulated by creating optimum conditions for it. In practice this means that litter must be added according to the waste container instructions, in order to absorb excess moisture and odours and aerate the mass.

Litter was added to the toilet waste in 19 households. Two or more mixtures at a time were generally used in the models with large containers. In addition to peat or wood-containing litter, seven families also added household waste to the toilet, and in two families only household waste was used as litter. Three families completely neglected to use litter.

Even if litter was added to the container, in the large containers there was often excess liquid at the bottom, sometimes even so much that it had to be emptied with a bucket. This implies that not enough litter was being used, or that its absorbing capacity was poor.

The formation of excess liquid should be minimised in order to prevent effluent from the composting toilet from passing into the environment. Excess liquid rich in nitrogen should be recycled into the waste container or cleaned separately before allowing it to be absorbed into the ground. Problems with excess liquid can be prevented by controlling the quantity and quality of litter and by mixing the waste mass regularly. The ventilation must also function efficiently.

The quality of the mass removed from the waste container

In the households under study, the composting toilets with large containers (*Aquatron, Clivus Multrum, Toa Throne, Vera, home-made models*) were emptied at intervals ranging from six months to five years, usually once every two years. Small toilets with integrated waste containers (*Ekotuoli Biolet, Husqvarna, Naturum*) were emptied every two to four weeks.

When emptying the toilets, the quality of pre-composted faeces varied considerably depending on the model and the individual toilet. Since, in the *Vera* carousel toilets, the composting process of the faeces mass may have for lasted years before the container is emptied and fresh and old material is not mixed, the old mass is ready to be used as soil improvement material. In the *Naturum* toilet's cylinder, peat, solid faeces, and toilet paper were mixed well before they moved on into the container for composting; when the container was emptied, the mass was dark, homogenous, and odourless, and it was post-composted either in a composter or heaped in the field.

In the *Husqvarna* electric toilet, solid faeces dries in the waste container. It had to be moistened before post-composting, and only afterwards it could be mixed with other organic waste. In the bio-chamber of the *Aquatron* toilet, worms disintegrated solid faeces and toilet paper almost completely. Adding litter once a week increased the aeration of the mass and stimulated the worms. When the bio-chamber was emptied, the mass was ready to be used as soil improvement material almost immediately.

Both of the families using the *Ekotuoli Biolet* composting toilet had had difficulties making the toilet work. The mixture of faeces and litter was either too dry or too moist, and it did not drop readily through the grating in the lower part of the seat into the waste box. The toilets had therefore to be emptied every couple of weeks through the toilet seat with a ladle and a bucket. The reeking mass was then composted in a composter.

The *Clivus Multrum*, *Toa Throne*, and *home-made toilets* were emptied every one to five years. Even though the composting process in the waste container was long, the quality of the mass varied considerably depending on the individual toilet. In composting toilets with large containers, pre-composted and fresh waste was mixed in the waste container, which is why the mass had to be post-composted heaped in the field for over six months.

Utilisation of composted toilet waste

A lot of biased attitudes are connected with the utilisation of composted toilet waste. Some of the composting toilet users we interviewed considered it unpleasant and strange to grow vegetables with a faeces-based fertiliser.

The usual way to utilise the compost was to spread it in the yard under bushes or on waste land. Every fifth family spread toilet waste on potato fields or vegetable plots. Some families buried the waste in the ground or even took it to a wastewater treatment plant. In these households the toilets failed to function properly.

In the Finnish households with separating toilets urine was not collected, but was led into grey waters or absorbed into the ground through a septic tank or a sand filter. These families had not acquired separate urine containers, because they considered them unnecessary or too expensive. They also did not know the nutrient content of urine or for which plants and at which point of the growing period it was safe to spread urine without rendering the plants unhygienic. The Swedish family with had a home-made separating toilet had located the urine container in the ground. Diluted urine was used for watering the garden, and once a year a nearby farmer emptied the container with a sludge van and spread the urine on his own field.

The families in this research project had no problems with finding a place for the toilet waste. Since the amount of composted faeces mass is small, it is easy to find a place for it in the yard of a one-family house. If, for example, there were composting toilets in a terraced house in a densely populated area, the amount of toilet waste could become too large. Where it could be safely placed, is another question which should be studied.

Conclusions

The use of composting toilets saves water, decreases environmental load, and increases the recycling of nutrients. In order to function well, composting toilets require positive attitudes and more concern from their users than water closets.

Typical problems with composting toilets included incomplete composting of the waste mass, the emptying of the containers, flies, and bad odour. Some of these problems can be solved by creating favourable conditions for microbes in the waste container. More attention should be paid to the use and quality of litter in order to optimise the moisture content of faeces mass with regard to composting, and to prevent the formation of excess liquid.

The utilisation of composted toilet waste and urine was insufficient in the households under study. Updated instructions for the local handling and spreading of toilet waste are necessary for an effective recycling of faeces nutrients. Users also need objective information on composting toilets and clear instruction manuals for each brand.

All malfunctions can not be prevented simply by changing the ways the toilets are used; composting toilets should also be improved technically. To increase the popularity of composting toilets, more reliable, easier to use, and inexpensive composting toilets are needed both for round-the-year and summer use. With the advent of ecological building, composting toilets for apartment buildings have also raised interest.

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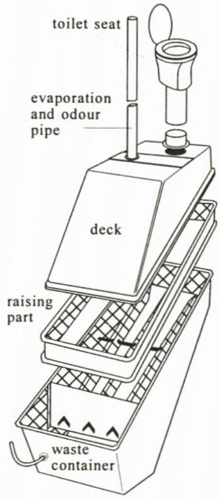
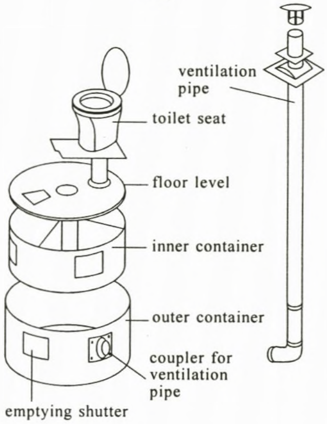
NON-SEPARATING COMPOSTING TOILETS

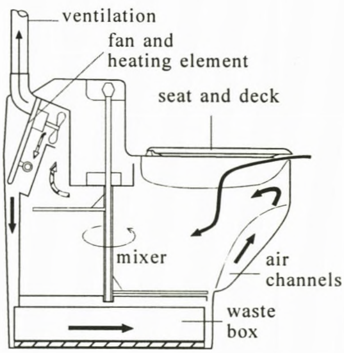


Clivus Multrum
(similar to Toa Thorne)
- needs space in two floors
- requires regular use of litter
- emptying interval is one year or more depending on the amount of waste and the size of the container
- fresh and pre-composted material is mixed together
- post-composting of the mass in a heap is necessary

Vera carousel toilet
- needs space in two floors
- the waste container is divided into four compartments, which are filled one by one
- requires regular use of litter
- first compartment is emptied when all the compartments are full and emptying is continued one compartment at a time once or twice a year depending on the number of the toilet users
- time required for composting of the faeces is long and fresh and old material is not mixed together
- the mass is ready to be used as a soil conditioner

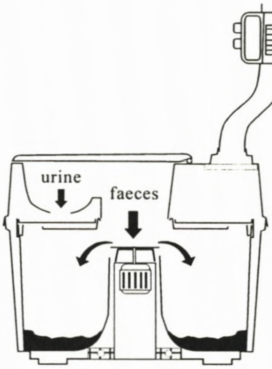


Ekotuoli Biolet composting toilet
- the waste container and the seat is combined
- urine is evaporated by an electrical fan and a heating element
- part of the urine is mixed with the faeces and the litter
- the waste mass is mixed with a scraper
- when emptying the toilet, the front shutter is loosened and the container is pulled out
- the waste mass is post-composted in a composter or in a heap before the end use
- the toilet can be installed in a room standardised to water closet
- proper functioning requires warm temperature



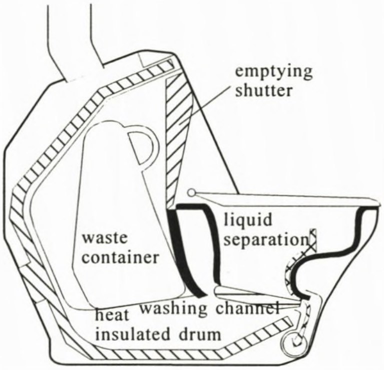
SEPARATING COMPOSTING TOILETS

Husqvarna electrical toilet



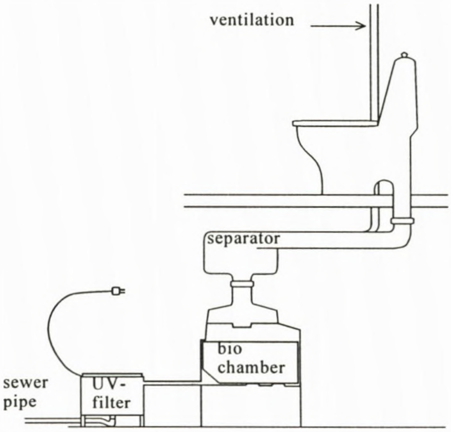
- the urine is lead down from the front part of the seat to automatically heated plates. The urine is evaporated by means of heating element and fan. Solid faeces and toilet paper are falling down from rotating flushing plates to the waste container
- emptying interval is from 2 to 8 weeks
- the waste mass is post-composted in a composter or in a heap
- the toilet can be installed in a room standardised to water closed
- proper functioning require warm temperature (+18°C)

Naturum drum composting toilet



- the urine and solid faeces is separated in a toilet seat with a liquid separator
- the urine is lead through a pipe to a sewer or to a container which is located outside the building into a ground
- the solid part is composted together with litter in a thermally insulated drum composter
- rotary movement leads the mass gradually to an emptying container, where the mass continues composting
- the full waste container in emptied into a composter or a heap for post–composting
- the toilet can be installed in a room standardised to water closet

Aquatron toilet



- needs space in two floors
- the toilet is composed of an ordinary toilet seat, separator, biochamber and UV-filter
- the liquid flows to UV-filter to be sterilised with radiation. Afterwards the liquid is lead to a sewer pipe and absorbed into the ground
- living worms are used to compost the waste at a temperature of 12-25°C in a biochamber
- toilet is emptied once per year
- excess liquid is lead to back edge and to a sewer pipe
- a proper use requires electricity and sewer

J. Petersen & S.O. Petersen (eds.) 1999. Use of municipal organic waste
Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
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Danish Institute of Agricultural Sciences report no. 13

Management of a farmer operated system for recycling municipal organic waste to agriculture - a case study from four Norwegian municipalities

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Introduction

In many European countries, the environmental authorities recommend that organic wastes from society be recycled to agriculture as a source of plant nutrients. In areas with available farm land that is not already heavily loaded with livestock manure, such recycling represents a sustainable and agronomically beneficial solution.

However, this leads to questions related environmental risks, economic aspects for the municipalities and the farmers, and different ways to operate the system. So far the farmers' role has been of a passive character, just receiving, without charge, a waste product from a treatment plant.

Technologies

To improve the interest among farmers, and to improve the environmental benefits, R&D projects aimed at developing technology and a management system have been carried out. The waste types given the highest preference were municipal sludge and food waste. The recycling is based on local solutions and small cycles, but still big enough to be economically competitive.

The basic idea is to give operational responsibility to that party which has the greatest interest in creating good control routines and a safe product, that is, the farmer himself. The technology is based on handling of all wastes as liquids, processing in an aerobic reactor, and application by commonly used slurry spreaders to farm land for growing cereals or grass. A new system for handling food wastes is now being developed. It integrates sewage sludge and food wastes into the same system and reduces the overall costs.

Results

Most of the project results have been, or will soon be, implemented for practical use. Four full scale plants, all located on farms in rural areas, have been in commercial operation for between one and three years. All parties involved have entered into agreements that regulate the operational management and the economy for the farmers, the community and the inhabitants.

Economic calculations show that the income for the farmer represents costs for at least half a man-year per year, when handling wastes from 700 dwellings. Environmental benefits are high, almost no nutrients are lost on their way from dwellings or other sources to the soil. The system operates without odour problems.

Experiences from these four municipalities, the commercial aspects and the management as a whole, is presented in this paper. Also systems for quality control of the recycled wastes, which in fact are raw materials for new food production, are discussed.

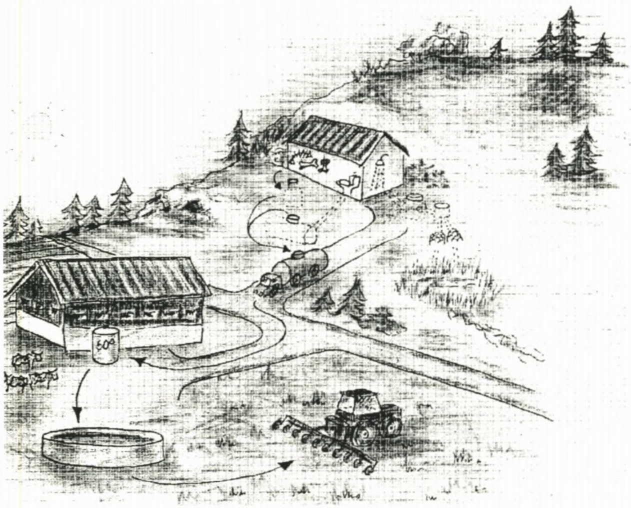

Figure 1. A farmer operated system for recycling municipal sludge and food waste to agriculture, independent of sewage pipes and central treatment plants. About 700 houses are linked to one processing plant, which converts the wastes into a hygienic and stabilized liquid product to be applied on farm land. The system gives the farmer an income corresponding to half a man-year.

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Proceedings of NJF seminar no. 292, November 23-25, 1998, Agricultural Research Centre, Jokioinen, Finland
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Source-separated household waste in urban-rural recycling

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Abstract

Organic waste from households is increasingly being composted in Sweden and used as soil amendment or a fertilizer in urban areas. The purpose of this paper is to discuss possibilities for use of such waste as fertilizer in agriculture.

The quality of source-separated household waste, composted on a small scale, was analysed. Main quality factors were the content of plant nutrients and heavy metals. In the study, the evaluation of composted household waste used the SEPA (Swedish Environmental Protection Agency) regulations for use of sewage sludge in agriculture (SNFS 1994:2) which is based on the total amount of ammonia, phosphorus and heavy metals applied per hectare and year.

The application of the materials investigated was limited by their phosphorus content rather than their content of heavy metals. For most heavy metals the amount applied was much lower than the maximum permitted by SNFS 1994:2. Cadmium, zinc and, in some cases, nickel, were exceptions.

Nutrient rich source-separated household waste, composted on a small scale, may be recycled in agriculture as PK-fertilizer. This is an important step for the closing of the cycle of nutrients between town and country.

Introduction

In Sweden, a large fraction of the domestic waste is incinerated or deposited in landfills. Incineration takes place in specialized incineration plants for mixed household waste, and the energy produced is delivered to neighbouring towns in municipal district heating systems. The ash from the process is not returned to agriculture, so phosphorus, potassium and other plant nutrients are lost. This means that both dumping and incineration of household waste without ash recycling is questionable from a recycling viewpoint (Svensson, 1997).

Household waste can be handled in a more recycling oriented manner by using biological methods such as fermentation and composting. A very important question has to be answered before large-scale treatment schemes for household wastes are initiated. Can well source-separated household waste be of sufficient quality with regard to plant nutrients and heavy metals to allow the by-products of biological waste treatment to be used on arable soil?

If the by-products are not of sufficiently high quality to be used in agriculture, this invalidates the entire suggestion that biological treatment processes can be used to complete the cycle of nutrients between town and country.

Quality of source-separated household waste

Controlled small-scale composting

The aim of this study was to determine the nutrient and heavy metal concentrations in compost based on source-separated household waste and other easily decomposed organic waste products from e.g. shops, restaurants and large-scale kitchens. The composting experiments were carried out in Borgeby (The Rural Economy and Agricultural Society of Malmöhus) in small, closed rotating compost bins (drum composters) manufactured by Joraform (Christensson, 1996).

Composted source-separated household waste

Source-separated household waste from Värpinge in Lund, which was composted with *clean* fractions of sawdust and straw, contained very low amounts of heavy metals and relatively high amounts of nutrients in the finished compost (Christensson, 1996). Of the material making up the compost in these experiments, 75% of the fresh weight derived from household waste, 10-20% from straw and 5-15% from sawdust.

The application of compost on arable land should be regulated according to its content of nutrients and heavy metals. Calculations using the median results of 5 chemical analyses of the compost based on source-separated household waste, sawdust and straw showed that the compost application rate was limited to 9.2 tonnes DM per hectare if regulations for the use of sewage sludge in agriculture (SNFS 1994:2) were applied. This application rate supplied 138 kg total nitrogen, 22 kg of phosphorus and 119 kg of potassium (Table 1). The application rate for the compost was limited by the phosphorus content.

The amount of plant nutrients which could be applied to arable land as compost fulfilled the P and K requirements of most agriculturals crop during one growing season. However, a supplement of readily soluble nitrogenous fertilizer would be necessary to meet the N reqirement of the crop. The compost contains relatively large amounts of nitrogen but these are mainly bound to the organic matter and their rate of release is usually slow compared to what the crop would require to develop properly.

It could be concluded from these experiments that the compost was of such high quality that it should be of interest for use in agriculture. The value of the nutrients supplied per hectare was approximately SEK 740, based on the following costs of equivalent amounts of artificial fertilizers: Nitrogen (N28) SEK 6.50/kg, phosphorus SEK 10/kg and potassium SEK 3.60/kg. Of the total nitrogen, it was estimated that 10% was released annually as readily-available nitrogen, which meant that the value of the total nitrogen was reduced to SEK 0.65/kg. The cost of spreading 9.2 tonnes DM of compost per hectare was assumed to be SEK 400.

Table 1. Maximum amounts of plant nutrients and heavy metals applied per hectare and year based on regulations for use of sewage sludge on arable land (SNFS 1994:2)

	N-tot	P	K	Pb	Cd	Cu	Cr	Hg	Ni	Zn
	kg	kg	kg	g	g	g	g	g	g	g
Controlled small-scale composting (Borgeby)										
Composted source-separated household waste	138	22*	119	7	1.3	89	18	0.6	46	578
Composted potatoes, carrot and white cabbage	116	19.4	194	11	1.75*	58	19	0.6	49	681
Small-scale composting in practice										
Stockholm area	122	22*	61	31	0,9	116	31	0.3	12	287
Malmö-Lund area	117	22*	50	11	1.3	85	16	0.5	5	436
Non-composted food waste and source-separated household waste										
Rondeco	166	22*	50	18	0.6	89	30	0.1	42	248
VAFAB	106	22*	42	28	0.63	99	25	0.2	11	242
SNFS limit from 1995	-	*22*	-	***100***	***1.75***	***600***	***100***	***2.5***	***50***	***800***

* = the element which limits the application according to current regulations (SNFS 1994:2) on use of sewage sludge on arable land. It should be noted that on soils with low P contents (soils of P-class I and II), the P application may be increased to 35 kg per hectare and year.

Composted potatoes, carrots and white cabbage

In another experiment at Borgeby, potatoes, carrots and white cabbage were composted with the same amounts of sawdust and straw as described previously for source-separated household waste. The vegetables used were of food quality and were purchased over the counter for each composting. Of the material making up the compost in this experiment, 75% of the fresh weight derived from the vegetables, 8-15% from straw and 7-15% from sawdust (Christensson, 1996).

Calculations based on the median results of 5 chemical analyses on the compost produced from potatoes, carrots and cabbage showed that a maximum of 116 kg of total N, 19.4 kg P and 194 kg K could be applied per hectare and year (Table 1). The application rate of the compost was limited to 9.7 tonnes DM per hectare by the cadmium content when regulations for the use of sewage sludge on arable land (SNFS 1994:2) were applied. It could be concluded from this experiment that a compost based on potatoes, carrots and white cabbage was also of sufficiently high quality for agricultural use, even if the application rate was limited by the cadmium content. The fertilizer value per hectare of this compost (9.2 t DM) was an estimated SEK 970.

The composting experiments carried out at Borgeby showed that under 'ideal' conditions, compost based on source-separated composted organic wastes from households, restaurants, large-scale kitchens etc. was of a sufficiently high quality with respect to plant nutrients and heavy metals that it could be used freely as an organic PK fertilizer in agriculture.

Small-scale composting in practice

The level of quality which can be obtained in practice in small-scale drum composting of source-separated household waste was determined in two 'spot sample investigations' in some residential areas in Stockholm and in the Malmö-Lund area.

Drum composters of type Jora 1400 were installed in residential areas in the suburbs of Stockholm (Botkyrka, Haninge, Huddinge, Nynäshamn and Salem). During composting of the source-separated household waste, sawdust or wood pellets were added to improve the structure of the material and as carbon sources. During the autumn of 1996, 5 spot samples of compost were extracted from the drum composters for chemical analysis.

Calculations based on the median values of 5 chemical analyses showed that a maximum of 122 kg N, 22 kg P and 61 kg K could be applied per hectare and year if regulations on the use of sewage sludge on arable land were applied (Table 1). The compost application rate was limited to around 6 t DM per hectare by the P content. The fertilizer value per hectare of this compost (6 t DM) was an estimated SEK 520.

In residential areas of Malmö (Gullvik and Jämlikheten) and Lund (Ladugårdsmarken) various types of drum composter were used as follows: rotary compost bins from Jora (JK 270), a Compox drum composter and two Nerthus Garden drum composters. During the spring of 1996 and 1997, spot samples of compost were extracted from some of these drums for chemical analysis. In total, 5 spot samples were taken of compost based on source-separated household waste and sawdust. During composting, 1 volume of sawdust was normally added to 3 volumes of source-separated household waste. In these proportions, the sawdust contributed 33 % of the DM content added to the composter (Karin Persson, *pers. comm).*

Calculations based on the median values of 5 chemical analyses showed that a maximum of 117 kg N, 22 kg P and 50 kg K could be applied per hectare and year (Table 1). If regulations on the use of sewage sludge on arable land were applied, the compost application rate was limited to around 5.4 t DM per hectare by the P content. The fertilizer value per hectare of this compost was an estimated SEK 480.

Non-composted food waste and source-separated household waste

The 'spot sample' experiments were complemented by two further 'analysis series' to determine the potential of source-separated household waste for re-cycling nutrients to agriculture. One series of analyses determined the average quality of the food waste fraction in samples of mixed household waste within the Rondeco project in Stora Vika, Nynäshamn (Berg *et al.*, 1998)

The company VAFAB in Västerås has also carried out an investigation into the nutrient and heavy metal contents of source-separated household waste from houses and apartments. Table 1 shows the results from 4 analyses during 1996 on waste from houses in Skultuna. Four analyses of source-separated household waste from apartments in Skultuna during 1996 showed that it was of the same high quality as that from the houses (Torbjörn Ånger, *pers. comm.)*

If the compost from the Rondeco project or VAFAB were to be spread directly on arable soil without any biological treatment, the fertilizer value of the material would be approx. SEK 440-500 per hectare. The application rate was limited by the P content of the material. This shows that the material was of high quality. Of course this does not mean that untreated waste should be spread on arable land, but once again the analyses showed that the quality of source-separated household waste can be sufficiently high with respect to nutrients and heavy metals for it to be returned to arable land.

Discussion and conclusions

This compilation shows that source-separated household wastes *can* be of very high quality with respect to nutrients and heavy metals. When the heavy metal and nutrient contents of source-separated household waste and compost were judged according to the environmental regulations adopted for the use of sewage sludge on arable land in Sweden (SNFS 1994:2), the amount which could be applied per hectare was limited by the phosphorus contents of the products. At the same time, the application of restricted heavy metals to arable land was often low in relation to the existing limits except for cadmium and to a certain extent nickel and zinc (Table 1 and figure 1).

The very high quality with respect to nutrients and heavy metals in this investigation was probably due to thorough separation at the source, to the use of only clean structure improvers and carbon sources as additives, and to the fact that composting took place in closed composting drums.

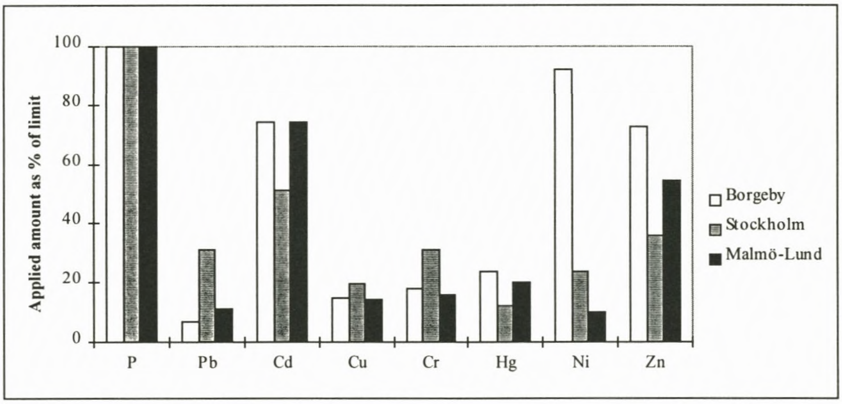

Figure 1. Applied amounts of phosphorus and heavy metals from composted source-separated household waste, from three areas, as a percentage of the limits according to regulations governing the use of sewage sludge on arable soil.

Apart from source-separated household waste, there are other easily decomposed organic by-products which have sufficiently high quality that they can be used as fertilizers in agriculture after composting. The project ”LRV Test Compost – Analysis of Organic Waste Fractions from Households, Large-scale Kitchens and Shops” showed that different source-separated organic waste fractions, which were composted with *clean* fractions of sawdust and straw, contained very low amounts of heavy metals and relatively high amounts of plant nutrients in the finished compost (Christensson, 1996).

Carefully source-separated household waste and other clean biologically degradable waste fractions *can* be of such high quality, with respect to heavy metals and nutrients, that the material produced by biological treatment can be returned to agriculture as PK-fertilizer. However, the high quality of the source-separated household waste must be maintained and not be ‘spoiled’ by the heavy metal content of added structure improvers such as sawdust and wood pellets.

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Results of Analyses

Analysis Ref. Nos. 22544, 22545, 22546, 23731 and 23733, Biospectron, Tågarp.

Analysis Ref. Nos. G97-16436 and G97-16437. AgroLab, Kristianstad

Analysis Ref. Nos. 96-013081 - 96-013085. KM Lab, Uppsala.

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ISSN 1397-9884