THE EFFECT OF PUTRESCIBLE DOMESTIC WASTE SEPARATES ON SELECTED SOIL CHARACTERISTICS AND PLANT GROWTH

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3.	23 February 1979	The Disposal of Toxic Waste. Chemical Society - Royal Institute of Chemistry Industrial Symposium University of Leeds.
4.	March 1979	Statistical routines on the computer - a short course run by Sheffield City Polytechnic's Computer services.
5.	28 - 30 March 1979	Decomposition by Basidiomycetes . British Mycological Society Queen Mary College, London.
6.	22 - 25 October 1979	Reclamation of Contaminated Land. Society for Chemical Industry Eastbourne.
7.	29 March 1980	Conflicts in Rural Land Use. Institute of Biology (Environmental Section) Royal Geographical Society, London.

8.	15 - 18 September 1980	Conflicts in Land Use. British Society of Soil Science University of Oxford. A poster demonstration of the present research work was displayed at this conference.
9.	17 October 1980	Using the graphics terminals - short course run by the Sheffield Polytechnic's Computer Services.
10.	14 November 1980	Ecology and Industrial Society - The Problem of Waste. NATFHE Biological Education section Conference, Sheffield City Polytechnic. A paper on the current research was presented.
11.	29 October - 3 December 1980	Experimental design - a short course for staff run by members of the Dept of Mathematics at Sheffield City Polytechnic.
12.	17 December 1980	Winter meeting of the British Ecological Society, University of Leeds.
13.	24 - 26 March 1981	Nitrogen as an Ecological Factor. British Ecological Society Symposium, University of Oxford.
14.	7 - 8 April 1981	Redevelopment of Contaminated Land. A course organised by Thames Polytechnic and the London Borough of Greenwich.
15.	23 September 1982	Water Specialists Meeting (Public Health Engineering Section) University of Warwick.

16. 14 December 1982 — Biodegradation of Waste Materials. Biodeterioration Society Winter Symposium, Sheffield City Polytechnic. Research paper was presented.

VISITS

1. 2 April 1979 — Biodeterioration Centre, University of Aston in Birmingham to survey relevant literature.

2. October 1981 — Waste Management Information Centre at U.K.A.E.A., Harwell, to survey relevant literature (2 visits).

3. 16 March 1982 — National Vegetable Research Station, Wellesbourne, Warwick, to undertake experimental work using pressure plate apparatus for soil moisture determinations.

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In addition, two articles describing the current research have been published in NAWDC News (Journal of the National Association of Waste Disposal Contractors) in June 1981 and June 1983 respectively.

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ABSTRACT

THE EFFECT OF PUTRESCIBLE DOMESTIC WASTE SEPARATES ON SELECTED SOIL CHARACTERISTICS AND PLANT GROWTH

Mary B. Saull

Shortage of suitable landfill sites in South Yorkshire has aroused interest in recycling domestic refuse. The Doncaster separating plant reclaims useful materials from domestic refuse; a non-useful by-product is the highly organic "putrescible" fraction.

Pilot studies established the biochemical nature of this material and indicated that it was amenable to biodegradation in soil and potentially valuable as a soil conditioner. This investigation aimed to establish optimum dosage rates for its disposal and identify patterns of biodegradation with or without nitrogen supplementation using digested sewage sludge. Effects on the soil, and the maintenance of healthy plant life were considered using field and laboratory experimentation.

Results showed that organic materials in the refuse followed an exponential pattern of breakdown, increasing in rate up to a dosage of 47 tonnes/hectare and thereafter declining. Growth of barley and ryegrass responded well to refuse additions but again the increased yield was suppressed slightly at high dosages. This suppression appeared to be closely linked with soil levels of available nitrogen.

It is generally assumed that nitrogen supplementation is required to aid the breakdown of carbonaceous domestic refuse in soil. Sewage sludge addition was not found to increase the rate of breakdown of the Doncaster refuse separates in the long term. However, mineralisation studies suggested that nitrogen in the refuse was immobilised during the early stages of biodegradation until microbial utilisation of the carbonaceous materials had reduced the carbon:nitrogen ratio to a level at which mineralisation could proceed.

Soil physical properties were generally improved by the refuse and soil humic material was increased. There was little evidence of phytotoxicity and toxic metals were at acceptable soil concentrations. The main problems were high initial levels of boron and the persistence of some non-biodegradable material present in the refuse separates.

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ABBREVIATIONS USED IN THE TEXT

H.M.S.O.	Her Majesty's Stationery Office
D.O.E.	Department of the Environment
M.A.F.F.	Ministry of Agriculture, Fisheries and Food.
A.D.A.S.	Agricultural Development and Advisory Service
S.Y.C.C.	South Yorkshire County Council
C:N ratio	carbon : nitrogen ratio
HCl	hydrochloric acid
KCl	potassium chloride
M	molar
R.G.R.	relative growth rate, defined as $d(\frac{\log_e Y}{dt})$ where Y = shoot dry weight in grammes, t = time.
A.W.C.	available water capacity
Z.E.	zinc equivalent
%	percentage
m	metre
cm	centimetre
mm	millimetre
t	tonne
kg	kilogramme
g	gramme
μg	microgramme
l	litre
ml	millilitre
ha.	hectare
ppm	parts per million
v/v	volume per volume
h.	hour
yr	year
^{o}C	degrees Centigrade
μS	microSiemens
var.	variety
±	plus or minus

$<$	less than
$>$	greater than
$\bar{x}$	mean
σ	standard deviation
V	coefficient of variation
p	probability
b	regression coefficient
r	correlation coefficient
l.s.d.	least significant difference
n.s.	not significant
n.r.	not recorded.
ANOVA	Analysis of variance

" He gave it for his opinion that whoever could make two ears of corn or two blades of grass upon a spot of ground where only one grew before, would deserve better of mankind, and do more essential service to his country than the whole race of politicians put together. "

Jonathan Swift

(Gulliver's Travels, "Voyage to Brobdingnog".)

CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

1.1 The problem of solid waste disposal

The sophistication of today's society has created many problems, among them the ever-increasing burden of waste disposal. The most recently available figures for the U.K. indicate that 16.4 million tonnes of household and commercial waste was produced in the financial year of 1977 - 1978, equivalent to approximately 334 kg per head of population (H.M.S.O., 1979). Disposal of this waste is still predominantly by landfill, the cheapest method. However, with a localised shortage of suitable sites, increasing attention is being given to alternative methods (Table 1).

Sparcity in available sites for landfill is a problem in South Yorkshire. This fact, along with a recent upsurge in interest in the recycling of materials has prompted the instigation of new approaches to the problems of waste disposal in the county.

As part of a Department of the Environment project on the reclamation and recycling of domestic refuse a separating plant was built at Doncaster and opened in 1979. Based on a pilot plant developed at Warren Spring Laboratory, it reclaims useful materials such as paper, textiles, metals and glass for resale to industry or for the production of new materials, e.g. paper/plastic fuel pellets and glass-based tiles. A by-product from the reclamation lines is the so-called "putrescible" fraction which is highly organic in nature (Davis and Mills, 1977) and is at present still disposed of by landfill.

This investigation gives consideration to an alternative method of disposal of this unique material, that of direct soil incorporation. This should allow rapid biodegradation of the organics, release of plant available nutrients and may also be of value to the soil as a conditioning agent.

1.2 The return of wastes to land - historical development

1.2.1 Introduction

Disposal of waste materials on the land, and their use as

	1974-75 %	1975-76 %	1976-77 %	1977-78 %
Landfill Untreated	77	75	74	71
Shredding/Pulverisation	3	3	4	3
Direct Incineration	7	8	9	9
Separation Incineration	2	1	1	1
Disposal by Contractors	11	12	12	14
Other Methods (mainly composting)	-	-	-	1

Table 1. Methods of treatment and disposal by 46 Waste Disposal Authorities in England (H.M.S.O. 1979)

fertilizers for improvement of plant growth is not new. The practice is almost as old as man's history on earth and references to it can be found in the Old Testament -

> " And thou shalt have a paddle upon thy weapon;
> and it shall be, when thou wilt ease thyself
> abroad, thou shalt dig therewith, and shalt
> turn back and cover that which cometh from thee."
>
> (Deuteronomy Ch.23 v.13)

Russell (1973) tracing early literature on agricultural chemistry notes Palissy's remarkable statement made in 1563 -

> " You will admit that when you bring dung into
> the field it is to return to the soil something
> that has been taken away."

Thus for many years it has been, and indeed still is today, common practice to return agricultural and human wastes to the land, both as a means of disposal and also of "manuring" the soil. More recently mining slag and many types of industrial wastes have been applied to land (Carlson, 1976) and interest has been shown in the successful disposal of oily and petroleum wastes in the soil (Phung and Ross, 1978; Huddleston and Meyers, 1978). Nowadays the focus of research attention is on the potential hazards of applying waste materials to land, for example, the sometimes excessive amounts of nitrate which may be present in the run-off waters from areas treated with animal wastes (Khaleel et. al., 1980).

1.2.2 Land disposal of sewage sludge

During the 1850's it was realised that the uncontrolled discharge of sewage into rivers was causing serious pollution and experimentation was begun at Rothamsted Experimental Station into the agricultural use of sewage. From that time until the turn of the century "sewage farming" was the accepted method of disposal. As cities grew larger the problem was tackled by the early attempts at sewage purification. This decreased the health hazards associated with sewage farming but led to the problem of sludge disposal. A Royal Commission was set up in 1917 to examine the value of sewage sludge and by this time the activated sludge process, which produced a sludge with a larger manurial value than other sludges had been introduced (D.O.E., 1977 b).

During the last sixty years there has been a gradual change in British agriculture away from livestock and towards more arable farming. This has both reduced the supply of farmyard manures, and increased the need for other organic fertilizers. In this respect sewage sludge has gone some way to make up the shortfall.

In 1942 Crowther and Bunting began the first major series of experiments looking at the role of organic manures in the supply of nutrients and contribution to soil structure. The results, (Bunting, 1963), showed that sludge treated soils never gave yields larger than those given by farmyard manure, and in some cases yields with sludge were little better than those on unmanured soils. This was thought to be due to the fixation of phosphorous by heavy metals in the sludge.

Since those early days a vast amount of literature has been produced in both Europe and the U.S.A. on the manurial value of sewage sludges, and their effects on the physical properties of soil. Details can be found elsewhere (D.O.E., 1977 a, b; Sabey, 1980) but, in general, it has been shown that increased yields and improvement in soil structure and water holding properties result from the addition of sewage sludge to soils.

More recently research into sludge disposal has concentrated on some adverse effects, such as the build up of toxic metals following repeated applications of contaminated sludges, and pollution caused by large amounts of nitrate being produced during the mineralisation of the sludge and its subsequent seepage into watercourses.

Interest in the possibility of contamination from heavy metals in sludges appears to have begun in the mid-1950's (Lunt, 1953), when it was noted that copper and zinc in sewage sludge could reduce crop growth. However, it is only in the last ten years that a large research effort has gone into the evaluation of the possible dangers to both crops and their consumers. Berrow and Webber (1972) studied sludge composition and concluded that zinc, copper and tin were the elements most likely to affect crop growth because the amounts found in sewage sludge were very much larger than those found in soil. Other elements, such as nickel, were found at very high levels in certain sludges. Page (1974) summarised these results and other similar ones from the U.S.A. in his review of heavy metal concerns associated with sludge disposal on land. He also gave data on soil and crop levels of selenium, mercury and arsenic.

More recently attention has been focused on elements which are toxic to man, which might accumulate in crops and so enter the food chain, e.g. cadmium and mercury. Cadmium is of particular concern since it can be sufficiently concentrated in the plant to become toxic to humans without being lethal to the plant. Cadmium content, therefore, is becoming a primary, or at least secondary guideline for the application rate of sludges, together with nitrogen and phosphorous contents (Sabey, 1980).

Webber (1980), in his review which examined the effects of metals from sewage sludge on crop growth and composition, noted that metal uptake was decreased at low pH values and high organic matter contents, and that phosphorous may reduce the uptake by plants of some heavy metals.

Plants differ in their tolerance to heavy metals, and while some plants are accumulators of heavy metals, most crops are not. Davis (1980) suggested the use of 'indicator' crops for assessing hazards associated with the accumulation of elements in the soil and presented tables of a number of crops showing their comparative efficiency for assimilating cadmium and zinc from soils. His data suggested which crops are most likely to be at risk when grown on contaminated soil and provide a basis for selecting crops for particular purposes.

A number of publications have discussed problems which arise due to overloading of nitrogen following sludge disposal to land (Kladivko and Nelson, 1979; Magdoff and Amadon, 1980; Sims and Boswell, 1980). The problems can be controlled by using yearly loading rates which supply, approximately, the nitrogen needs of the crop being grown. This can, however, be a major limitation if large quantities of sludge need to be disposed of. Sabey (1980), for example, suggests:

$$\text{Sludge application rate (tonnes/hectare)} = \frac{\text{Crop N requirement (kg/hectare)} - \text{Residual N (kg/hectare)}}{\text{kg available N/tonne sludge.}}$$

Risks from pathogens in sludge treated soil are relatively small if the sludge has been previously digested or composted, but the application of raw or untreated primary sludge on land is not recommended (Epstein and Chaney, 1978).

Guidelines relating to the 'safe' disposal of sewage sludge on land have been issued by a working party of the Department of the Environment (D.O.E., 1977 a). In recent years the cost of dewatering primary sewage sludge has meant that liquid sludge is often pumped straight on to land (D.O.E., 1977 b). This behaves somewhat differently to dried sludges but, since the sludge used in this investigation was dried and digested, a discussion of the problems associated with the land disposal of liquid sludges is considered to be outside the scope of this thesis.

1.2.3 Land disposal of composted domestic refuse

Composting involves the decomposition of organic matter by a mixed population of micro-organisms in a moist, warm, aerobic environment. Domestic refuse is gathered into heaps so that the heat of fermentation is conserved, the temperature rises and reaction rates are speeded up. Organic matter is converted to more stable materials such as humic acids, and carbon dioxide and water are evolved. Practical considerations of the composting process have been reviewed by Gray et.al.(1971).

Composting of municipal refuse on a large scale appears to have originated in Holland in 1929 with the setting up of N. V. Vuilafvoer Maatschappij (V A M), a utility company formed by the Dutch government for the disposal of city refuse, (Gray and Biddlestone, 1980). Unfortunately, the process has received less support in the U.K. and it was concluded by a Government Commission (H.M.S.O., 1954) that composted wastes could make little contribution to the supply of humus in the soil, nor could refuse help to any degree the problem of disposal of crude sewage sludge by co-composting. A further report from the Working Party on Refuse Disposal (H.M.S.O., 1971) considered that municipal composting should be regarded purely as a means of refuse disposal, and not as a method of improving soil fertility.

Partly as a result of this lack of government support, little research effort was directed towards consideration of the 'manurial' value of refuse composts until the 1960's, by which time indications from both Europe and the U.S.A. were that the potential was there. Kick (1960), for example, showed that composts of refuse and refuse/ sewage sludge mixtures were valuable for rapidly improving acid agricultural and forest soils poor in humus and nutrients, but for horticultural purposes the alkalinity and salt content were rather high. He found that the nutritional effect of 50 tonnes of 'fermented' compost was equivalent to about 25 kg nitrogen, 65 kg phosphate, 125 kg potassium and 2 - 2.5 tonnes calcium carbonate applied as commercial fertilizer. Only 10 - 12% of the nitrogen, 8 - 10% of the phosphate and 13 - 16% of the potassium were available to plants in the year of application and metal availability was low.

By 1966, Garner had published the results of experiments at Rothamsted comparing refuse compost with other organic manures such as farmyard manure, sewage sludges and sludge-straw composts. He showed that refuse compost almost always increased potato yields but to a lesser extent than animal manures. Residual effects from refuse composts were small.

In 1967, Gray and Biddlestone started a series of field experiments to investigate the effects of four municipal composts, two of which had been composted with sewage sludge and two alone, on the growth of dwarf-beans, lettuce, potatoes and spinach-beet. At this time little was known about the trace metal content of municipal refuse but in view of the many constituents of town refuse, it was anticipated that a wide range of trace metals would be present. High dressing rates of compost were used to try to produce toxic effects in a short-term trial.

Results of these experiments (Gray and Biddlestone, 1980) showed that the composts did contain appreciable amounts of some heavy metals and that with heavy applications, the extractable concentrations of these elements were greatly increased. However, the availability of these metals was low. They noted several factors which restricted trace metal uptake in the trials, viz. pH, organic matter and probably calcium ions in the compost, and expressed concern that if the pH and organic matter fell rapidly later on, increased metal availability would occur.

Rothwell and Hortensteine (1969) compared the rates of microbial activity, carbon dioxide evolution and nitrification of incubated refuse compost, sewage sludge, chicken manure and cow manure, each mixed with Arredondo fine sand. The fungal population increased, levelled off, and then began a second slow but steady increase after 15 - 20 days. The bacterial population multiplied rapidly within the first few days, then decreased rapidly to a steady level. Cumulative carbon dioxide evolved was increased by all the organic materials, but the rate of increase was greatest with the materials highest in nitrogen (sewage sludge and chicken manure). Very little nitrification was measured for the refuse compost, and when mixed with cow manure, nitrification was actually reduced by the compost.

A series of field experiments was conducted by Mays et. al. (1973) to evaluate the effects of heavy annual municipal compost application on crop yields and on the physical and chemical properties of soil under field conditions. The compost (made from municipal refuse and sewage sludge) increased yields of bermuda grass at application rates of up to 80 tonnes/hectare, of sorghum at rates up to 143 tonnes/hectare, and of corn at rates up to 112 tonnes/hectare. The application of 82 tonnes/hectare of compost produced yields equivalent to those produced with 90 kg/hectare of fertiliser-nitrogen. Moisture holding capacity of the soil was increased, and bulk density and compression strength were decreased by incorporation of the compost over a two-year period. The extractable soil zinc content increased dramatically, reaching 490 kg/hectare with the highest treatment of 327 tonnes/hectare of compost; although no plant toxicity symptoms were observed, zinc uptake by sorghum was doubled at the highest compost treatment.

Purves and Mackenzie (1973), investigated the effects of applications of municipal compost on the uptake of copper, zinc and boron by garden vegetables. Their analyses indicated that the compost can contain up to 100 times as much available boron and copper and 300 times as much available zinc and lead as uncontaminated rural soils in Scotland. In a series of field trials using refuse composts at rates of 25, 50 and 100 tonnes/hectare available soil copper, zinc and boron were increased. Enhanced uptake of copper, zinc and boron was noticed for lettuces and beans; zinc and boron with potatoes; and boron with peas. No significant increases in any of these elements was obtained in two experiments with cabbages. Compost treatment resulted in severe phytotoxic effects on beans, but yield increased with peas and potatoes.

The physical and chemical effects of municipal compost fortified with sewage sludge and nitrogen-fertilizer on soil and corn plants were investigated by Duggan and Wiles (1976). Their results showed that corn grain yields were increased and soil physical properties improved with annual compost application rates of 200 tons/acre (500 tonnes/hectare) for 5 years. Corn showed positive yield responses even after compost applications were terminated. No metal toxicity was noted but the authors suggest that less tolerant crops could be adversely affected.

It would seem, therefore, that municipal compost applications do increase plant yields but that equivalent yield responses can be obtained more cheaply with mineral fertilizers. However, refuse compost can improve soil physical properties such as structure and moisture retention. Thus, in the words of Tietjen and Hart (1969) composts are likely to be "utilized as a soil conditioner, and not as a fertilizer. For the foreseeable future, compost will not be used beneficially and economically in general agriculture".

The future of municipal composts may well lie in an alternative and potentially valuable use in the reclamation of poor soils. In the Netherlands, where the composting of municipal refuse began on a large scale, there has been a development in usage since the 1920's from agricultural, through horticultural to recreational. This was initially because the price for agriculture was too high, but later became due to limitations from heavy metal content. Nowadays, 90% of the municipal waste compost is applied to soils used for recreational purposes (de Haan, 1981). The same holds true in Canada where refuse/sewage sludge composts from a new Resource Recovery Plant are being considered as soil amendments for gravel pit reclamation (non-food), top soil supplements, landscaping in parks and container pot growing media. Application of the compost to food-growing lands is not envisaged (P. J. Provias, Ministry of Environment, Ontario - Personal Communication).

1.2.4 Land disposal of fresh domestic refuse

The advantages of composting domestic refuse are that it stabilises the material, reduces the carbon:nitrogen ratio of the refuse and helps eliminate the health hazards associated with fresh refuse (Cottrell, 1975). However, the disadvantages are that the process is time consuming, costly and may concentrate trace elements to toxic levels.

There have been few reports dealing with the direct application of solid domestic refuse to the soil, and most of these have considered non-sorted, shredded material. Hart *et. al.* (1970) incorporated coarsely ground unsorted municipal refuse into surface soil at rates of 50 to 400 tons dry weight per acre (125.5 to 1000 tonnes/hectare). They referred to the practice as "garbage farming" and were considering it mainly as a method of waste disposal rather than for soil or crop improvement. Nitrogen fertilizer was added to reduce the C:N ratio of the refuse. After one year an unidentifiable organic residue plus glass, metals and plastic remained. The major problem was found to be a physical one, that of mixing one foot of topsoil with up to 18 inches of refuse (at the top dosage) and still having sufficient topsoil to prevent paper blowing about.

The effect of a single application (280 or 560 tonnes/hectare) of coarsely ground solid waste to a soil and the growth of corn and rye was investigated by Webber and King (1974). They found increased corn grain yields on refuse treated soils and although crop concentrations of zinc, cadmium and lead were increased they did not reach toxic levels. Decomposition of the refuse after twelve months had proceeded to the point where sufficient nitrogen was available for the crop.

King *et.al.*(1974), considered the feasibility of using land for both waste disposal and crop production by applying pulverised municipal refuse alone, and in combination with liquid sewage sludge to land, which was later planted with rye, and then corn. Dosage rates were 188 tonnes dry refuse per hectare, with or without 2.3 cm of liquid sewage sludge; and a double rate of 376 tonnes/hectare with 4.6 cm of sludge. Yields of rye were not affected by the refuse, but corn yields were increased at the lower rate although they fell back at the double dosage. Zinc and cadmium uptake by the rye was increased with both treatments, and copper uptake increased slightly with the sewage treatments, although no toxicity symptoms were observed. Lead uptake was unaffected by treatment. After two years approximately 80% of the paper had decomposed suggesting that even with an initially high C:N ratio of 65:1 mineralisation of the nitrogen had proceeded, possibly aided by the additional nitrogen supplied by the soil. Nitrogen content of the crops were not significantly affected by the waste treatments although soil nitrate was increased.

A lysimeter study was conducted as a corollary to the field experiments (King _et_. _al_., 1977). This showed that levels of zinc, copper and cadmium were increased by treatments containing refuse and that the double rate treatment resulted in corn stover cadmium levels greater than the acceptable level for foodstuffs of 0.5 ppm (stover is defined as the total plant less the grain). Nitrate-nitrogen losses from the refuse treatments were identical to those from the control indicating that addition of refuse alone did not cause immobilisation of nitrogen. The effect of manipulating the C:N ratios of the treatments was inconclusive. The chemical oxygen demand of the leachate was high ($>$ 1000 ppm) for all waste treatments but the nitrate leaching loss was low, particularly for the double rate treatment.

Webber and Doyle (1975) reviewed some of the problems in using waste amended soils for crop production. They noted that soil physical conditions were improved by materials such as organic manures, sewage sludges, refuse, and wastewaters, and state that utilisation of organic materials in the applied waste by micro-organisms results in the formation of mucilaginous gums of a polysaccharide nature. These gums play an important part in the aggregation of soil particles. The formation of a stable soil structure, good drainage and aeration are obviously beneficial to plant growth. The authors also pointed out that soil erosion may be reduced by the use of shredded refuse, but that one of the main problems associated with the use of shredded refuse lies in producing adequate incorporation into the soil and sufficient compaction of the surface layer in seedbed preparation. It was shown that essential plant nutrients were provided by the wastes, but in-depth research was recommended to determine safe dietary levels of heavy metals from food produced on waste amended soils.

Cottrell (1975) applied solid municipal waste to a sandy soil at rates of 0, 100, 200 and 400 tons/acre (0, 251, 502, 1004 tonnes/hectare), equating to 0, 67, 133, 276 tons of dry matter per acre. Liquid sewage sludge was then added at 55 gallons per ton of solid waste. The plots were irrigated for two crop seasons and ammonium-sulphate fertilizer was added. It was found that the solid waste decomposed rapidly, with only plastic, rusted metal, and rubber remaining on the plots by the end of the first growing season. Soil bulk density decreased and organic matter content and moisture retention increased during the first year. Effective wind erosion control was obtained by the solid waste applications. Crop yields were increased at all but the highest dosage rate, and these increases were further augmented by the addition of low amounts of nitrogen fertilizer. High rates of fertilization caused increased acidity and consequent mobility of zinc and manganese. The main problem with regard to toxicity arose from high levels of boron which increased up to 60-fold with waste addition. The boron was soluble and levels decreased rapidly during the first year, but remained greater than 2 ppm at the highest dosage rate.

Two reviews concerned with the land application of wastes have been published, (Volk, 1976; Stewart and Webber, 1976). The former dealt with the effects of "trash" and "garbage" application to agricultural lands on soil physical properties, elemental uptake and crop production. The latter was a more general consideration of the capability of the soil to accept waste.

Volk concluded that the addition of shredded or composted municipal refuse can be achieved wihout loss of crop productivity, but he remarked that ideally the recyclable components such as glass, metal and plastics should be removed since these could cause equipment problems as well as reducing the aesthetic appearance of the treated area. He considered that most composts, and all fresh wastes require additional nitrogen for maximum crop production, but that those nutrients which are immobilised in the organic fractions may be released slowly over time. Problems of boron toxicity can be alleviated by the use of tolerant crops and controlled irrigation.

According to Stewart and Webber (1976) "soil is a natural acceptor of waste, and its microbial population can effectively decompose and purify a wide range of substances". However, they also recognised that soils have a finite capacity for wastes and the factors which govern this must be fully understood for effective management. Effects on decomposition of physical, chemical and biological characteristics of the soil, and the nature and amounts of wastes were considered in detail along with the influence of C:N ratio. Reference was also made to beneficial effects in terms of increased levels and availability of plant nutrients such as phosphorous and potassium, although it was pointed out that nitrogen availability in some cases is reduced. Improvement in soil physical properties from the addition of wastes was also discussed. Potential hazards considered included excessive production of nitrate, high levels of soluble salts and accumulation of heavy metals.

Although many of the reports indicate conflicting results with respect to crop yields, nitrogen mobilisation and metal availability, the beneficial effects of waste additions to soil physical properties seem to have been fairly well established. Webber (1978) conducted a series of field experiments to determine the effects of applying as much as 376 tonnes/hectare of raw, non-segregated shredded refuse with 4.6 cm of digested sewage sludge on soil carbon and nitrogen, soil physical properties and corn yield. He found that bulk density decreased, aggregate stability, soil carbon and nitrogen increased, but available water capacity was not affected by waste addition. Corn grain yields were not found to be related to kinds or amounts of wastes added (c.f.King et. al., 1974).

Experiments were carried out in 1980 by the Ministry of Agriculture's Agricultural Development & Advisory Service at a refuse pulverisation plant at High Wycombe, Buckinghamshire. They attempted to assess the potential of unsorted, pulverised refuse for the growth of vegetable crops; no results have been published. Cadmium levels three times the normal levels for foodstuffs were found and boron phytotoxicity was a problem. By mixing the refuse with soil yields were increased and the boron levels were reduced from 9 or 10 μg/g to 4 μg/g. (personal communication)

Khaleel *et. al.* (1981), provided an overview of the changes in soil physical properties due to organic waste applications. Based on data from twelve sources, twenty one soil types, seven waste types and eight crop types, a linear regression analysis of observed increases in soil organic carbon as a result of waste applications on percentage reduction in bulk density produced a highly significant relationship ($r^2 = 0.69$). Eighty percent of the observed variations in percentage increases in water holding capacity (at both 0.05 and 15 bars suction pressure), could be attributed to variations in soil texture and organic carbon increases.

In recent years it would appear that interest in soil disposal of fresh domestic waste has waned slightly, due mainly to problems of materials handling, the relatively low manurial value, and the build up of toxic elements. These problems may be significant where disposal is on crop-producing land, but the criteria for feasibility change if the refuse is used for reclamation purposes on derelict land. Manurial value becomes less important and indeed a lower value may be advantageous in terms of maintenance costs (e.g. mowing costs kept to a minimum). The critical levels for toxic elements are likely to be higher where after-use is for amenity purposes rather than agricultural.

Refuse has been used in the past to reclaim areas such as old mine sites. Dobson and Wilson (1973) found that decomposition of fresh domestic refuse actually increased when mixed with spoil material from strip-mined areas. The refuse apparently contained a sufficient number and variety of micro-organisms to initiate and continue decomposition, and the low pH of the spoils (3.7 and 4.4) did not affect decomposition rates. As with composted refuse, it seems likely that the potential value of fresh refuse as a soil amendment will be in recreational rather than agricultural terms.

A pilot study (Davis and Mills, 1977) established the biochemical nature of the separated "putrescible" fraction of the refuse from Doncaster (Table 2). This composition will tend to vary at different times of the year. The work showed that the ethanol/water soluble fraction decreased by 75% and the hemicellulose fraction decreased by 50% in the first four months of biodegradation following incorporation into soil. The breakdown of cellulose and lignin was slower but it

	%
Ethanol/water solubles	3.85
Hemicellulose fraction	8.72
Cellulose fraction	57.07
Lignin fraction	7.17
Soluble extracts	76.80
Insoluble residue	23.20
Humic acids	4.07

Table 2 Percentage composition of Doncaster separates (dry weight basis).
(from Davis and Mills, 1977)

was noted that the paper initially present in the waste was not visible after four months. The humic acid content of the soil increased from 2.4 to 3.7% over the same period indicating that the refuse fractions would degrade fairly rapidly in the soil, and that the material was of potential value as a soil additive.

1.3 Scope and aims

In their discussions of the limitations of presently available data, Khaleel et. al. (1981) state that:

> " Standardisation of data is needed so that research results from different locations can be compared .... most of the data were collected after one or two years of waste application with little or no information on the changes of various properties over time to that point .... presently no mathematical algorithms are available to describe and estimate the fate of added carbon in soil waste systems .... similar models (to those available for the decomposition of plant residues) are needed to describe the decomposition of organic wastes as a function of time ....".

These considerations formed the basis for the specific aims of this investigation; the details are outlined below:-

(1) To follow in detail the biodegradation of the components of the refuse separates over time, and thus to establish the rates and patterns of breakdown, and express them as mathematical functions.

(2) To assess the effect of mixing the refuse separates with other waste materials, namely sewage sludge and colliery shale on breakdown and stabilisation. Sewage sludge is rich in nitrogen and since the refuse is likely to be highly carbonaceous, it was suggested that mixing the two together would result in a carbon:nitrogen ratio which would allow sufficient nitrogen mineralisation for plant growth, but prevent excessive leaching loss of nitrate-nitrogen.

(3) To establish optimal dosage rates for refuse separates addition to soil since excessive loading could result in oxygen depletion, slower decomposition, and accumulation of odorous, phytotoxic end products that could reduce soil productivity (Doran et. al., 1977).

(4) To assess the effect of repeated application of refuse separates upon the receiving soil.

(5) To consider the establishment and maintenance of healthy plant life on treated soil, including an investigation into the conditioning effects of the refuse separates on soil physical properties and the release and availability of plant nutrients.

(6) To consider any possible pollution hazards or toxic metal build-up which may be caused by the incorporation of the refuse separates; and to explore the potential for the release of heavy metals as the wastes undergo decomposition.

It is proposed that if these points can be adequately covered in field and laboratory experiments an understanding of the behaviour of these waste fractions in the soil will be achieved. From this basis the potential value of the incorporation of similar materials into soils with different characteristics can be inferred, and proposals for future management suggested.

1.4 Layout of the thesis

A review of the literature concerned with the application of wastes to land is presented in Chapter 1. This is followed by a section detailing the aims and objectives of the project.

Chapter 2 outlines the investigation, the materials used and the procedures followed in the laboratory and field. The investigative work fell into two main parts:

(1) The effect of using sewage sludge as an additive on the biodegradation of the refuse separates, soil conditions and plant growth (including some ecological work).

(2) The effect of varying the application rate on biodegradation, soil conditions and plant growth.

These two form the basis of Chapters 3 and 4 in which the results of the field and laboratory experiments are presented and discussed. A short concluding section is included at the end of each set of results.

Finally, in Chapter 5, general conclusions are drawn from the preceding Chapters, some recommendations for the management of disposal sites are made and suggestions for further investigative work presented.

CHAPTER 2

MATERIALS AND METHODS

2.1 Materials

2.1.1 Doncaster refuse separates

The Doncaster Waste Treatment Plant, officially opened in December, 1979, aims to recover many of the valuable materials which are lost when conventional methods of tipping are employed. Waste arriving at the plant is initially size-screened in a rotating trommel and the fine fraction, nominally that which is less than 40 mm, is then fed to a secondary vibrating screen, which eliminates dirt and ashes (material less than 15 mm), and leaves a mixture suitable for processing in the glass recovery circuit (Fig.1). The initial stage in this circuit involves the primary stoner decks from which the rejects, or "less dense fraction" (at present constituting up to 45% of the incoming refuse) comprise the highly putrescible material investigated in this study.

This highly heterogeneous material consists largely of vegetable matter (peelings and other discards) as well as paper and textile scraps, ash, fats and some plastic material, small stones and coal clinker. Chemical properties are tabulated in Section 3.1.3 (Table 4).

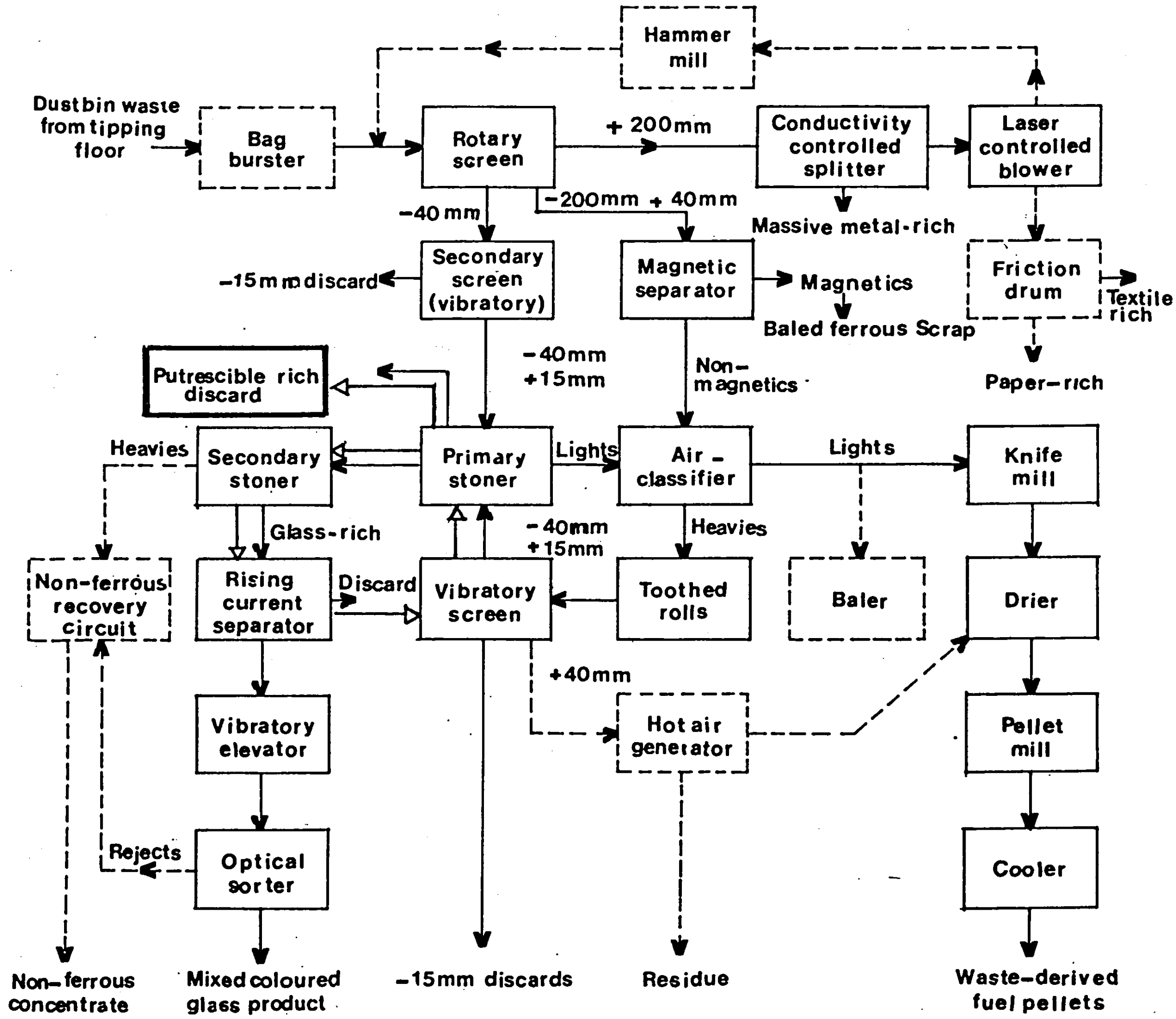
The refuse separates used in this investigation differ from those which are normally bulked with sewage sludge in composting plants in that they contain a lower proportion of carbonaceous material such as newspaper and textiles and more of the high putrescible vegetable matter described above.

2.1.2 Sewage sludge

The dried, digested sludge was used as supplied from Lund Wood Sewage Works, Barnsley, South Yorkshire. Raw sludge from the primary settlement tanks had been dewatered and anaerobically digested at 32^{o}C. for thirty days: the digested sludge was subsequently dried to a cake containing approximately 25% moisture prior to supply. Properties of the sludge are presented in Table 4, Section 3.1.3.

2.1.3 Colliery shale

The shale (pH 3.5), originated from the Barnsley Main coal seam



The equipment indicated in a broken line is not included in the current Doncaster flowsheet but could be included at a later date.

FIG.1

Flowsheet describing the derivation of the putrescible-rich fraction of the Doncaster refuse (white arrows)
(Courtesy of S.Y.C.C.)

and was used as supplied by South Yorkshire County Council.

2.1.4 Field site

Field studies were carried out on a site measuring one third of a hectare, provided by South Yorkshire County Council at Hellaby near Rotherham. The site, which had previously been unused for many years, was covered by an imported topsoil to a depth of approximately 0.5 m. It is described below, using the terminology and definitions of the Soil Survey of England and Wales (Hodgson, 1976).

Grid Reference	:	506940 (1:50,000 series.O.S.Sheet No.11, Sheffield & Doncaster)
Elevation above O.D.	:	360 - 370 ft.
Relief	:	Gently sloping (3 - 5^{o}). Disturbed site.
Aspect	:	South-facing.
Slope form	:	Straight.
Land use	:	Experimental plots.
Profile Description	:	(see also Plate 1)

L	0 - 0.5 cm	Litter layer: undecomposed fresh litter, predominant broad-leaved grasses.
F - H	0.5 - 2 cm	Partly/well decomposed layer; abundant roots; diffuse boundary to -
A(p)	2 - 15 cm	Colour 7.5 YR 3/4. Dark brown with a few fine mottles; clay/clay loam where material is incorporated by fauna; few small stones; small/medium angular blocky structures with weak strength; this boundary disturbed by washing down of organic matter and faunal incorporation; relatively porous; abundant roots; gradual change to -
B	15 - 45 cm	7.5 YR 3/4. Dark brown with abundant large mottles - 5 YR 4/8 - reddish-brown indicating intermittant waterlogging; silty clay loam; few small stones; coarse

		angular blocky, un-natural structures; more compact horizon with weaker structures than A(p) and few small roots, rooting almost restricted to around ped faces.
BC	45 cm	Buried profile - dry, well-structured shaley subsoil. 10 YR 3/2 Brownish black indicating relatively high organic matter content of "topsoil" prior to burial; haphazard un-natural structures with a relatively high porosity; abundant old but few new roots.

2.1.5 Plant species

Perennial ryegrass, *Lolium perenne* var. *Melle*, being a popular landscaping grass, was used in field growth trials. Barley, *Hordeum vulgare* var. *Ark Royal*, was chosen for use in the laboratory growth experiments mainly due to its capacity for rapid growth.

2.1.6 Climatological data

Monthly temperature and rainfall figures were supplied by the meteorological station at Finningley, South Yorkshire (height 10 m.) to cover the period February, 1979, to December 1981. The data is presented in Appendix A.

2.2 Experimental procedures and statistical analysis

Some limitations on the methodology employed, in both the laboratory and field, were imposed by the availability of equipment. Analytical procedures were chosen as being as sensitive and specific as possible in this type of investigation without being too specific to be applied to the fields of solid waste management and soil biochemistry which both involve complex and heterogeneous multi-factorial systems.

The two main areas of investigation were:

(a) the effect of additives (colliery shale and sewage sludge) and

(b) the effect of dosage rate on refuse decomposition, soil conditions and plant growth.

PLATE 1. Field site : soil profile pit.

Each of the two involved both field and laboratory experimentation.

2.2.1 Effect of additives

In previous studies on refuse application to soils (King et.al., 1974; 1977; Duggan and Wiles, 1976; Webber, 1978) and composting of refuse (Stead, 1978; Golueke et.al., 1980), it was found that the carbon:nitrogen ratio of unsorted refuse was higher than the 30-35:1 considered optimum for biodegradation (Gray et.al., 1971; Poincelot, 1975; Mote and Griffis, 1980). Sewage sludge contains up to 19% protein compared with only 2.6% in domestic refuse (Rees, 1980) and has been widely used as a nitrogen source to aid the breakdown of organic materials containing a high proportion of carbonaceous material. Dried, digested sewage sludge was, therefore, combined with the Doncaster refuse separates to investigate the necessity of nitrogen supplementation in aiding the breakdown of the refuse and allowing sufficient nitrogen mineralisation for plant growth.

Colliery shale was initially included in the investigation at the request of the South Yorkshire County Council (S.Y.C.C.) since it also was presenting disposal problems.

Field decomposition trials

The waste materials were hand dug into the top 20 cm of the cover soil in plots measuring 2 x 2 m. After consultation with S.Y.C.C. an equivalent dosage rate of approximately 20 tonnes/hectare was decided upon.

The treatments were:-

(1) Untreated soil cover (as control).

(2) Soil plus refuse separates.

(3) Soil plus refuse, with a reapplication at the same dosage in December, 1979, (after 10 months).

(4) Soil plus refuse plus colliery shale.

(5) Soil plus colliery shale.

(6) Soil plus refuse plus sewage sludge.

(7) Soil plus sewage sludge.

(8) Soil plus refuse, plus colliery shale plus sewage sludge.

Treatments 4 and 5 were later abandoned after initial results indicated extreme difficulty in analysis and interpretation of results (many workers have found similar difficulties in the analysis of colliery shale - Dr P Williams - Personal Communication). Treatment 8 was not included in the trace element analyses.

In view of the heterogeneity of the Doncaster refuse, coupled with field site variability, suitable design of the field experiment was important. A randomised block design was used with a 10-fold replication of the 8 treatments (Plate 2). This type of design aims to reduce experimental error by high replication of each treatment, random allocation of treatments to plots, and blocking. Local control such as blocking requires the allocation of plots to "blocks" so that units within a block are relatively homogeneous, and heterogeneity between the blocks is maximised. Bias due to site variation is thereby reduced.

Representative sampling is considered to be critical in this kind of investigation and it was decided to take three samples from each individual plot and pool them for laboratory analysis. An initial experiment (Table 3) indicated that intra-plot variability initially was high but had decreased by 50% after 12 months biodegradation.

Adverse weather conditions in spring 1978 prevented the completion of treatments 4, 5, 6, 7 and 8 until the end of April although treatments 1, 2 and 3 had been completed in early February. Samples were taken in May, 1979, (one month after incorporation of treatments 4 - 8, but four months after that for treatments 2 and 3) and thereafter at approximate six monthly intervals. A zero time sample was also included in the toxic metal analyses.

Ecological aspects

The plots were colonised in the summer of 1979 with ruderal weeds, some indigenous and some introduced in the waste material. Ecological information on the diversity and competitive ability of the ground cover was provided using a point quadrat method to estimate percentage

PLATE 2. Field trials site showing the randomised block design with a 10 - fold replication of the eight treatments.
Wooden pegs mark the corners of the 2 x 2 m plots.

Sample			% Organic Carbon $\bar{x}$	σ	v
March	'79	A_2	6.50	1.10	16.92
		F_5	5.56	1.55	27.88
		I_8	4.94	1.80	36.44
Total			5.64	1.67	29.61
November	'79	C_1	3.81	0.52	13.75
		F_5	3.51	0.53	15.14
		I_8	3.49	0.65	18.76
Total			3.61	0.59	16.43
April	'80	C_1	2.54	0.21	8.10
		F_5	3.16	0.38	12.10
		I_8	3.13	0.48	15.20
Total			2.94	0.47	15.93

Table 3. Reduction in variability in soil organic carbon in the first 12 months following refuse incorporation. All samples taken from soil plus refuse plots (Treatment 2) See Appendix E.

($\bar{x}$ = mean of 10 replicates, σ = standard deviation,
v = coefficient of variation, i.e. $\frac{\sigma}{\bar{x}} \times 100$.)

cover of the species present (Chapman, 1976). The occurrence of the first species touched by each of 50 probes on a point quadrat was recorded and the values multiplied by two to give percentage cover. Six replicate plots of each treatment were studied. Quantitative information on the productivity of the plots after eighteen months was obtained from the dry weight biomass (oven dried at 80°C.) of plant material occurring within a 25 cm^2 quadrat.

Plant growth experiment

A controlled experiment using barley (*Hordeum vulgare* var. *Ark Royal*) was carried out in the growth room to compare the productivity of the different treatments. The treatments used in the field trials were simplified in the laboratory growth experiment to four:-

(1) Soil control.

(2) Soil and refuse separates.

(3) Soil and sewage sludge.

(4) Soil, refuse and sewage sludge.

The materials were mixed in proportions equivalent to those in the field plots, and placed in 13 cm. diameter plant pots. Each treatment was replicated nine times. The pots were each planted with three germinated seedlings and arranged in a completely randomised design in the growth cabinet. Day and night length were controlled using fluorescent strip lighting to twelve hours each and temperature maintained at about 12°C. at night and 20°C. during the day. Relative humidity was kept around 45%.

Five shoot samples were taken at random from each treatment at each of five harvest times during the ten-week experiment. The samples were oven-dried at 80°C. and weighed. Inorganic nitrogen content of the soil was measured (Section 2.2.3) at four of the harvest times and then seven weeks after the end of the experiment.

2.2.2 Effect of dosage rate

Soils have a finite capacity for waste assimilation. This capacity will be determined by the volume of soil to which waste is added, the biological, chemical and physical properties of the soil, and the nature and composition of the wastes. In view of the unique nature of the Doncaster putrescibles it was felt necessary to ascertain optimum rates of dosage for the soil disposal of this material.

In the past low dosages of organic manures and sludges, (less than 25 tonnes/hectare), were applied on crop and recreational land to correct nutritional deficiencies. More recently, disposal of over-abundant waste products has become the objective, and application rates have been increased (Giordano and Mays, 1977 b). In this investigation, dosages of up to 65 tonnes/hectare were applied to assess the effect of dosage on biodegradation rate, plant growth; and possible deleterious effects such as high levels of toxic metal and immobilisation of plant available nitrogen. Dosages above 65 tonnes/hectare are unlikely to be applied in practice, due mainly to the physical problems of incorporation.

Field decomposition trials

Five trial plots, measuring 2 x 5 m, were set up on site at Hellaby in July, 1980. One plot of unamended soil acted as the control, and the others were treated with refuse to dosages of 20 tonnes/hectare, (to compare with the additives treatment), 32.5 tonnes/hectare, 47.5 tonnes/hectare, and 65 tonnes/hectare. To facilitate incorporation of the refuse into the soil a mechanical rotovator was used. At each time of sampling six replicates were taken from each of the treatments. To gain an understanding of the early stages of biodegradation samples were taken one week after incorporation and then after a further five weeks. Thereafter six-monthly sampling was considered satisfactory.

Plant growth experiments

The plots were seeded with perennial ryegrass (*Lolium perenne* var. *Melle*) one week after initial incorporation (July, 1980). The grass occurring within a 25 cm^2 quadrat (replicated by three) was collected at monthly intervals up to December, 1980, and the dry weights recorded. Numerical analysis of the data will be discussed later. Plant tissue was analysed for kjeldahl nitrogen at each sample time.

A controlled growth experiment was conducted in the growth room, using barley as the test plant. A range of soil/refuse mixtures was placed in 13 cm. diameter plant pots. One kilogram of air dried site soil filled the pots and provided the control treatment. The other five treatments contained mixtures in proportions similar to those in the field trials (i.e. 20, 40, 60, 80 and 100% by volume of air-dried ground refuse). Each treatment was replicated nine times, the pots planted with three germinated seedlings and placed in a completely randomised design in the growth cabinet. The same controlled condition as outlined for the additives plant growth experiment were used. Four plant samples were taken at each of six harvest times during the 10-week experiment and their dry weight and kjeldahl nitrogen recorded. Ammonium and nitrate-nitrogen content of the soil was measured at the start of the experiment and at four harvest times.

Incubation experiments

Amounts of mineralisable nitrogen in samples collected from the field plots in October, 1980, and October, 1981, were estimated in laboratory incubation experiments. Five millilitres of deionised water was added to air-dried samples (10 g) which were then incubated for four weeks in glass beakers plugged with non-absorbent cotton wool, in the dark at 27^{o}C. (Allen, 1974). Evaporation losses were replaced by addition of deionised water every 2 days. After 0 and 28 days the ammonium - and nitrate - nitrogen content of triplicate samples was determined, and the difference between the two recorded as the amount mineralised.

2.2.3 Soil analysis

Soil samples collected from both sets of trial plots on site were analysed on return to the laboratory for percentage moisture and pH. The samples were then air-dried overnight, homogenised in a high-speed grinder, dried to constant weight at 80^{o}C. and stored at 4^{o}C. for subsequent analysis.

Percentage moisture and water holding capacity

Samples were oven-dried at 80^{o}C to constant weight and percentage moisture estimated gravimetrically. Plant available water is considered to be the volume retained between field capacity and permanent wilting point (conventionally taken at moisture tensions of

0.05 and 15 bars respectively). The moisture content at 0.05 bars was obtained by equilibrating samples on a sand suction table and then measuring percentage moisture gravimetrically. The same procedure, but using a pressure plate apparatus, was followed to obtain moisture content held at 15 bars (Avery and Bascomb, 1974).

pH analysis

The pH of the samples was measured using a Pye model 79 glass electrode pH meter after adding distilled water to the samples in an approximate 1:1 ratio, mixing and allowing to equilibrate for 20 seconds. The values obtained in the laboratory correspond well with measures made on site with a W.P.A. field pH meter.

Organic carbon

The loss-on-ignition estimation of organic carbon (Allen, 1974) was used for the oven-dried field samples. This method was decided upon after a series of comparisons with the widely used Walkley-Black dichromate oxidation method for estimation of oxidisable carbon (Hesse, 1971). Ball (1964) found that results for loss-on-ignition were substantially less variable than those obtained with the Walkley-Black technique.

The comparison carried out on 30 samples collected in May, 1980, showed that the correlation coefficient between the results from the two methods was 0.8 at a 99% level of significance. Considering the highly variable nature of the samples, the loss-on-ignition method was considered more suitable since a much larger sample could be accommodated. For the laboratory trials in view of the smaller sample size, and the smaller number of samples, the Walkley-Black method was considered satisfactory.

Organic nitrogen

Percentage nitrogen was measured in oven-dried samples by the kjeldahl semi-micro determination (Allen, 1974). Percentage nitrogen is calculated from the equation -

$$\% \, N_2 = \frac{T \text{ (ml)} \times \text{solution volume (ml)}}{10^2 \times \text{aliquot(ml)} \times \text{sample weight(g)}}$$

where T (ml) = volume of M/140 HCl required for the titration following the digestion of the samples and the distillation of the ammonia thus formed. This method strictly only estimates total organic nitrogen and ammonium nitrogen. Nitrates are not included. In this thesis it will be termed "organic nitrogen".

Inorganic nitrogen

Fresh samples were extracted with 2 M KCl for one hour after which the residue was removed by filtration. A steam distillation method using magnesium oxide and Devarda's alloy was used to measure ammonium and nitrate-nitrogen in the filtrate (Bremner, 1965). Results were expressed on a dry weight basis.

Proximate Analysis

The proximate analysis of the soil samples was based upon methods of Chang (1967) and Mills (1973). Initial experiments indicated that although the method for estimation of cellulosic material was accurate when measuring known amounts of cellulose added to pure sand, somewhat conflicting results occurred when the method was applied to the soil and refuse mixtures. Extraction procedures, in fact, appeared to be extracting mineral as well as cellulosic material from the soil. It was decided, therefore, to base analysis of the biodegradation processes upon the measurement of ethanol/water soluble materials, mineralisation of organic nitrogen to ammonium - and nitrate - nitrogen, humic acid estimation, and carbon to nitrogen ratios. By using these parameters the biodegradation of high molecular weight insoluble polymers such as cellulose and lignin can be followed indirectly by noting their microbial transformation into soluble materials.

Ethanol/water soluble fraction

Oven-dried samples were extracted in a soxhlet apparatus for 6 hours with an 80% (v/v) ethanol/water mixture. The extracted material was oven-dried at 80^{o}C. and then placed in a vacuum dessicator for 10 minutes before re-weighing to calculate the percentage extracted material. This fraction contains reducing and non-reducing sugars, glucosides, inorganic nitrogen, amide and amino nitrogen, basic cyclic and alkaloidal nitrogen, oils and lipids, chlorophyll, carotene, xanthophyll, flavones and tannins, anthroquinones, gums and resins (Davis and Mills, 1977).

Humic acid analysis

The end product of organic matter decomposition in the soil is humus, and the portion of this which is soluble in aqueous alkali and precipitated by acidification of the alkaline extract is termed "humic acid". There appears to be confusion of the actual biochemical nature and composition of humic acids but it is clear that this complex mixture provides a "storehouse" of nutrients for microbial and plant growth which are released during the slow breakdown of the humus itself.

Humic acids were extracted with 0.1 M NaOH by shaking in stoppered bottles for 18 h. The sediment was removed by high speed centrifugation and the supernatant acidified to pH 2.0 with 10% (v/v) HCl in order to precipitate the humic acids. These were collected on pre-dried, pre-weighed filter papers, washed, and allowed to air-dry overnight before placing in a vacuum dessicator for 2 hours prior to weighing (Sharp and Mills, 1973).

Trace elements

Copper, zinc, lead, cadmium and boron were selected for measurement as being those elements with plentiful sources in domestic refuse, and which could cause toxicity problems if present in large amounts.

Oven-dried, 10 g. samples of soil/refuse mixtures were extracted with 100 ml 0.1 M HCl for 1 h. and analysed for copper, zinc, lead and cadmium by atomic absorption spectrophotometry (Allen et.al., 1974). This extractant was decided upon after early results using 2.5% acetic acid yielded very low levels. Neuhauser and Hartenstein (1980) suggest that 0.1 M HCl is a more suitable extractant of metals in sludges than 2.5% acetic acid although it should be remembered that the pH of 0.1 M HCl is unlikely to be achieved realistically in the soil. They found that this extractant, in a 1:10 ratio, yielded approximately one-third of the total amounts of cadmium and zinc, but considerably lower proportions of the total lead and copper. Total metal concentrations were not measured since these are considered more relevant where a build-up of contamination could result from repeated dressings of toxic material. It is unlikely that in practice there will be more than two applications of the Doncaster putrescibles.

Plant materials after being thoroughly washed in tap and then distilled water were ashed overnight in a muffle furnace at 300°C., the temperature being deliberately kept low to prevent losses of cadmium and lead. Metals were then extracted using 6 M HCl and concentrated nitric acid and then diluted to give a final ratio of 1:25. Zinc, copper and lead were quantified using flame atomic absorption spectrophotometry. The much lower levels of cadmium were analysed using a flameless operation in a controlled temperature furnace atomiser, with ignition at 1750°C. During all stages acid-washed glassware was used, and blanks and standards contained acid in concentration equivalent to the samples. None of the four elements analysed are considered to have any serious problems of interference.

Levels of water-extractable boron in the soil samples were analysed spectrophotometrically using curcumin as the colouring reagent. No borosilicate glassware was used in this analysis. Nitrates, which interfere with the spectrophotometric analysis were removed by preliminary evaporation of the samples with calcium hydroxide.

Plant tissue was mixed with calcium hydroxide, ignited at 500°C. for one hour and extracted with concentrated hydrochloric acid. It was then analysed using the same curcumin spectrophotometric method. Equivalent concentrations of hydrochloric acid were used in the standards.

2.2.4 Statistical analyses

Analysis of variance was applied to data from the field and laboratory experiments and to the plant biomass estimations from the additives plots. A regression analysis was used to quantify rates of decomposition in terms of organic matter decrease over time. Data from the plant growth experiments was analysed using a regression procedure to fit smooth curves to dry weight increase over time. This was developed using a response - surface analysis to investigate the three-way relationship of growth over time, at different refuse dosage rates.

More specifically, the analyses were:

(a) For the randomised block design of the additive field trials:

A two-way analysis of variance was carried out on data for measured soil and plant parameters, followed by an F-test giving levels of significant difference; values of p were tabulated. For "post hoc" comparisons to establish wherein the variation lies in data for which the preliminary ANOVA had indicated significant differences, values of Least Significant Difference ($p < 0.05$) were calculated (Parker, 1979).

(b) For the completely randomised design used in the additives laboratory experiments, and incubation experiments: A one-way analysis of variance, again followed by a comparison of treatment means using Least Significant Difference.

Analysis of variance was not considered relevant for the graded data of the dosage experiments (field and laboratory). Here mean data and standard deviations are tabulated.

(c) The most significant kinetic expression of organic matter decomposition is in the form of exponential functions (Chase and Gray, 1957; Broadbent and Nakashima, 1974; Hsieh, 1981). Rates of decomposition have been expressed as the regression coefficient estimated for the natural logarithm of absolute weight of organic carbon as a function of time (Lanning and Williams, 1979). Bearing this in mind certain parameters were selected from those measured and a linear regression analysis applied to either the log-transformation of the absolute data, or the data expressed as a percentage of the control at each sample time (to eliminate external variation), against time in years. The strength of the relationship was tested using the product moment correlation coefficient. Quantification of the rate of decomposition was obtained from the angle of slope of the regression line (regression coefficient).

(d) The step-wise regression analysis (up to the third order) of Hunt and Parsons (1974) was used to analyse plant growth in terms of shoot dry weight from field and laboratory experiments. This analysis, applied to logarithmically transformed data fits smooth curves to the functions of increasing dry weight over time, and relative growth rate; this can be described simply as

$$RGR = d\,\frac{(\log_e Y)}{dt} \qquad \text{where } Y = \text{shoot dry weight in g.} \quad t = \text{time}$$

95% confidence limits are included to allow for comparison between the treatments.

Developing from the plant growth curve derivation a separate program applied response-surface analysis to the plant growth data from the dosage experiments. This type of analysis has a general aim of examining the response of more than one variable to different levels of a stimulus (Mead and Pike, 1975); in this case the response of plant growth rates over time to different levels of refuse application. More specifically, it aims to estimate the point of maximum response. This analysis, therefore, was an attempt to summarise quantitatively the response in terms of plant growth during the early stages of biodegradation of the soil/refuse mixtures. The technique again used a step-wise regression procedure which minimises the sums of squares in the relationship between the three variables.

CHAPTER 3

EFFECT OF ADDITIVES

3.1 Decomposition of the refuse separates and effects on the amended soil

Organic components of samples taken from the various test treatments outlined in Section 2.2.1, were measured over a 28 month period in order to assess the effect of those treatments on the patterns and rates of breakdown. Soil improvement during the period was considered in terms of mineralisation of the proteinaceous components of the waste to soluble and plant available forms of nitrogen, build up of humic material, and changes in soil pH and moisture holding capacity.

3.1.1 Results of field trials.

Preliminary analysis of materials

Table 4 presents a full analysis of the materials used in this investigation, including the top-soil on the field test site. The recorded values established control levels of the various parameters under test. The C:N ratio of the Doncaster refuse separates was shown to be lower than that normally found in unsorted domestic refuse (e.g. 65:1, Webber, 1978, and 50:1, Rees, 1980), presumably due to the higher proportion of putrescible material containing complex nitrogenous compounds, and proportionately less newspaper and rags. The C:N ratios of the putrescible and paper containing fractions of municipal wastes are approximately 15:1 and 100:1 respectively (Galler and Partridge, 1969).

Biodegradation of organic materials

Results from measurement of the ethanol/water soluble fraction, representing the readily decomposable carbohydrates and fats in the refuse are shown in Table 5 and Figure 2. There was a rapid decrease in the percentage material occurring in this fraction in all treatments during the first 10 months, particularly Treatment 2 which dropped to a level well below the control. This was probably due to rapid microbial assimilation. The exception to the rapid drop in ethanol/ water soluble material is Treatment 3, to which additional refuse was

	Soil	Refuse separates	Sewage sludge
pH	7.6	5.7	6.1
Organic Carbon %	1.83 ± 0.2	32.8 ± 0.9	17.22 ± 1.3
Oxidisable Carbon %	1.11 ± 0.09	15.85 ± 2.33	7.60 ± 0.57
Organic Nitrogen %	0.14 ± 0.008	1.02 ± 0.15	1.58 ± 0.12
C:N Ratio	13:1	32.2 : 1	10.8 : 1
Ammonium-Nitrogen μg/g	14.3 ± 2.0	2320.6 ± 429.7	75.2 ± 6.1
Nitrate Nitrogen μg/g	25.9 ± 3.8	0	388.2 ± 62.3
Ethanol/water solubles %	0.86 ± 0.3	28.35 *	17.05 *
Humic Acid %	0.76 ± 0.31	6.53 ± 1.46	4.99 ± 1.4
Moisture %	13.09 ± 1.49	59.4 ± 7.3	40.47 ± 1.36
Lead μg/g	4.0 - 5.9 (4.9)	15.0 - 28.0 (19.8)	20.4 - 31.1 (26.6)
Copper μg/g	3.9 - 6.0 (5.0)	12.0 - 98.0 (31.4)	13.0 - 17.0 (14.6)
Zinc μg/g	9.3 - 12.5 (11.3)	94.5 - 163.5 (112.5)	540.4 - 616.8 (575.8)
Cadmium μg/g	1.8 - 2.0 (1.89)	2.05 - 8.65 (3.4)	14.0 - 18.8 (16.83)
Boron μg/g	0.6 - 2.88 (1.5)	9.64 - 10.16* (9.9)	4.44 - 5.36 * (4.9)

Six replicates used throughout, except where starred, here two replicates used.
For extractable metals, ranges are given with mean figures in brackets.

TABLE 4. Analysis of "Raw Materials"

Time (months)	Treatment 1	2	3	6	7	8	p	l.s.d.
4	0.86	1.30	1.41	1.89	2.26	2.26	0.001	0.46
10	1.06	0.42	1.51	1.27	1.13	1.14	0.001	0.30
16	0.61	0.86	0.91	1.08	0.70	1.05	0.001	0.23
22	0.42	0.36	0.48	0.70	0.75	0.50	n.s.	-
28	0.33	0.30	0.34	0.58	0.50	0.48	n.s.	-

l.s.d. = least significant difference. n.s. = not significant.

TABLE 5. Percentage ethanol/water soluble material (dry weight basis)

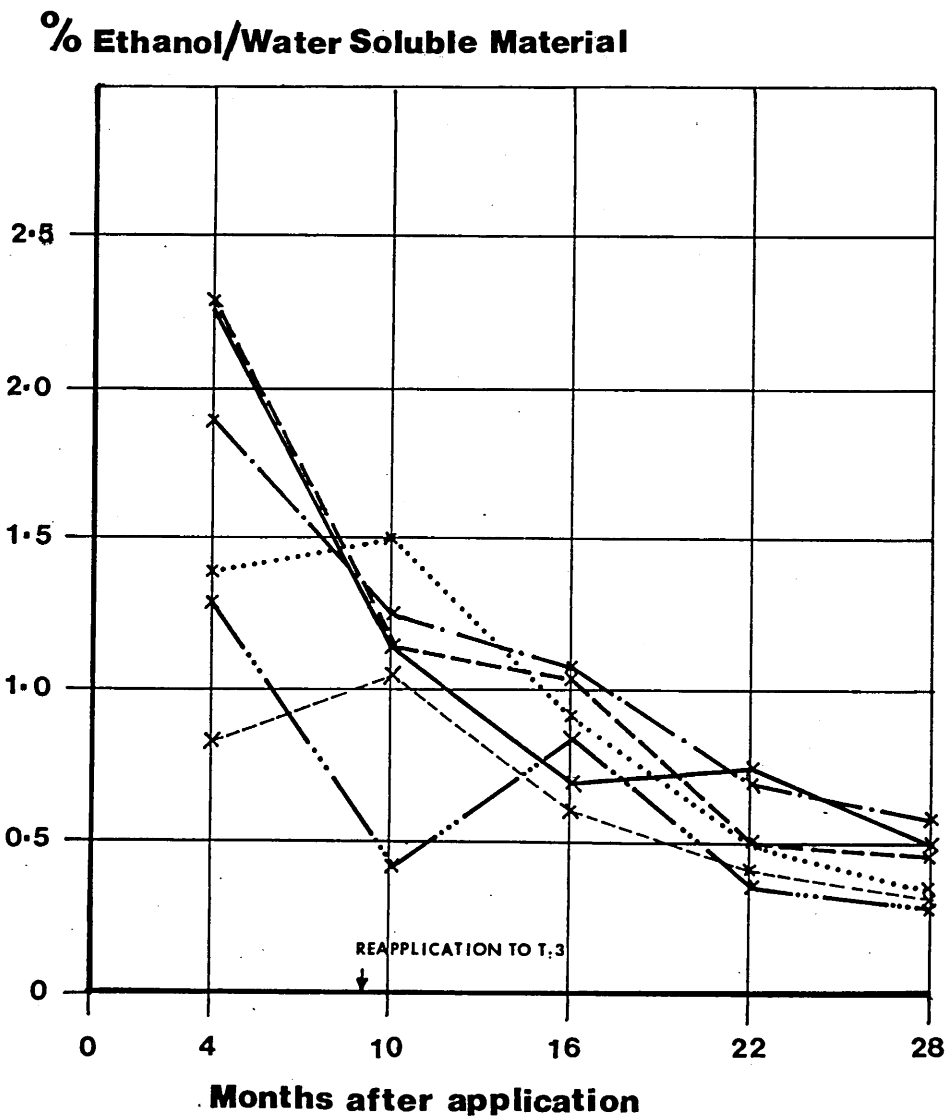


FIG.2
Changes in soil content of Ethanol/Water soluble material under different treatments

KEY

------	=	**Soil Control**
—···—	=	**Soil+Refuse**
··· ····	=	**Soil+Refuse x 2.**
—·—	=	**Soil+ Refuse + Sludge**
——	=	**Soil + Sludge**
— — —	=	**Soil + Refuse + Sludge+Colliery Shale**

added prior to the second analysis.

The more resistant higher molecular weight materials found in the putrescibles, for example cellulose and hemicelluloses will degrade more slowly. Slight upturns in the general downslope trend of the graph presumably indicates the inclusion of the soluble breakdown products from the cellulose, hemicellulose, lignin and wax during later stages of decomposition. Each of these materials will have a range of biodegradability; biodegradation of cellulosics, for example, varies depending upon the extent to which they have been processed, and the degree of crystallinity and lignification (Rees, 1980).

Less than 2 years after incorporation, there was no significant difference in any treatment from the control ($p > 0.05$).

Results from the analyses of soil organic carbon are tabulated (Table 6) and shown graphically (Figure 3).

Decreases of up to 35% occurred during the first six months, the more rapid decrease being observed in the treatments containing higher levels of organic material initially, and especially in those containing sewage sludge. Seasonal fluctuations, and the build up of soil organic matter from decomposing plant litter on all plots complicated the data and so in the subsequent estimations of decomposition rates by regression analysis the figures were taken as percentages of the control at each sample time. Results of this regression analysis are shown in Table 7.

Rates of decomposition (as represented by the regression coefficients) appear to increase from Treatment 2 to Treatment 8, although the variation is small: these results will be discussed more fully in Section 3.1.2.

Levels of organic carbon remained significantly higher than the control ($p < 0.05$) in all treatments more than two years after initial incorporation of the refuse separates.

When organic additions are made to a soil, microbial processes will tend to bring the organic matter content to an equilibrium value, and "stabilise" the soil carbon content. The time period for

Time (months)	Treatment 1	2	3	6	7	8	p	l.s.d.
4	1.78	2.96	3.04	4.94	5.70	6.81	0.001	1.14
10	2.04	2.47	3.76	3.88	3.80	4.39	0.001	0.92
16	2.24	2.88	3.87	3.22	4.03	4.35	0.001	0.41
22	2.12	2.70	3.20	3.76	3.78	4.34	0.001	0.72
28	2.69	3.45	3.66	3.92	4.35	4.64	0.001	0.66

l.s.d. = least significant difference.

TABLE 6. Percentage organic carbon (dry weight basis)

NOTE: Data for Treatment 8 are slightly distorted due to the inclusion of some elemental carbon from the colliery shale.

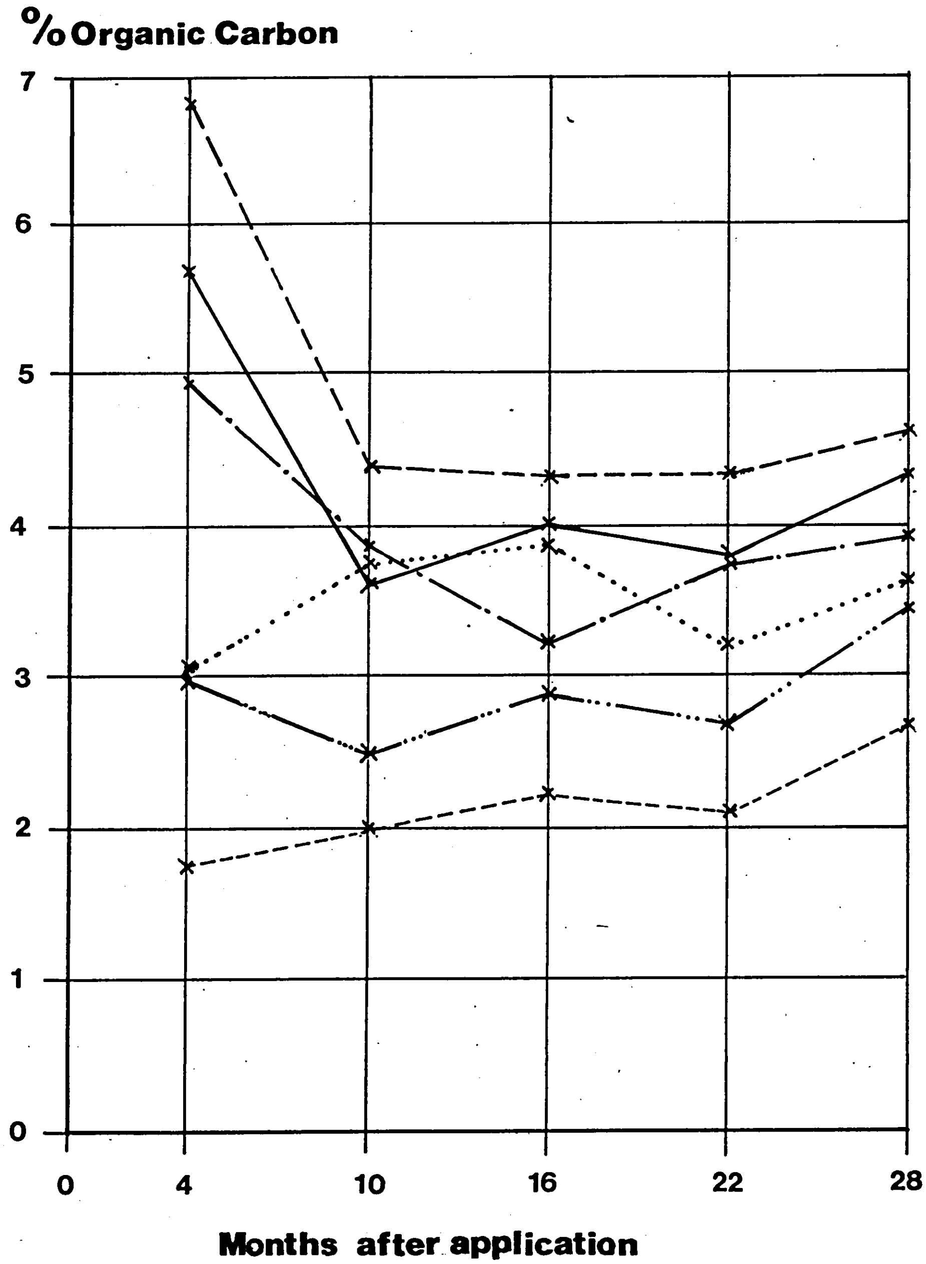


FIG. 3
Changes in soil Carbon content under different treatments

KEY

- - - - -	=	Soil Control
—···—	=	Soil + Refuse
· · · · ·	=	Soil + Refuse x 2
—· —	=	Soil + Refuse + Sludge
———	=	Soil + Sludge
— — —	=	Soil + Refuse + Sludge + Colliery Shale

Treatment	b	r	p
2	-0.094	-0.59	n.s.
3	-0.209	-0.99	0.01
6	-0.272	-0.80	0.10
7	-0.280	-0.82	0.10
8	-0.329	-0.84	0.10

b = regression coefficient of the natural logarithm of percentage carbon as a function of time (years).

r = correlation coefficient.

TABLE 7. Relationship between carbon decomposition and time under different treatments.

attaining the equilibrium state usually depends on the supply of soil nitrogen. Stewart and Webber (1976) state that the carbon:nitrogen ratio of added wastes is perhaps the most important determinant of decomposition rate. Wastes with a high C:N ratio will result in slow decomposition due to lack of available nitrogen for conversion into microbial biomass; a low C:N ratio (less than 10:1) may result in rapid mineralisation of the organic nitrogen producing excessive quantities of nitrate which can reduce crop quality or percolate into water supplies.

Results of the organic nitrogen analyses (Table 8, Figure 4) show slow decomposition, although only after a lag of approximately six months.

Rates of mineralisation of organic nitrogen approximated to 20% during the first year in all treatments except the sewage sludge alone (Treatment 7) where there was no net loss of organic nitrogen over the first year. The reason for this is likely to be that the sludge used was in a well-digested and stable state, whereas the Doncaster refuse was fresh and underwent rapid decomposition. Mineralisation of organic nitrogen following waste additions to soil tends to follow an exponential decay series (Loehr, 1979); the results here indicate very little change in levels of organic nitrogen after the first year in all treatments.

Humification

Measurement of the humic acid content of the soil over the test period gave an indication of the extent to which humification occurred during and after the biodegradation of the added wastes. The results are shown in Table 9 and Figure 5.

There appears to have been a rapid reduction in the levels of humic acids during the first six months when decomposition of organic matter was at its maximum. After this the small fluctuations indicate stability. With regard to long-term build up of humic material it is worth noting that all treatments remained significantly different from the control ($p < 0.001$) more than two years after initial incorporation. Table 10 showing the data expressed as a percentage of the

Time (months)	Treatment						p	l.s.d.
	1	2	3	6	7	8		
4	0.14	0.21	0.22	0.33	0.33	0.38	0.001	0.05
10	0.16	0.23	0.26	0.34	0.32	0.36	0.001	0.06
16	0.17	0.17	0.25	0.28	0.33	0.29	0.001	0.04
22	0.19	0.22	0.25	0.29	0.30	0.26	0.001	0.04
28	0.17	0.21	0.23	0.27	0.35	0.26	0.001	0.04

l.s.d. = least significant difference.

TABLE 8. Percentage organic nitrogen (dry weight basis)

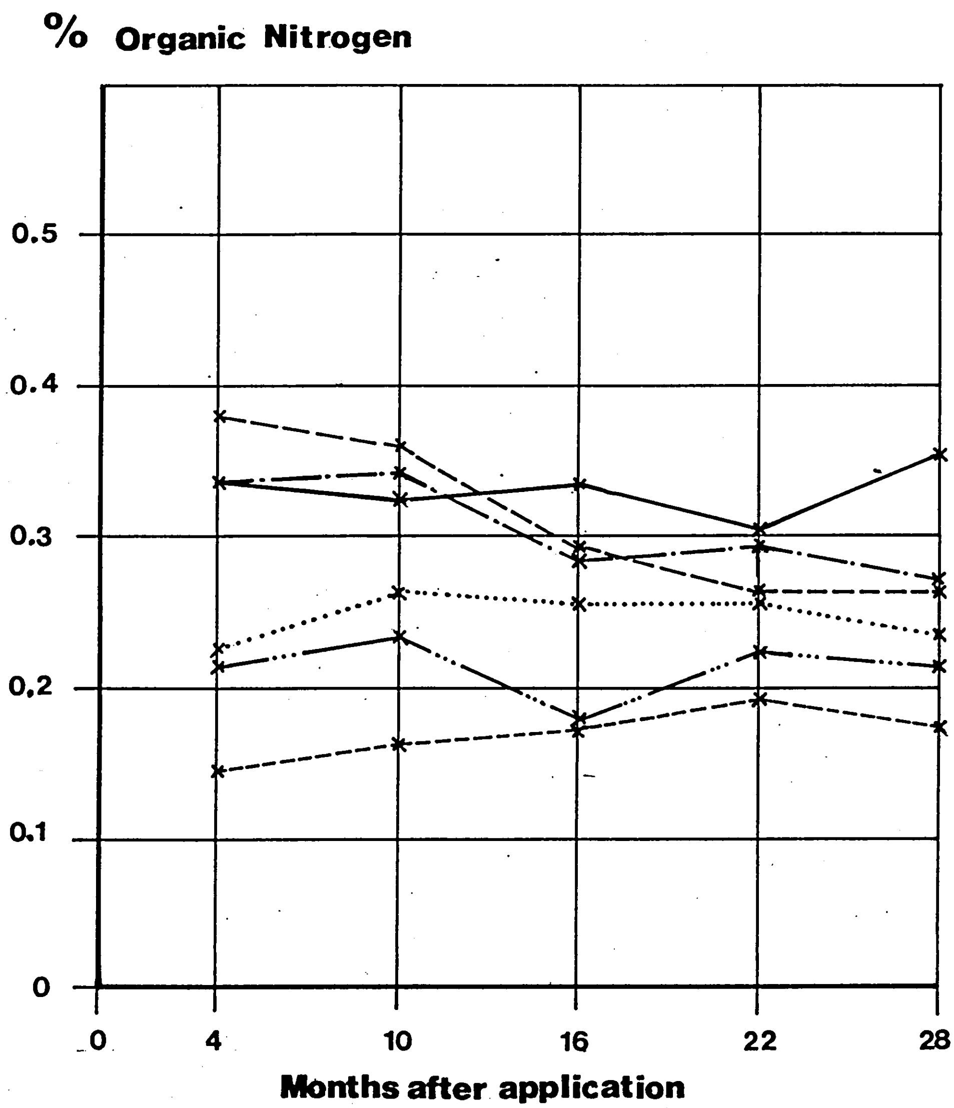


FIG.4
Changes in soil Nitrogen content under different treatments

KEY

----	=	Soil Control
—··—	=	Soil + Refuse
········	=	Soil + Refuse x 2
—·—	=	Soil + Refuse + Sludge
——	=	Soil + Sludge
— — —	=	Soil + Refuse + Sludge + Colliery Shale

Time (months)	Treatment 1	2	3	6	7	8	p	l.s.d.
4	0.76	1.48	1.38	2.85	2.19	1.60	0.001	0.53
10	0.77	0.92	1.07	0.78	0.85	0.86	n.s.	-
16	0.54	0.68	1.19	0.86	0.99	0.89	0.001	0.21
22	0.32	0.52	0.68	0.70	0.79	0.60	0.001	0.17
28	0.32	0.60	0.58	0.77	1.15	0.75	0.001	0.19

l.s.d. = least significant difference. n.s. = not significant.

TABLE 9. Percentage soil humic acid content (dry weight basis).

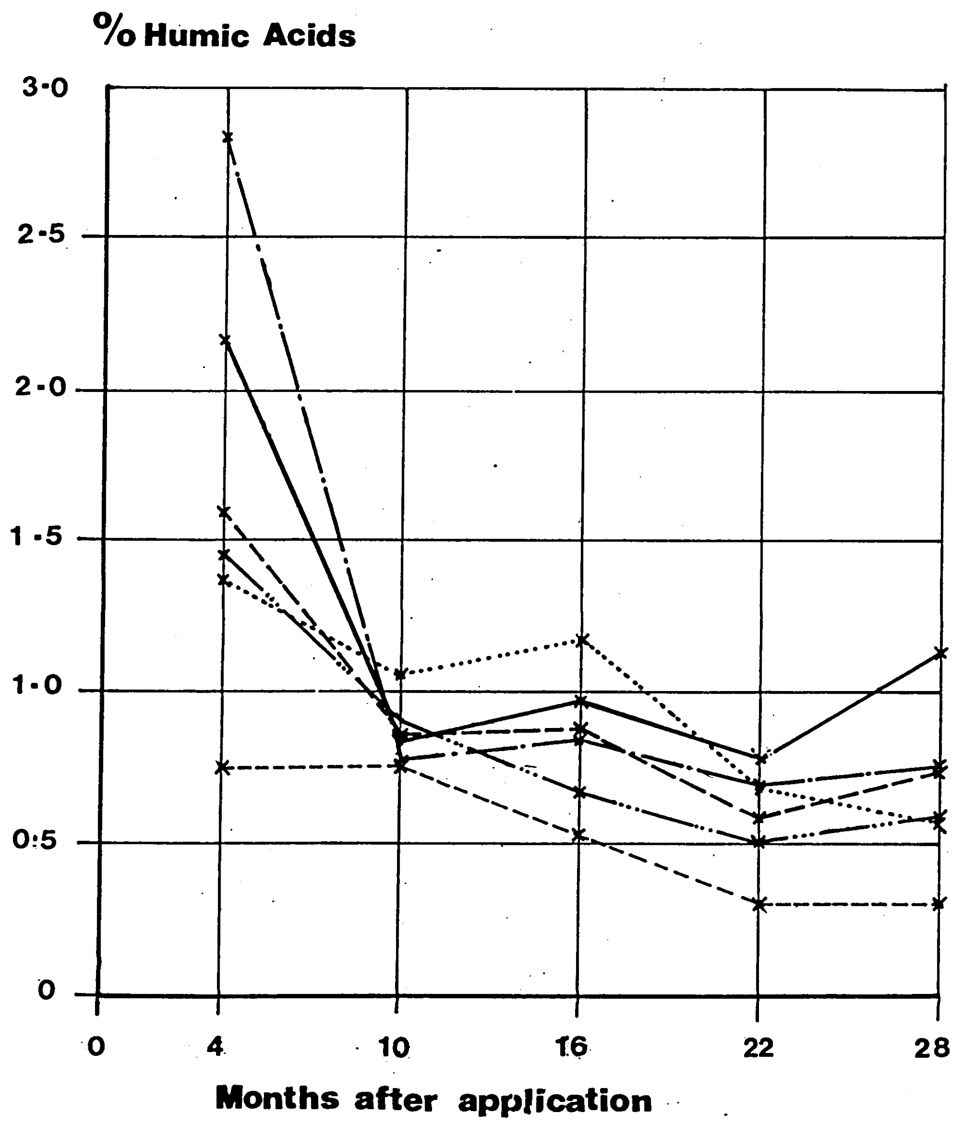


FIG.5
Changes in soil Humic Acid content under different treatments

KEY

- - - - -	=	**Soil Control**
—··· —	=	**Soil + Refuse**
· · · · · ·	=	**Soil + Refuse x 2**
— · —	=	**Soil + Refuse + Sludge**
———	=	**Soil + Sludge**
— — —	=	**Soil + Refuse + Sludge + Colliery Shale**

Time	Treatment					
(months)		2	3	6	7	8
4		194.7	181.6	375.0	288.2	210.5
10		119.5	138.9	101.3	110.4	111.69
16		125.9	220.4	159.3	183.3	164.8
22		162.5	212.5	218.75	246.9	187.5
28		187.5	181.25	240.6	359.4	234.4

TABLE 10. Soil humic acid content expressed as percentage of soil control at each sample time.

soil control adds weight to the suggestion that in the long-term the humification of the organic wastes gives a lasting reserve of humic material, with the advantages in terms of soil improvement that this humification brings.

Notable here are the greater levels of humification on the plots containing sewage sludge.

Soil pH

Soil acidity was measured at each sample time and the results (Table 11) show only marginal differences between any of the waste treatments and the control. The slight drop in pH at 4 months may reflect the production of organic acids or other biochemical alterations during early biodegradation, or they may simply be due to the lower pH values of the sludge and refuse themselves. At subsequent sampling times the pH of all plots followed the control very closely.

Moisture holding properties

Table 12 shows that percentage moisture of all the plots was increased following the addition of the waste materials.

The effect would seem to be relatively long-lived [all treatments except Treatment 8 show levels above the controls ($p < 0.05$) even after two years]. The effect was still apparent during times of stress, for example in May 1979 (16 months) during a period of unusually low rainfall in South Yorkshire (see Finningley data, Appendix 1).

Percentage moisture content of a soil, though a useful comparative measure in this investigation, is not of direct relevance to plant growth. Plant available water is considered to be the volume retained between field capacity and permanent wilting point (conventionally taken as moisture tension at 0.05 and 15 bars respectively).

Samples were taken from the plots in March 1982 and equilibrated at suction pressures of 0.05 and 15 bars in order to measure the effect, if any, of waste additions to soil moisture holding properties. The results are shown in Table 13.

Table	Treatment					
(months)	1	2	3	6	7	8
4	7.57	6.95	7.15	6.68	6.42	6.56
10	6.77	6.64	6.74	6.66	6.52	6.44
16	7.38	7.25	7.14	7.24	7.24	7.10
22	7.88	7.81	7.72	7.70	7.62	7.67
28	6.81	6.84	6.87	6.87	6.73	6.82

TABLE 11. Soil pH.

Time (months)	Treatment 1	2	3	6	7	8	p	l.s.d.
4	13.09	15.90	16.68	19.15	18.85	18.14	0.001	2.65
10	18.81	22.27	24.40	24.27	23.67	28.87	0.001	2.25
16	9.06	10.27	10.66	10.81	10.60	10.93	0.05	1.20
22	20.80	21.10	22.48	23.81	22.77	21.38	0.05	2.03
28	18.67	20.30	21.59	20.81	23.15	19.64	0.001	1.80

TABLE 12. Percentage moisture content of treated soil

Treatment	Moisture held at 0.05 bars. %	Moisture held at 15 bars. %	A.W.C.
1	25.1	10.45	14.65
2	23.3	10.5	12.8
3	25.2	10.7	14.5
6	23.4	11.3	12.1
7	26.7	12.6	14.1
p l.s.d.	n.s. -	0.05 0.83	n.s. -

A.W.C. = available water capacity.

l.s.d. = least significant difference.

n.s. = not significant.

TABLE 13. Moisture holding capacity of treated soil after three years.

The apparent increase in moisture held at 15 bars in Treatments 2 and 3 containing the Doncaster refuse is not significant ($p > 0.05$). Treatments 6 and 7, however, did show significantly higher ($p < 0.05$) amounts of moisture retained at this pressure (equivalent to permanent wilting point). Neither refuse nor sewage sludge addition had any significant effect on water held at field capacity (0.05 bars). The difference between the two gives the available water capacity, which here indicated that waste addition caused a decline in the overall storage capacity of the treated soil.

It is perhaps, relevant to note that these samples were taken some three years after initial incorporation of the refuse and sludge, which was at a fairly low dosage. It may be that earlier effects had by then disappeared.

Trace elements

Zinc is a necessary trace element for most organisms as an activator of certain enzyme systems. It tends to be considered a problem due to its phytotoxicity when present in excessive amounts, but there is no evidence of any adverse effects on human health associated with zinc contaminated foodstuffs (D.O.E., 1980). It has numerous sources in refuse stemming from its widespread use in the smelting, refining and galvanising industries. Normal ranges in soil are 10 - 300 μg/g total zinc and 1 - 40 μg/g extractable zinc; and in plant materials 15 - 100 μg/g (Allen, 1974).

Table 14 shows extractable zinc in soil samples from the additive treatment plots over the 28 month test period. Levels of extractable zinc in the refuse and sewage sludge are recorded in Table 4.

Shoots of couch grass (*Agropyron repens*) were collected for analysis in the summers of 1979 and 1981, and treated according to the method outlined in Chapter 2. The results are shown in Table 15.

These two sets of results indicate that both extractable zinc in the soil, and plant uptake of zinc was increased by each of the waste treatments, more notably by the sewage sludge (which had a higher zinc content itself) than by the refuse.

Time (months)	Treatment 1	2	3	6	7
0	9.3-12.5 (11.3)	26.5-41.0 (32.7)	26.5-41.0 (32.7)	109.5-208.5 (144.8)	145.5-226.0 (214.3)
4	9.3-12.5 (11.3)	13.3-53.7 (34.1)	21.4-106.3 (62.3)	54.7-128.3 (97.6)	67.4-230.9 (150.0)
16	10.5-13.5 (12.6)	24.5-32.5 (29.1)	27.5-46.5 (37.7)	24 - 84.5 (54.5)	84.5-123.0 (93.8)
28	12.5-17.5 (14.6)	21.5-50.0 (31.9)	22 - 46.5 (36.1)	40.0-86.5 (64.7)	86.0-105.5 (96.5)

TABLE 14. 0.1 M HCl extractable zinc (μg/g) in treated plots
(Ranges are given, plus means of 5 samples in brackets)

	Treatment				
	1	2	3	6	7
July '79 (4 months)	23.75-56.25 (30.3)	30 - 50 (39.51)	30 - 68.75 (41.49)	37.5-130 (63.5)	54.99-83.3 (66.16)
Sept '81 (30 months)	25-36.25 (29)	17.5-32.5 (24.25)	25-28.75 (26.75)	17.5-37.5 (28.25)	18.75-30 (23.5)

TABLE 15. Total zinc content of couch grass shoots (μg/g)
(ranges given, plus means of 3 samples in brackets)

There appeared to be little change over time in levels of extractable zinc in the refuse amended plots, but a fairly rapid decrease occurred in the plots amended with sewage sludge (Treatments 6 and 7), coinciding with plant uptake during the first growing season.

After 2 years no treatment showed zinc content in plant tissues to be any different from that in the controls. It would seem from this that there is indeed a decrease in availability of zinc by this time, presumably due to the formation of metal complexes with the more stable organic substances in the soil.

One additional application of refuse (Treatment 3) seemed to have little effect on the long-term build-up of zinc and levels were only marginally higher than those in Treatment 2 at the end of this investigation.

It is interesting to note that plant uptake of zinc was as high on the refuse and sewage sludge treatment (Treatment 6) as on the sewage sludge alone (Treatment 7), although extractable soil levels were lower. In addition, a higher proportion of the zinc in the refuse treatment was taken into the plants than from the sewage sludge treatment. Zinc in the refuse, therefore, would seem to be more readily available than zinc in the sludge.

Plant requirement for copper is lower than for zinc, and copper again is essential for plants to activate certain enzyme systems, especially those linked with oxidation processes. Excess copper can give rise very readily to toxicity in lambs, and in clover which is one of the most sensitive crops (Williams, 1980). Clover is, of course, a popular landscaping species and this sensitivity should be borne in mind if areas reclaimed using Doncaster refuse are being revegetated. Normal ranges of copper in soils are 5 - 100 μg/g (total), 0.1 to 30 μg/g (extractable), and the range in plant materials is 2.5 to 25 μg/g (Allen, 1974).

Measured levels of copper in the treated soils, and shoots of couch grass (*Agropyron repens*) are shown in Tables 16 and 17.

All waste additions increased levels of extractable copper above

Time (months)	1	2	3	6	7
0	3.9-6.0 (5.0)	4.0-9.0 (5.8)	4.0-9.0 (5.8)	2.0-8.0 (4.8)	5.0-9.0 (6.8)
4	3.9-6.0 (5.0)	3.5-19.9 (9.5)	5.7-38.8 (18.4)	3.9-13.9 (7.8)	4.4-14.5 (9.5)
16	3.0-4.0 (3.5)	4.5-10.0 (6.56)	5.0-12.5 (9.6)	4.1-9.8 (7.5)	4.9-7.3 (6.3)
28	3.0.-4.0 (3.5)	4.3-8.0 (5.9)	5.0-12.0 (8.6)	6.0-10.5 (7.9)	6.5-10.0 (7.8)

TABLE 16. 0.1 M HCl extractable copper (μg/g) in treated plots (ranges given, plus means of 5 samples in brackets)

	Treatment				
	1	2	3	6	7
July '79 (4 months)	3.0-8.75 (6.35)	2.45-9.25 (4.44)	3.0-6.25 (4.63)	4.0-6.25 (5.0)	4.5-7.5 (5.2)
Sept '81 (30 months)	2.45-3.5 (3.0)	2.45-4.25 (3.22)	2.45-4.5 (3.14)	2.45-3.0 (2.59)	2.45-5.5 (3.17)

TABLE 17. Total copper content of couch grass shoots (μg/g) (ranges plus means of 3 samples in brackets)

the control, by a greater amount in the case of the refuse putrescibles than the sewage sludge (reflecting the higher levels found in the refuse itself). Plant uptake of copper seems to be unrelated to extractable levels, possibly due to accumulation in the roots of the grass (only shoot material was analysed), or to binding by organic matter. There was no significant change over time in the levels of extractable copper in any of the treatments.

Lead has been shown to be phytotoxic to many plant species although only at very high concentrations. A few species, for example, leadwort (*Minuarta verna*) and thrift (*Armeria maritima*) have evolved highly tolerant genotypes (Antonovics et.al., 1971). It is both toxic and cumulative in animals and man. The main risks from lead in soils are from the surface contamination of crops and the ingestion of lead contaminated soil or sludge by children or animals (Williams, 1980). Lead is widely distributed through its uses in piping, batteries, pigments, dyeing and glass. It is used in combination with arsenic (lead arsenate) in sprays. Also, at present there is an unavoidable enrichment in soil concentration from atmospheric lead pollution, and concentrations of 2000 - 3000 μg/g lead are not uncommon in street dusts (D.O.E., 1980).

The normal lead content of soils ranges from 2 - 200 μg/g (total), 1 - 10 μg/g (extractable), and 1 - 10 μg/g in plant materials (Williams, 1980). It should be remembered that the proportion extractable by standard methods tends to be relatively low and in consequence may not truly represent "plant available" lead.

Results of lead analyses are shown in Tables 18 and 19.

All waste additions resulted in a notable increase in extractable lead, although this decreased quite markedly over time, again suggesting a decrease in availability as the waste materials break down and become stable.

Uptake of lead by couch grass seemed to be unrelated to both treatment, and concentration of lead in the soil. This can be explained either by accumulation of lead in the roots, or by a lack of availability in the soil. It would appear, therefore, that the data

Time (months)	Treatment				
	1	2	3	6	7
0	4.0-5.9 (4.9)	14.0-22.0 (16.0)	14.0-22.0 (16.0)	12.0-20.0 (14.4)	18.0-24.0 (19.6)
4	4.0-5.9 (4.9)	5.9-17.7 (10.3)	7.6-21.1 (14.6)	9.9-25.5 (16.1)	8.8-19.4 (12.7)
16	3.0-6.0 (4.4)	3.0-8.0 (4.4)	6.0-10.0 (7.8)	5.0-8.0 (6.8)	8.0-11.0 (9.8)
28	3.0-5.0 (4.2)	4.0-7.0 (5.0)	4.0-10.0 (5.8)	4.0-10.0 (6.6)	5.0-18.0 (9.8)

TABLE 18. 0.1 M HCl extractable lead (μg/g) in treated plots (ranges plus means of 5 samples in brackets)

	Treatment 1	2	3	6	7
July '79 (4 months)	7.5-22.5 (15.7)	11.35-21.75 (15.15)	10.5-21.75 (17.09)	13.75 - 23 (18.2)	10.5-16.66 (13.28)
Sept '81 (30 months)	7.5-14.25 (9.85)	5 - 10.5 (8.20)	5.75-18 (10.19)	5.75-6.75 (6.5)	6.75-16 (10.05)

TABLE 19. Total lead content of couch grass shoots (μg/g) (ranges plus means of 3 samples in brackets)

here supports the general view that organic matter addition to soil results in a decrease in lead availability to plants, even when the additions increase the levels of the element in that soil.

Cadmium causes concern not only for its marked phytotoxicity when present in excess, but also because, like lead, it is a cumulative poison to mammals. It normally enters the environment as an industrial waste product from metal refining, plating, chemical and paint industries (Allen, 1974). As such it is more usually found as a contaminant in sewage sludge than in domestic refuse. In the present investigation levels in the sludge were approximately 5 times as high as those in the refuse (Table 4). Soils normally contain 0.1 - 2 μg/g cadmium and plant materials 0.2 - 0.5 μg/g (Williams, 1980). Cadmium tends to be very soluble and according to the D.O.E. report 1980, there may be very little difference between 'soluble' and 'total' cadmium levels for certain soil types.

Results of cadmium analyses are shown in Tables 20 and 21.

Initial levels of cadmium in the control soil appeared to be rather high, but the addition of the Doncaster refuse raised the level only slightly. Sewage sludge addition, however, did increase the extractable cadmium to levels considered to be contaminating. Soil cadmium concentrations appeared to decrease slightly over time in the treated plots, although this element is known to exhibit a low capacity for organic matter binding. The highest concentrations of extractable cadmium were found in Treatment 7 and these were also reflected in higher plant tissue concentration.

The data presented here does show increased levels of cadmium in plant tissue, although these increases are small for both sewage sludge and refuse addition treatments and fall within the range normally found in plants growing on uncontaminated soil.

Boron is required as a trace element by plants and is important in metabolism due to its involvement in cell structure and sugar translocation. The tolerance range between toxicity and deficiency for plants appears to be low (less than 1 μg/g in soil may cause deficiency, greater than 3 μg/g can be toxic) (Hesse, 1971). For this reason boron can create a pollution hazard due to its widespread use in the

Time (months)	Treatment 1	2	3	6	7
0	1.8-2.0 (1.89)	2.05-2.3 (2.12)	2.05-2.3 (2.12)	5.6-8.7 (6.7)	7.0-11.2 (8.86)
4	1.8-2.0 (1.89)	1.95-2.1 (2.01)	2.0-4.3 (2.46)	3.5-4.95 (4.27)	4.25-10.7 (7.23)
16	1.7-2.3 (1.94)	1.7-2.6 (1.94)	1.7-2.3 (2.0)	2.5-5.4 (3.68)	5.9-7.2 (6.6)
28	1.5-2.0 (1.74)	1.5-2.0 (1.9)	1.7-2.3 (2.0)	3.0-5.2 (4.25)	6.6-9.6 (7.64)

TABLE 20. 0.1 M HCl extractable cadmium (μg/g) in treated plots (ranges plus means of 5 samples in brackets)

	Treatment				
	1	2	3	6	7
July '79 (4 months)	0.12-0.20 (0.16)	0.14-0.27 (0.21)	0.19-0.22 (0.21)	0.20-0.31 (0.26)	0.24-0.25 (0.25)
Sept '81 (30 months)	0.13-0.13 (0.13)	0.19-0.23 (0.21)	0.23-0.24 (0.24)	0.15-0.19 (0.17)	0.27-0.31 (0.29)

TABLE 21. Total cadmium content of couch grass shoots (μg/g)
(ranges plus means of 3 samples in brackets)

paper, wood, glassware, pottery and glue-making industries. The main risks are to plant growth, but boron compounds tend to be highly soluble in the soil and rapid leaching means that any adverse effect on plant growth is likely to be of limited duration. Levels of 5µg/g have been known to cause abortion in cows (A.E.Higginson - Personal Communication).

The normal range of extractable boron in soil is 0.2 - 5 µg/g and in plant materials is 10 - 80 µg/g (Allen, 1974).

Results of soil analysis for water extractable boron are shown in Table 22.

All waste treatments increased the level of boron, with the refuse separates having a more profound effect than the sewage sludge: this would be expected from the recorded concentration of boron in the materials themselves (Table 4). After two years the added boron had been reduced through leaching processes, but the levels were still generally above those on the control plots.

No plant tissue from the plots was analysed for boron concentra-ion.

3.1.2 Discussion of results

Biodegradation

Biodegradation of the decomposable materials in the Doncaster refuse followed an exponential pattern of breakdown, with rates decreasing over time. This pattern is commonly accepted as the best representation of organic matter decomposition in soil (Broadbent and Nakashima, 1974; Hsieh et. al., 1981). The initial rapid reduction in the ethanol/water soluble, and organic carbon fraction represents the decay of easily decomposable components by the activities of micro-organisms and small soil invertebrates and by the leaching of the soluble products of decomposition. After the first six months the rate of decrease in both fractions slowed down and probably represents breakdown of the more recalcitrant components, and the incorporation of some of the added carbon into more stable organic matter (Jenkinson, 1965; Lemming and Williams, 1979). This incorporation into the more resistant resident soil organic matter was seen by the

Time (months)	Treatment 1	2	3	6	7
0	0.6-2.8 (1.5)	1.5-3.4 (2.3)	3.5-6.0 (4.7)	5.6-6.2 (5.9)	2.6-5.4 (3.6)
16	1.3-2.6 (1.9)	2.0-3.3 (2.6)	3.2-5.3 (3.7)	1.2-3.6 (2.4)	2.0-3.6 (2.4)
28	0.6-2.0 (1.4)	1.2-3.9 (2.0)	1.5-3.8 (2.8)	0.9-2.6 (2.0)	0.8-2.3 (1.8)

TABLE 22. Water extractable boron (μg/g) in treated plots (ranges plus means of 5 samples in brackets)

gradual increase, relative to the control, of humic acids in the treated soil plots over the subsequent 2 years.

Rate of breakdown of the refuse, as represented for example by change in organic carbon (Table 7), appeared to be more rapid in the treatments containing sewage sludge. This would agree with the generally accepted view that nitrogen supplementation is a necessary aid to the breakdown of domestic refuse in soil (King et. al., 1974, 1977; Volk, 1976; Webber, 1978). However, in view of the timing problems associated with the setting up of Treatments 2 and 3 (Section 2.2.1) it is suggested that the regression coefficients representing rates of breakdown for these treatments shown in Table 7 are inaccurate, due to the fact that the first 4 months of rapid breakdown were not included. If the regression coefficient recorded for the comparable treatment in the later 'dosage' trials of - 0.33 (Table 32) is taken as a more reliable figure it would seem that there was little difference in decomposition rate of the refuse whether it was supplemented or not with nitrogen from the sludge. No statistically significant differences were in fact found between the treatments if this later regression coefficent for Treatment 2 was used. It would be expected that the decomposition of the digested sludge would be slower than the 'fresher' refuse since the available energy material (the soluble organic carbon compounds) and nutrients for microbial growth are fewer (Sabey, 1980). Terry et. al. (1979) found that the major portion, viz. 55 to 80%, of sewage sludge organic carbon was resistant to decomposition in soil. Solid domestic refuse, on the other hand, is much less resistant to decomposition and normally 50 - 85% of carbon is released from decaying food or composts (Hartenstein, 1981). Table 6 shows that the reduction in organic carbon on the sewage sludge treated plots (Treatment 7) was indeed only 23% after 28 months. If the data for the refuse treatment is again taken from the more reliable results in the dosage trials (Table 31) it is evident that 41% of the carbon has been released from the refuse at the dosage of 20 tonnes/hectare (the same as in the additive treatments) and this percentage increases at the higher dosage rates. The Doncaster refuse, therefore, was much more amenable to decomposition in soil than was the digested sewage sludge used in this investigation.

The importance of the C:N ratio of waste materials in determining the rate of their decomposition in soil and release of plant available nitrogen has already been stressed. According to Stewart and Webber (1976) it is generally found that decomposing materials with a C:N ratio of 30:1 or greater will decrease the available nitrogen in the soil. Materials with ratios between 15:1 and 30:1 will not markedly affect nitrogen availability but below this will usually increase available nitrogen. The Doncaster refuse had a C:N ratio of 32:1 and was therefore likely to have only a marginal effect on nitrogen availability, but may have caused a small amount of nitrogen immobilisation due to utilisation by microorganisms and incorporation into microbial biomass during the early stages of carbon decomposition. It did not appear that nitrogen supplementation was necessary to enhance the rate of breakdown in the long term, although it may be necessary in order to improve crop growth following addition of refuse and refuse composts to soil, as found by Mays et. al. (1973) and Duggan and Wiles (1976). Possible immobilisation and later release of plant available nitrogen was the subject of more detailed investigation in the plant growth experiment (Section 3.3).

The C:N ratio is a difficult measure to apply since the effect of organic materials when mixed with soil not only depends upon the C:N ratio but also upon the relative availability of carbon and nitrogen. This will be affected by soil properties and also by the degree of mixing between the material and the soil. The soil itself may also provide a considerable amount of nitrogen which would help offset the effect of adding highly carbonaceous material. Table 23 does, however, show the overall trend towards stabilisation of the organic refuse giving C:N ratios for all treatments similar to (or below in the case of the sewage sludge treatments) those on the control plots.

Table 8 indicated that mineralisation of organic nitrogen began only after a lag of about 6 months. This is probably due to the more readily available nitrogen sources (as measured in the ethanol/water soluble fraction) being utilised first by the microorganisms. A similar time lag was found by Broadbent and Nakashima (1974) in their study of rates of decomposition and mineralisation of barley roots which had a C:N ratio of 35.7:1 (c.f. 32.2:1 for the Doncaster refuse). They found no time lag before net mineralisation of nitrogen in barley tops (C:N ratio of 17:1).

Time (months)	Treatment 1	2	3	6	7
4	12.71	14.10	13.82	14.97	17.81
10	12.75	10.74	14.46	11.40	11.88
16	13.18	16.94	15.48	11.50	12.21
22	11.16	12.27	12.80	12.97	12.60
28	15.82	16.43	15.91	14.52	12.43

TABLE 23. Carbon:Nitrogen Ratios (Treated plots)

Humification

Heterotrophic decomposition results in eventual humification of the organic material when it is no longer amenable to decay processes. Humification is generally considered to be an oxidative process involving the production of phenolic and carboxylic benzene rings which are then polymerized through short-lived free radicals whose production is probably catalysed enzymatically. Once formed, the free-radicals rapidly condense. Bonds are also formed between nitrogen-and-sulphur containing compounds and the polymers or their reactive phenolic precursors, and between functional groups on carbohydrates, fatty acids, and other classes of compounds. Among the consequences of humification are the build up of labile organic materials, increases in the cation exchange capacity of the soil, the prevention of odours and the destruction of most types of pathogen. (Hartenstein, 1981)

Humic materials are, of course, themselves subject to decomposition, and since humus is a heterogeneous mixture it would not be expected to decompose uniformly, some humus fractions may be converted into others during the process (Russell 1973). Visser (1962), found that with progressing decomposition of papyrus in tropical swamps over 18 months there was a proportional increase in humic acid content. Over the same time the percentage fulvic acid (the alkali soluble portion not precipitated by acid) decreased. In fact, fulvic acids are considered to be the antecedents of humic acids during humification (Hartenstein, 1981). The humic acid content of the treated plots as shown in Table 9 does indicate a certain amount of decomposition but the fact that the levels increase relative to the control (Table 10) verify that humification and stabilisation of the waste material is progressing. Nevertheless, it should not be assumed that this humification represents a large proportion of the added organic matter from the refuse. Russell (1973) quotes the work of Jenkinson who added ^{14}C labelled ryegrass shoots and roots to soil and monitored their decomposition. He showed that it takes a long time for all the added carbon to become fully humified. Even after 4 years much of it was still in an appreciably different form from the bulk of the soil humus. More than two thirds of the carbon was lost as carbon-dioxide, mostly during the first six months of decomposition. It could, therefore, be assumed that humification of the organic matter in the refuse will continue for some years to come.

A greater build up of humic acids was noted on the sewage sludge treated plots compared with the refuse treatments (Table 10). This can be explained by the fact that fresh, organic matter will decompose rapidly with a large production of carbon dioxide, whereas more stable material, in this case the sewage sludge, decomposes more slowly, leaving a larger residue in the humus fraction.

Soil pH and moisture effects

Addition of the Doncaster refuse, with or without sewage sludge, appeared to have little effect on soil pH at the fairly low dosage applied in this trial (Table 11). The slight depression at the 4 month sample time is probably due to the slightly lower pH values of the sludge and refuse themselves rather than chemical effects in the soil. This effect was not apparent at later sample dates by which time progressive decomposition had produced a more homogeneous mixture.

There appears to be little reported information on the effect of fresh solid refuse on soil pH, but refuse compost additions to acid soils have generally been found to increase pH (May et. al., 1973; Duggan and Wiles, 1977) or maintain pH in comparison to acid-forming fertilizers (Hartenstein and Rothwell, 1973; Volk, 1976). Nitrogenous fertilizers, or waste materials containing large amounts of ammonia-nitrogen generally lead to a decrease in soil pH as nitrification takes place, the bases on the soil exchange complexes being replaced by hydrogen ions (Webber and Doyle, 1975). Cottrell (1975) found that the addition of solid domestic waste decreased soil pH slightly.

Addition of both Doncaster refuse putrescibles and digested sewage sludge, increased the soil moisture content by a significant amount at all sample times (Table 12). This agrees with the findings of Duggan and Wiles (1976) who observed an increase in moisture content of 16% in plots to which municipal compost had been added. Working in semi-arid areas they found this to be an important factor in times of drought stress.

In terms of the water holding capacity of the soil, Table 13 showed that although the moisture held at a suction pressure of 15 bars (permanent wilting point) was increased by the refuse, and more so by the sludge; that held at 0.05 bars was not increased, and the available water capacity was actually decreased by the waste addition. The same effect was reported by Webber (1978). He found that the soil moisture held at 15 bars (on a weight basis) was increased by both shredded waste and liquid sewage sludge, by amounts which were related to the organic carbon in the samples. He also reported no significant differences with waste treatment on available water capacity.

Volk and Ullery (1973), however, found that the addition of up to 400 tons/acre (1000 tonnes/hectare) of shredded solid waste and sewage sludge on a sagehill sand in Oregon improved water retention at field capacity (0.05 bars) but again found that the overall storage capacity of the soil declined. It would seem that the influence of waste materials on soil moisture properties will vary according to the texture of those soils; this will be discussed further in Section 4.1.2. The nature of the organic materials themselves will also play some part; sewage sludge has a more pronounced effect than refuse. Epstein (1973), for example, reported that although additions of sewage sludge to soil did not result in an increase in the volume of available water, the sewage treated soils retained appreciably more moisture than untreated soils at any particular tension. Evidently sewage addition has slightly different effects on soil moisture characteristics than does domestic refuse. The most likely explanation of this is that refuse contains more 'bulky' material, e.g. glass, stones and plastic, than the fairly homogeneous sewage sludge, and hence will have a greater influence on the bulk density of the soil (i.e. the weight per unit volume). A decrease in the bulk density will reduce the volume of capillary pores that normally retain the available moisture. Certainly, the results in Table 13 suggest that sewage sludge has a more pronounced effect on moisture content held at both 0.05 and 15 bars than does the Doncaster refuse.

Trace elements

The part of this investigation dealing with trace elements and toxic metals aimed to ascertain whether levels of zinc, copper, lead, cadmium and boron in the refuse were acceptable, bearing in mind the after-use of treated land. It also aimed to investigate changes in metal availability in the soil and plant uptake during the decomposition of the refuse. These two areas of investigation will be treated separately. Detailed discussion on toxic metal contamination in soil, relative availability, and needs and criteria for guidelines have been omitted, but can be found elsewhere (Williams, 1980; D.O.E.,1980).

Recorded levels of zinc for all treatments at every sample time (Table 14) fall below the acceptable levels of 280 - 560 μg/g available zinc for areas classed as "public open space" (e.g. parkland and informal recreation areas) (tentative guidelines, D.O.E., 1980). 'Available' zinc in these guidelines is based on its extractability with 0.5 M acetic acid which recovers approximately 70% of that extracted by 0.1 M HCl (Neuhauser and Hartenstein, 1980); the results reported here, therefore, fall well within the safety limits. Williams (1980) noted that several root crops and horticultural species showed toxicity symptoms at 'extractable' zinc levels above 100 μg/g. Initial levels in the plots containing sewage sludge were approaching this level, though there was no visible evidence of phytotoxicity in any of the species growing on the plots. This may be due to the preponderance of grasses and weeds which are known to be considerably more resistant to zinc than most vegetable and horticultural species (Giordano and Mays, 1977; D.O.E., 1980).

Recorded levels of soil extractable copper (Table 16) were again well below the D.O.E. tentative guidelines of 140 - 280 μg/g for amenity grassland or public open space.

The 'zinc equivalent concept' has sometimes been used as a measure of phytotoxicity, particularly relating to sludge application on land. The toxic effects of zinc, nickel and copper are considered to be additive; 1 part nickel being equivalent to 4 parts copper and 8 parts zinc [i.e. Z.E. 1 (Zn) + 2(Cu) + 8(Ni)]. However, recent work has shown that the comparative toxicity of nickel in particular is higher for some crops than was previously assumed, but there is insufficient data on which to base a change. Consequently, these three elements are now treated separately (D.O.E., 1980). Nickel levels were not measured in this investigation due to limited time and resources.

D.O.E. guidelines for acceptable levels of 'total' lead are

1500 µg/g for amenity grassland and 2000 µg/g for public open space. The proportion of 'total' lead which is acid soluble varies widely, from about 10% to 90% or more (D.O.E., 1980) so that it is not really relevant to estimate what proportion of the total has been extracted in the data reported here (Table 18). However, it is worth noting that even if the figures represent only 10% of the 'total' lead, they still fall well within the guidelines. In addition, the suggested guidelines can be compared to those in the Toys and Graphic Instruments (Safety) Regulations (1974) which give a maximum of 250 µg/g 0.1 M HCl extractable lead; these relate to direct ingestion of lead. Again the reported lead levels in the treated soils were well within this limit.

The tentative guidelines for cadmium (total) are 12 µg/g for amenity grassland and 15 µg/g for public open space. The Doncaster refuse separates, therefore, seem to be safe with respect to cadmium (Table 20) but if repeated applications of the sewage sludge used in this investigation were required, the land should be carefully monitored since levels of up to 11 µg/g were recorded, even at the relatively low dosage of 20 tonnes/hectare.

The most serious threat of contamination which may result from the land application of the Doncaster refuse putrescibles seems to arise from the high levels of boron found in the material. The D.O.E. gives a guideline of 6 µg/g extractable boron for areas to be used as amenity grassland or public open space, although 3 µg/g can cause phytotoxicity. Table 22 shows that over 5 µg/g water extractable boron was recorded, and does indicate that it is an area for concern, especially since the application rate here is low. The repeated application in Treatment 3 increased the boron level quite significantly and repeated dosages of the refuse could cause problems, although leaching of this highly soluble element is rapid.

One of the consequences of organic waste application to soil and its subsequent humification is the ability to hold trace elements and metals by the increased cation exchange capacity. More important than the cation exchange rate with respect to the toxic 'heavy' metals (e.g. zinc, lead and copper) is the chelation effect in which heavy metals occur in an insoluble, stable combination with humic substances in the humus (Webber and Doyle, 1975; Kirkham, 1977). This has lead to the practice of applying waste materials, rich in organic matter to reduce plant uptake of toxic metals even where the waste materials themselves may contain large amounts of heavy metals (Kirkham, 1977).

Webber (1977) states that the generally accepted view is that as organic matter declines, metal availability may increase. This may well be an oversimplification, since the various metals tend to behave slightly differently in the soil. For example, there is an accepted order for decreasing chelating tendency of heavy metals, which is $Cu^{2+} > Ni^{2+} > Co^{2+} > Fe^{2+}$ with Mn^{2+} and Zn^{2+} having the least tendency (Leeper, 1972; Kirkham, 1977; Webber, 1980). Zinc is considered to be more tightly bound to organic matter than cadmium (Kirkham, 1977), but it has been reported that the retaining power of organic matter for cadmium is predominantly through its cation exchange capacity (Haghiri, 1974; Webber, 1980). Changes in metal availability associated with organic matter decomposition will, therefore, depend to a large extent upon the strength and form by which it is held to that organic matter. Other influences will come from the soil type. Gaynor and Halstead (1976) amended two sandy loam soils and one clay soil with sludge and found that metal extractability in sandy loam soils was not greatly changed after 11 months' incubation, but extractable cadmium, copper, lead and zinc were reduced in the clay soil following incubation. Wetting and drying of soil has also been shown to affect availability by increasing the complexing of zinc with organic matter (Keefer,1975; Kirkham, 1977).

The nature of the organic material itself, when added to soil will also affect the behaviour of the metals present within it possibly due to indirect effects on soil properties, such as acidity. Chaney (1975), for instance, showed that plants growing on composted sewage sludge accumulated less cadmium and zinc than those growing on digested sludge. In this investigation it was shown that the zinc in the Doncaster refuse was more available than that in the sewage sludge. This agrees with King _et_. _al_. (1974) who found that uptake of zinc by corn from treatments containing shredded refuse was greater than those containing sewage sludge although equal quantities of zinc were supplied by each treatment. It does, however, appear to conflict with El Bassam and Thorman (1979) who found that zinc concentrations in potato, rye and oats were strongly influenced by the rate of application of urban sewage sludge, but not by refuse compost, applied in 3 year intervals up to a total amount of 965 tonnes dry matter per hectare.

Many workers have reported increased levels of zinc in soil and plant tissues following applications of non-composted refuse (King et. al.,1974, 1977; Cottrell, 1975) and composted refuse (Purves and Mackenzie, 1973; Duggan and Wiles, 1976; Gray and Biddlestone, 1980; de Haan, 1981). Mays et. al. (1973) applied municipal composted refuse to a silt loam at dosage rates up to 327 tonnes/hectare and their soil and plant analyses indicated that potentially toxic amounts of zinc could accumulate in the soil if compost was applied at rates totalling several hundred tonnes/hectare over a few years. There seems to be little reference, however, in these reports to changes in availability of zinc as the organic matter in the refuse declines.

The present investigation has shown that extractable zinc in the soil does not change markedly over time as the waste materials (refuse separates and sewage sludge) degrade, but that there is a decline in plant tissue levels from the first to the second growing season. This agrees with the findings of Cottrell (1975) who showed that zinc content of fescue and alfalfa grown on a loamy fine sand soil amended with shredded municipal waste (250 or 500 tonnes/hectare) increased dramatically compared with the controls. However, the levels decreased from 200 μg/g to 50 μg/g in the second growing season. Similar results were found by Hinesly et. al. (1976) in their investigations on soybean responses to zinc and cadmium from sewage sludge amended soils. They noted that as long as sludge applications were made on an annual basis, zinc accumulated in the soil from the previous years was maintained in forms available to plants. When annual sludge applications were terminated zinc contents were significantly decreased in plant tissues. The organic constituents of the sludge, mainly fats, waxes and oils were found to provide little protection to plants against zinc in the soil and so these workers rejected previous theories that organic matter supplied as a constituent of sewage sludge chelates toxic metals and makes them less available to plants.

It is likely that zinc, present in available forms in both the refuse and sludge 'reverts' in time to chemical forms less available to plants. Reversion has been clearly established for zinc and can be quite rapid, although the process is not fully understood (Epstein and Chaney, 1978).

It is suggested therefore that unless repeated applications of the Doncaster refuse are envisaged for an area of land, phytotoxic effects from zinc are unlikely to be a problem.

Copper in soil tends to be very stable over time and is known to be strongly adsorbed on to organic matter (Kirkham, 1977). It forms highly stable complexes with humic and fulvic acids (Stevenson and Fitch, 1981). Large applications of copper contaminated sludges have not given any significant increases in copper uptake by cereals and vegetable crops (Williams, 1980).

The results in Section 3.1.1 showed that although the waste materials did increase extractable copper levels in the treated plots, there was virtually no change in availability over time. Plant uptake was not increased by the waste additions. Cottrell (1975) also found that the copper content of fescue and alfalfa was not affected by addition of shredded municipal waste to a loamy fine sand soil. King et. al. (1974; 1977), however, in both field and lysimeter studies did find increased levels of copper in corn (maize) stover on treatments containing shredded domestic refuse, and liquid sewage sludge. There was no effect however of the treatments on copper concentrations in corn grain. Duggan and Wiles (1976) also noted increased copper concentrations up to three times the control level in corn leaves following municipal compost application of up to 200 tons/acre (500 tonnes/hectare), but no differences were noted in the grain. It would seem, therefore, that exclusion mechanisms exist in various parts of the plants to prevent copper reaching, in this case, the corn grain. Different plant species have also been shown to accumulate copper to a greater or lesser extent. Gray and Biddlestone (1980) applied municipal refuse compost at a rate of 250 tonnes / hectare / year and noted trends of increased soil levels of copper from 1968 - 1970, and a corresponding increase in concentration of copper in dwarf beans, lettuce, and potatoes; concentrations decreased, however, over time in spinach beet. Other studies on refuse compost application have shown increases of up to 300% in soil copper concentrations but a mere 10% increase in vegetable crop uptake (de Haan, 1981). It would seem that reactions between the refuse, sewage sludge and soil do indeed transform the added copper to relatively unavailable forms, transformations which have been shown to happen following sewage sludge application to land (Soon et. al., 1980).

Increased levels of lead in soils to which municipal refuse and refuse composts have been added have been reported by several workers (King et. al., 1974; 1977; Gray and Biddlestone, 1980; de Haan, 1981). The present investigation has indicated that uptake of lead by couch grass was unrelated to levels in the soil, but that extractable lead decreased during the 28 months of decomposition. The lack of uptake of lead would be expected since this element is known to be highly insoluble in soil and is normally unavailable to plants. When added with sewage sludge lead is not considered to be phytotoxic because the sludge contains large amounts of phosphate that "tie up" the lead and prevent transport in and injury to plants (Epstein and Chaney, 1978). Lead concentrations in corn and bromegrass were not found to be significantly increased by applications of 310 kg lead/hectare from digested sewage sludge (Soon et. al., 1980). The Doncaster refuse also contains high levels of phosphate (Appendix B) and so a similar "tying up" of the lead may be expected.

Other workers have found a similar low availability of lead in domestic refuse applied to the land. King et. al. (1974) found no increase in lead uptake by corn grain on plots treated with shredded refuse alone or in combination with sewage sludge at a dosage of 188 tonnes/hectare. By doubling the dosage rate, however, they found a 70% increase in stover lead concentration over the control. Gray and Biddlestone (1980) found that applications of municipal refuse compost increased lead levels in lettuce, potatoes and sugar beat, but not in broad beans. de Haan (1981), again investigating the effects of compost application, found that a 400% increase in soil concentration of lead resulted in an increase of only 10% more lead in the vegetable crops grown on treated soil.

None of these workers described what, if any, changes in availability occurred as the added organic matter degraded. It is likely that binding by phosphates and other pH dependent reactions will be more important in controlling lead uptake than the presence of organic matter. Zimdahl and Foster (1976) believed that organic matter additions did not offer promise as a method of reducing lead availability to plants. They said that massive amounts of organic matter would be required to exert an effect on lead accumulation by plants.

Cadmium in organic wastes added to soil tends to be very available to plants and numerous instances of phytotoxicity have been reported following sludge application to land. Anderson (1977), for example, found that the cadmium content of crops was approximately doubled when the content in the soil was trebled. Its behaviour seems to be similar to zinc in that extractable levels in soils can be correlated with those in plant material (Webber and Doyle, 1975; Soon et. al., 1980). There is also evidence that cadmium acts antagonistically with zinc in soil (D.O.E., 1980) although Haghiri(1974) found that soil applications of zinc as an agent in reducing the uptake of cadmium by soybean tops did not prove to be practical since the suppression of cadmium occurred only when large amounts of zinc were added. At these levels plant yield was drastically reduced.

Consideration of the results from this investigation (Tables 20 and 21) is complicated by the fact that the cadmium concentration of the refuse was relatively low, and its distribution in the treated plots was very patchy. The sewage sludge treated plots had greater concentrations of cadmium and all increases over the control were reflected in larger plant tissue concentration.

Large increases in cadmium concentration in vegetable crops grown on pulverised domestic refuse (containing 5 μg/g cadmium) were found by A.D.A.S. in their work at High Wycombe. They noted levels twice those recommended in the D.O.E. guidelines for cadmium in foodstuffs. Mixing the refuse with soil decreased the plant concentration slightly. (unpublished data - personal communication) King et. al. (1974) reported increased levels of cadmium in corn growing on plots treated with shredded domestic refuse. With regard to refuse compost, Duggan and Wiles (1976) noted a 10-fold increase in soil cadmium levels at an application of 200 tons/acre (500 tonnes/hectare) resulting in a two-fold increase in concentration although de Haan (1981) recorded a 3-fold increase in soil levels, but no increase in crop cadmium levels.

Again there appears to be little information regarding changes in cadmium availability over time. However, Hinesly et. al. (1976) investigated soybean responses to cadmium and zinc in sewage sludge and found the 2 elements to behave similarly. Their results have been discussed previously, and seem to bear out the observation reported in Section 3.1.1 that plant uptake of cadmium decreases once the application of sludge and refuse has ceased. The reasons for this are not well understood. Results of the water-extractable boron analyses will be discussed in Section 4.1.2.

3.1.3 Conclusions

Decomposition of the Doncaster refuse putrescibles followed an exponential pattern and they appeared to be in a relatively stable state after the first year. Rate of breakdown was not increased by the addition of a nitrogen-rich sewage sludge and if anything the refuse was more amenable to breakdown than the sludge which had already undergone a certain amount of biodegradation and stabilisation during digestion. This increased stabilisation of the sludge was also apparent in the higher levels of humic acid build up on the sewage sludge treated plots; the more rapid breakdown of the 'fresher' refuse putrescibles caused more of the organic matter to be lost as carbon dioxide and ammonia, and less to remain as humified material.

The refuse did not appear to influence soil pH but soil moisture content was increased. More water was held at permanent wilting point (15 bars) but there was no change in overall available water capacity. The reduced bulk density following the addition of the refuse may have influenced the water holding properties of the soil. The sewage sludge, which contained less bulky material than the refuse, had a smaller effect on bulk density but a more pronounced effect on water holding properties.

No visible symptoms of metal toxicity were noted on any of the waste treated plots, and levels of zinc, copper and lead fell within acceptable limits. Cadmium and boron, on the other hand, did reach levels which may be considered dangerous, boron being more of a phytotoxic hazard, but one which is relatively short-lived. Contamination from cadmium was more evident on the sewage sludge treated plots, although the double dosage treatment of refuse (Treatment 3) did reach 4.3 μg/g extractable cadmium. Careful monitoring of these two elements would be essential following application of the Doncaster refuse to land, particularly if it is in combination with sludge containing high levels of cadmium.

Copper and lead exhibited a low availability to plants, copper probably formed complexes with organic matter and lead was possibly bound by phosphate. Zinc and cadmium were initially more available to plants but this availability was reduced over time, probably due either to increased organic matter binding, or 'reversion' to unavailable forms.

Conclusions drawn from the analyses of toxic metals and plant uptake should be considered with caution, and may only apply to the site used in the field trials. Whilst the general behaviour of many of the toxic metals can be, and have been, identified, specific levels of 'available' metals will be governed not only by organic matter, but also soil pH, redox potential, macronutrient status and cation exchange capacity. It is possible that some of the effects of organic waste addition noted in this investigation may reflect indirect influences on these soil properties. In addition, plant analyses for toxic metals are fraught with difficulties such as problems of timing, and dilution effects over the growing season.

"Total" metals in soil can also vary. They may be reduced by leaching processes, or increased (per unit volume of soil) by a concentration effect during organic matter decay, or by aerial deposition

(D.O.E., 1980). Guidelines for acceptable levels of metals in soils, therefore, can only be considered tentative, and will depend heavily on the suggested after-use of the land, and the relative sensitivity of the vegetation it will support. Moreover, concentrations exceeding the guideline values are not necessarily 'unacceptable', but they may be 'undesirable'.

No work was included here on the movement of the toxic metals in soil profiles, although other work has shown that mobility of the metals in municipal compost is minimal even under severe leaching conditions (Giordono and Mays, 1977). Emmerich et. al. (1982) found no evidence of movement of metals out of the top sludge-soil layer following sludge application; they noted, however, that mineralisation of organic nitrogen decreased the pH value in the soil below the mixed layer and suggested that if this pH reduction spread to the mixed layer it could increase metal solubility. Based on these findings, it is unlikely that disposal rates of municipal wastes applied to land pose a threat to the water supply, at least in terms of heavy metals.

3.2 Ecological aspects of soil disposal of refuse separates

Preliminary investigative work on the ingress of ruderal weeds and other plants was carried out in August 1979 in an attempt to note differences in plant establishment with soil treatment, and to quantify species composition. The information thus derived may be helpful to the successful management of an area reclaimed by the addition of refuse separates to the soil.

Plants such as cherry (*Prunus avium*), canary grass (*Phalaris canariensis*) and tomato (*Lycopersicon esculentum*) were found and were obviously introduced in the waste materials. Species characteristic of the sewage sludge treated plots were those known to flourish in a high nitrogen environment, such as tomato and nettle (*Urtica dioica*). In contrast, nitrogen-fixing species such as hop trefoil (*Trifolium campestre*) were found only on the soil control plots where the nitrogen status was much lower.

The more acidic environment of the colliery shale supported some notable calcifuges such as sheep's sorrel (*Rumex acetosella*) and black bent (*Agrostis gigantea*). Species tolerant of soil compaction, such as plantains (*Plantago sp.*) were also noted on the colliery shale plots. Oxford ragwort (*Senecio squalidus*), a plant whose native environment is the ash strewn slopes of Italian volcanoes (Darlington, 1969) was found growing on plots amended with refuse and colliery shale (both with a high ash content).

Notably salt-tolerant species of the *Chenopodiacae* were commonly found on the refuse amended plots, along with other nutrient demanding weeds such as shepherd's purse (*Capsella bursa-pastoris*) and groundsel (*Senecio vulgaris*) both of which are commonly found on disused refuse tips (Bradshaw and Chadwick, 1980).

During the first few months after incorporation there was still scope for each component of the multi-treatment plots to exert an influence on species composition. Later on, dominance of species such as couch grass (*Agropyron repens*) and spreading orache (*Atriplex patula*) became more apparent. Species density began to decrease over time during the summer following the pattern proposed by Grime(1977).

In order to quantify species density a point quadrat method of recording was employed. Percentage cover of the five major species and bare ground are shown in Table 24. The "dominance index" of each of the species as proposed by Grime (1973; 1979) is also included. This index relates to the plant's ability to attain dominance over its neighbours through attributes such as high potential growth rate and persistence of litter.

Couch grass and spreading orache were initially dominant on all treatment plots. Nipplewort (*Lapsana communis*) and corn poppy (*Papaver rhoeas*) occurred characteristically on the low nutrient status plots (soil and soil plus colliery shale) but had little competitive ability with, for example, spreading orache, on the more fertile refuse and sewage amended plots. Bare ground was encountered only on the soil control, and colliery shale amended plots. The flora of these plots showed a distinct yellowing, characteristic of nitrogen deficiency. Investigation of the release of plant available nitrogen from the refuse separates is discussed in more detail in

	Dominance index *	Treatment							
		1	2	3	4	5	6	7	8
<u>Agropyron repens</u>	6.5	49	51.8	63.2	23.6	33.6	30.4	17.6	46.2
<u>Atriplex patula</u>	2.5	11.2	32.4	14.8	50.0	14.8	31.2	43.9	14.4
<u>Lapsana communis</u>	2.5	11.6	2.4	3.6	7.4	13.2	3.2	2.8	12.0
<u>Polygonum aviculare</u>	1.5	2.8	5.6	3.2	6.4	3.2	6.8	4.4	9.2
<u>Papaver rhoeas</u>	1.5	11.2	1.2	5.6	2.0	2.8	3.2	3.6	1.2
Bare ground		9.6	0	0	1.3	16.4	0	0	0
Total %		95.4	93.4	90.4	90.7	84.0	74.8	72.3	83

<u>TABLE 24</u>. <u>Percentage cover by selected plant species on treated plots, August 1979</u>.
(Figures are means of six replicates.)
(* See text for explanation)

Sections 3.3, 4.2 and 4.3.

Quantification of productivity (measured after 18 months) is recorded in Table 25.

A two-way analysis of variance showed significant difference between the treatments for vegetational biomass ($p < 0.001$) with a least significant difference of 13.98. This effectively groups the treatments into the less productive control and colliery shale plots, and the more productive refuse, and sewage sludge amended plots. It would seem, therefore, that the addition of the refuse separates and sewage sludge to the soil increases vegetational biomass. The addition of refuse to sewage sludge and colliery shale improves productivity over those treatments where these additives are applied singly to the soil.

Twelve months later, in July 1981, a rough estimate of long term productivity was provided by the collection of 10 individual shoots of couch grass (*Agropyron repens*) from each treatment. The dry weights were recorded and the data is presented in Table 26.

The refuse treatments (2 and 3) by this time showed significantly higher productivity, measured in terms of couch grass shoot yield. This would suggest that the beneficial effects to plant growth of the Doncaster refuse are longer lasting than those of the digested sludge used in this investigation, although competition from other species growing on the plots may have distorted these results.

3.3 Soil/refuse mixtures as media for plant growth

3.3.1 Results of growth room analysis

Results of growth room plant growth experiment are depicted graphically (Figures 6 and 7). The graphs show increasing dry weight of barley over time, and the growth function Relative Growth Rate ($d \text{ Log } Y/dT$, where Y = shoot dry weight in grams and T = time in days). The data from which the graphs are derived is given in Appendix C. Mathematical smoothing of the curves was performed by regression analysis. Plate 3 shows differences in yield of barley after 71 days.

Plant growth curves typically show three phases:-

Treatment	Biomass (g.) $\bar{x}$ of 6
1	22.593
2	40.797
3	55.155
4	26.542
5	15.285
6	44.683
7	41.355
8	47.632
p	0.001
l.s.d.	13.98

l.s.d. = least significant difference.

TABLE 25. Plant biomass occurring within a 25 cm^2 quadrat on treated plots (August 1980).

Treatment	Dry weight $\bar{x}$ of 10
1	0.87
2	1.32
3	1.52
6	0.87
7	0.99
8	0.84
p	0.001
l.s.d.	0.339

l.s.d. = least significant difference.

TABLE 26. Shoot dry weight of couch grass growing on treated plots (July 1981).

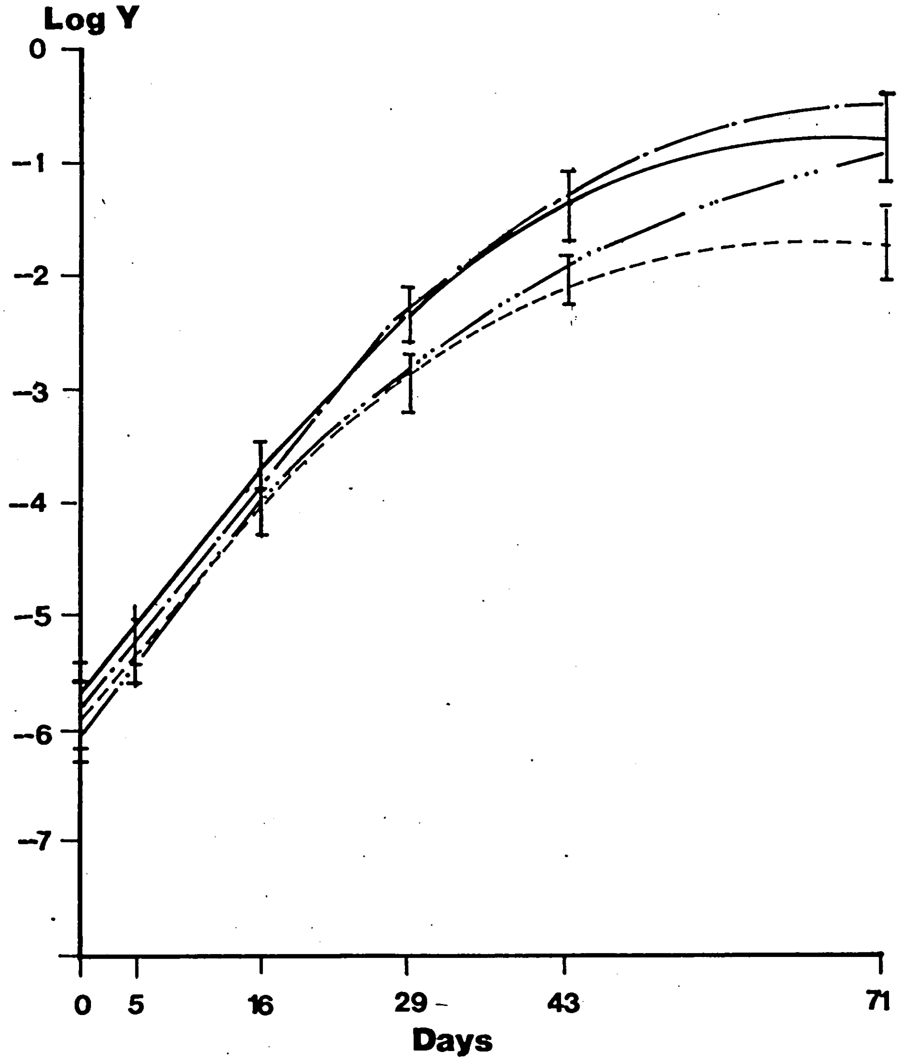


FIG.6
Plant dry weight increase with different soil additives (pot experiment)

-----	=	**Soil Control** (+confidence limits)
--···--	=	**Soil + Refuse**
——	=	**Soil + Sludge** (+confidence limits)
—·—	=	**Soil + Refuse + Sludge**

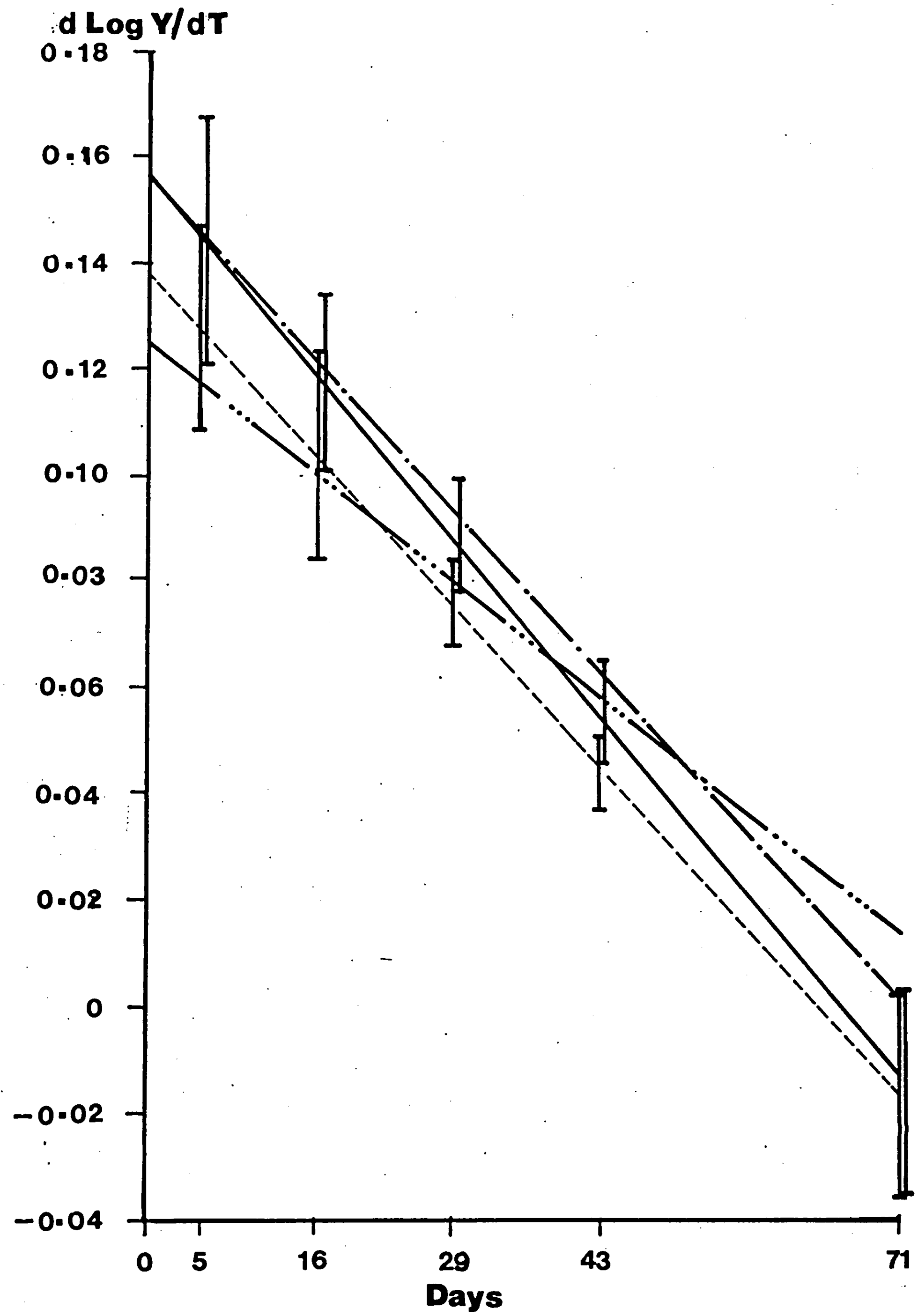


FIG.7
Relative growth rate (pot experiment)

KEY

- - - -	= **Soil Control** (+ CONFIDENCE LIMITS)
—··—··—	= **Soil + Refuse**
———	= **Soil + Sludge** (+ CONFIDENCE LIMITS)
—·—·—	= **Soil + Refuse + Sludge**

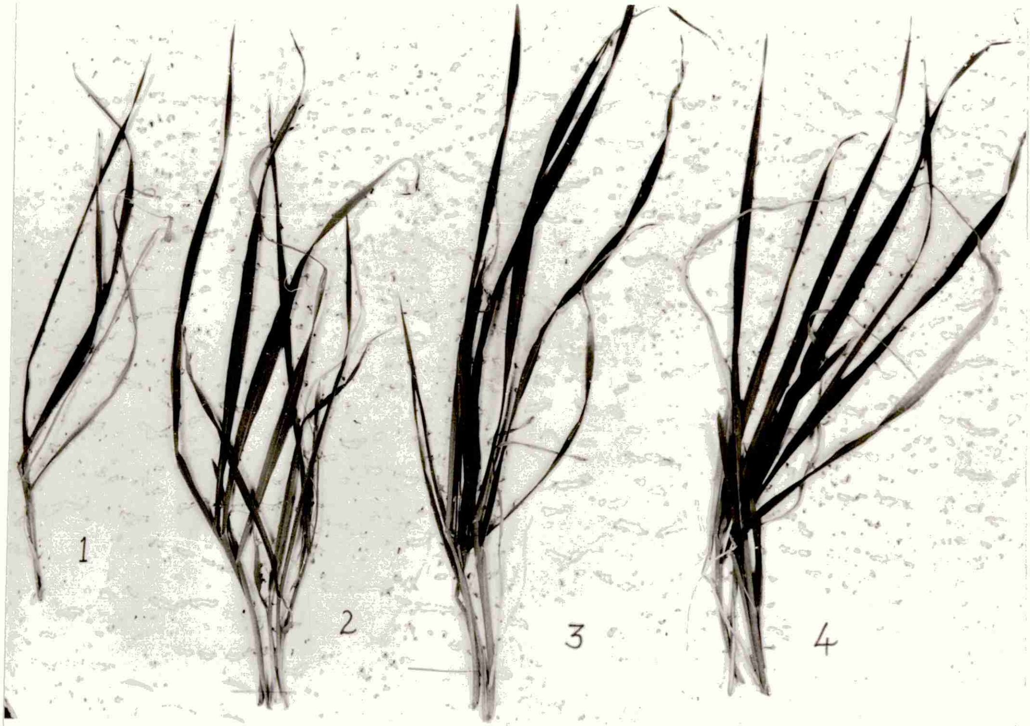


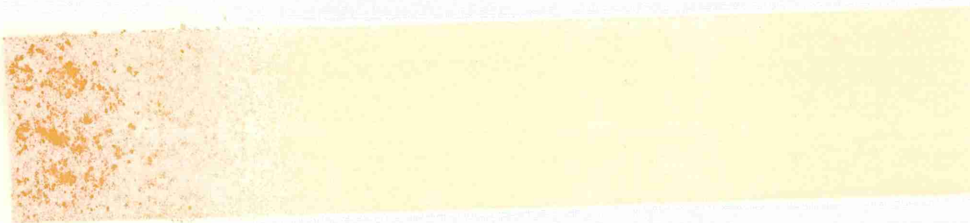
PLATE 3. Yield of barley (*Hordeum vulgare*) after 71 days (pot experiment).

Treatment: 1. Soil control.

2. Soil and refuse.

3. Soil and sewage sludge.

4. Soil, refuse and sewage sludge.



(1) Initially no change in dry weight, then a drop.

(2) "Grand" period of growth with substantial carbon assimilation and continuous dry weight increase.

(3) Net dry weight increase ceases.

The data here cover only stages two and three since the preliminary phase covers the initial stages of seed germination and seedling growth for which no data was recorded.

Figure 6 indicates that plant yield (in terms of dry weight) was significantly higher in the treatments containing sewage sludge (3 and 4) than in the controls, or the soil plus refuse treatment. Only in the final stages of the experiment was the refuse treatment significantly different from the control. Relative growth rates in Treatments 3 and 4 were also significantly higher than the control initially, but slowed to a similar level by the time of the final harvest, having reached the stable stage (phase three) of the generalised growth curve. Treatment 2, on the other hand, shows a less steep curve than the others, ending with a higher relative growth rate at the final harvest. The constant growth phase does not seem to have been reached by the plants in this treatment.

Soil samples from the pots were taken at regular intervals throughout the growth experiment and analysed for inorganic nitrogen. The results are shown in Tables 27 and 28.

At the start of the experiment nitrate-nitrogen was significantly lower, and ammonium-nitrogen higher in the refuse than in the sludge which had previously been digested. For the first six weeks nitrate-nitrogen in Treatment 2 was less than the control (Treatment 1) which indicates that there may have been some immobilisation of this inorganic nitrogen due to the high C:N ratio of the fresh refuse. It may, however, simply reflect an inhibition of nitrification of the ammonium-nitrogen due to environmental conditions such as high moisture content. After this time, mineralisation and nitrification produced amounts of nitrate-nitrogen similar to those in the sewage sludge. This pattern of release of available nitrogen corresponds well with the variation in plant growth rates.

3.3.2 Discussion of results

The lack of available data for the very early stages of growth was not considered critical for the interpretation of final growth differences, since it was important to ensure minimal failure of plant samples. However, the addition of fresh organic refuse to soils can result in poor germination due to high levels of toxic salts (Stewart and Webber, 1976; A. E. Higginson - Personal Communication). No specific work on germination was carried out in this investigation.

Many investigators have reported increased yields following the land application of refuse (King et. al., 1974; 1977) and refuse composts (May et. al., 1973; Duggan and Wiles, 1977; Gray and Biddlestone, 1980). Sewage sludge has been used as a fertiliser for centuries (D.O.E., 1977b; Sabey, 1981). It would appear from this study that the Doncaster refuse separates encouraged increased plant yields by an amount comparable to that for digested sewage sludge. The enhanced growth seemed to be related to the rate of production of plant available nitrogen, particularly nitrate-nitrogen. Nitrification was retarded slightly in the refuse treated pots due to the high C:N ratio of the fresh refuse. This meant that levels of nitrate-nitrogen and plant response were initially lower than in the pots treated with the more stable digested sewage sludge. After about six weeks the C:N ratio in the refuse treated pots had stabilised sufficiently to allow for mineralisation and nitrification. Sabey (1980), also noted reduced levels of nitrogen mineralisation following the addition of fresh organic wastes to soil when compared with well digested wastes such as sewage sludge.

Mixing the refuse separates and sewage sludge together provided no significant advantage over applying them separately. It may be that the Doncaster refuse alone would provide the more fertile treatment in the long-term since plant response for this treatment had begun to overtake that for the sewage sludge by the end of the ten week experiment.

The plant responses discussed in this section seem to verify those found from the field biomass data. These findings agree with those of King et. al. (1974; 1977) who showed in both field and lysimeter

Time (Days)	Treatment 1	2	3	4	p	l.s.d.
1	42.1	51.1	47.2	58.2	0.002	7.4
16	35.4	22.5	12.3	14.9	0.01	12.2
43	10.4	23.7	16.9	17.0	0.01	7.0
71	10.8	16.1	14.4	13.6	n.s.	-
120	0	10.7	15.5	12.1	n.s.	-

l.s.d. = least significant difference.

TABLE 27. Soil ammonium-nitrogen (μg/g). Plant growth experiment.

Time (Days)	Treatment 1	2	3	4	p	l.s.d.
1	24.3	20.2	113.2	87.3	0.001	33.2
16	30.0	28.5	123.7	94.8	0.001	37.5
43	34.6	156.6	233.7	167.2	0.01	96.7
71	27.8	108.8	135.0	190.4	0.02	72.8
120	47.9	177.1	141.1	198.9	0.002	96.2

l.s.d. = least significant difference.

TABLE 28. Soil nitrate-nitrogen (μg/g). Plant growth experiment.

studies that unsorted, shredded refuse applied at a dosage of 188 tonnes/hectare produced significantly higher corn yields than liquid sewage sludge, alone, or in combination with shredded refuse. Application of municipal waste with much higher C:N ratios than reported here, however, may result in little change, or even a depression of yields unless supplemented nitrogen is added. Cottrell (1975) found that fescue and alfalfa yields were not changed markedly by the initial incorporation of up to 400 tons/acre (1000 tonnes/hectare) of shredded municipal waste with a C:N of 100:1 in a sandy soil. Maximum yields were obtained with application of 1,000 lb of nitrogen per acre (1121 Kg/hectare).

3.3.3 Conclusion

Both laboratory and field data support the finding that the Doncaster refuse putrescibles and digested sewage sludge, have a positive effect on plant yield when added to soil. Encouragement of plant growth appeared to be related to the production of nitrate-nitrogen during the breakdown of the waste materials. This production was initially slower in the refuse treatments where early decomposition of the carbonaceous material inhibited nitrification. Later on, the refuse began to produce more nitrate than the sludge, and a corresponding increase in plant growth was noted.

CHAPTER 4

EFFECT OF DOSAGE RATE

4.1 Decomposition and effects on the amended soil

Organic components from the test treatments were measured over an eighteen month period and particular emphasis was laid on understanding the breakdown and mineralisation of organic nitrogen in the test material. In order to facilitate comparison, the parameters and methods of analysis were similar to those used in the investigation of different additives.

4.1.1 Results of field trials

Biodegradation of organic materials

Results of analyses of ethanol/water soluble material which includes the readily decomposable fraction of the waste are shown in Table 29 and Figure 8. These indicate a very rapid decrease in rate of production, even during the first six weeks; this higher percentage decrease corresponding to the higher waste additions. Levels of these extractables had stabilised at less than 1% at all dosages only six months after initial incorporation. Since this material represents compounds which have been produced by the biodegradation of more complex organics, it would appear that the most rapid period of decomposition of the refuse separates had been completed by this time.

Table 30 shows the regression analysis of the data (taken as a percentage of the control to minimise seasonal fluctuation). This indicates some limitation in the rate of production of this readily available material at the top dosage. Up to the dosage of 47.5 tonnes/hectare the rate of production corresponded well with increasing organic loading.

Measurements of change in organic carbon content of the plots and regression analysis of the data are shown in Figure 9 and Tables 31 and 32. Rates of decrease were slower than for the more easily utilised ethanol/water soluble material although decreases of up to 65% during the first six months corresponded well with the 50 to 85% release of carbon from decaying food or composts quoted by Hartenstein

Time (Months)	Dosage (tonnes/hectare)				
	Soil Control	20	32.5	47.5	65
0.25	0.50 ± 0.19	1.47 ± 0.70	2.83 ± 0.58	3.53 ± 0.62	6.92 ± 2.05
1.5	0.30 ± 0.15	0.90 ± 0.38	1.07 ± 0.22	0.84 ± 0.50	1.43 ± 0.74
6	0.24 ± 0.04	0.12 ± 0.07	0.50 ± 0.22	0.33 ± 0.20	0.71 ± 0.27
12	0.1 ± 0.1	0.2 ± 0.02	0.20 ± 0.16	0.3 ± 0.15	0.7 ± 0.47
18	0.33 ± 0.15	0.2 ± 0.23	0.40 ± 0.26	0.28 ± 0.1	0.67 ± 0.36

TABLE 29. Percentage ethanol/water soluble material (dry weight basis).

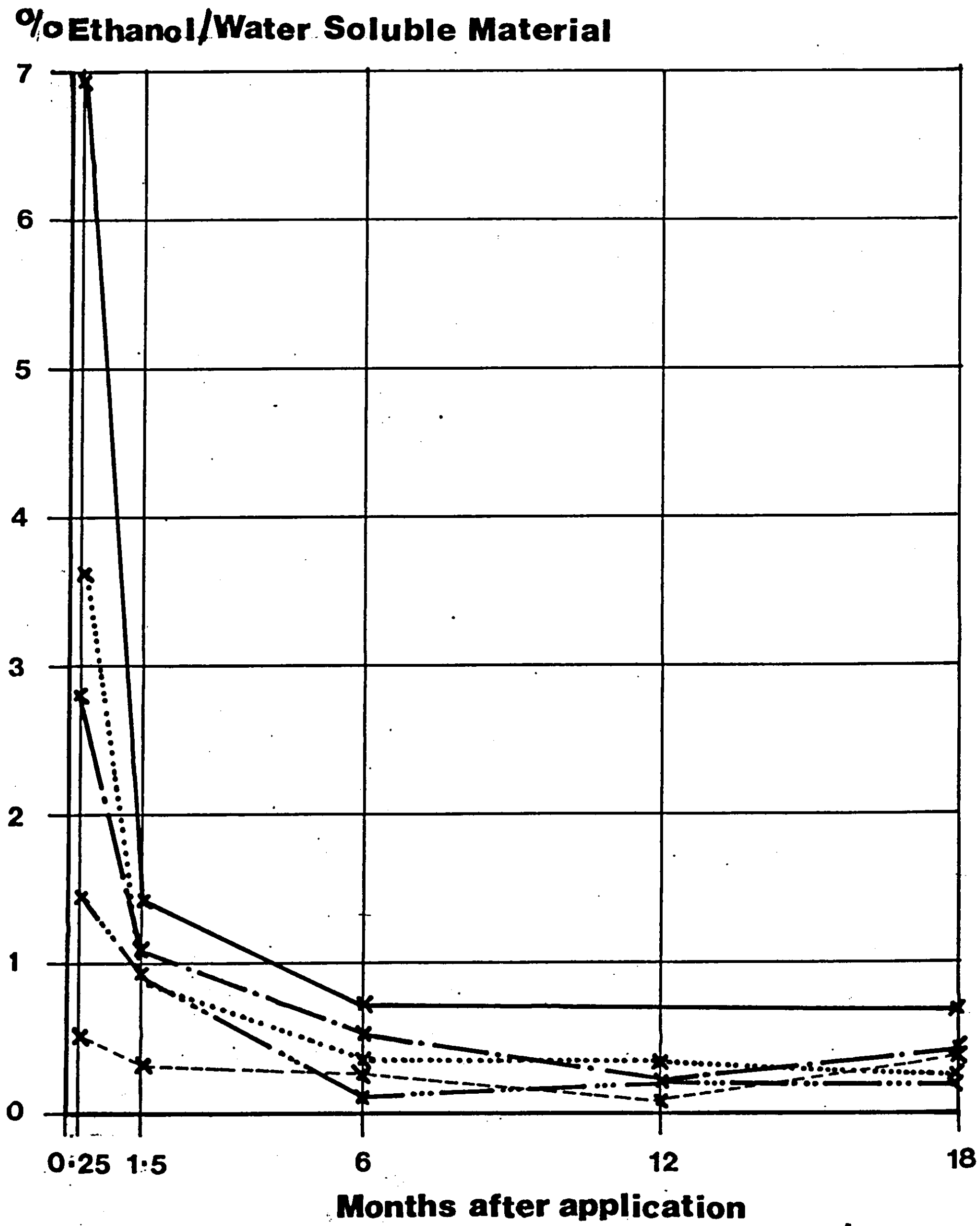


FIG.8

Changes in Ethanol/Water soluble material following refuse application at different dosage rates

KEY

— - — -	=	Soil Control
—···—	=	20t/hectare
—·—	=	32·5t/hectare
······	=	47·5t/hectare
——	=	65t/hectare

Dosage (tonnes/hectare)	b	r	p
20	- 1.11	- 0.77	n.s.
32.5	- 1.15	- 0.84	0.10
47.5	- 1.34	- 0.91	0.05
65	- 1.04	- 0.83	0.10

b = regression coefficient of the natural logarithm of percentage ethanol/water soluble material as a function of time (years).

r = correlation coefficient.

n.s. = not significant.

TABLE 30. Relationship between amount of ethanol/water soluble material in the treated soil and time at different dosages.

Time (Months)	Dosage (tonnes/hectare)				
	Soil control	20	32.5	47.5	65
0.25	2.64 ± 0.05	5.69 ± 0.75	9.34 ± 3.51	13.15 ± 3.17	17.32 ± 3.52
1.5	2.46 ± 0.13	4.15 ± 0.25	5.62 ± 1.13	7.11 ± 2.63	11.40 ± 2.50
6	2.68 ± 0.17	3.20 ± 0.28	4.02 ± 0.52	5.28 ± 1.34	6.04 ± 1.81
12	2.47 ± 0.26	3.05 ± 0.44	3.22 ± 0.37	4.47 ± 0.92	6.07 ± 0.79
18	2.83 ± 0.61	3.33 ± 0.70	4.59 ± 0.70	3.85 ± 0.43	7.31 ± 3.17

TABLE 31. Percentage organic carbon (dry weight basis).

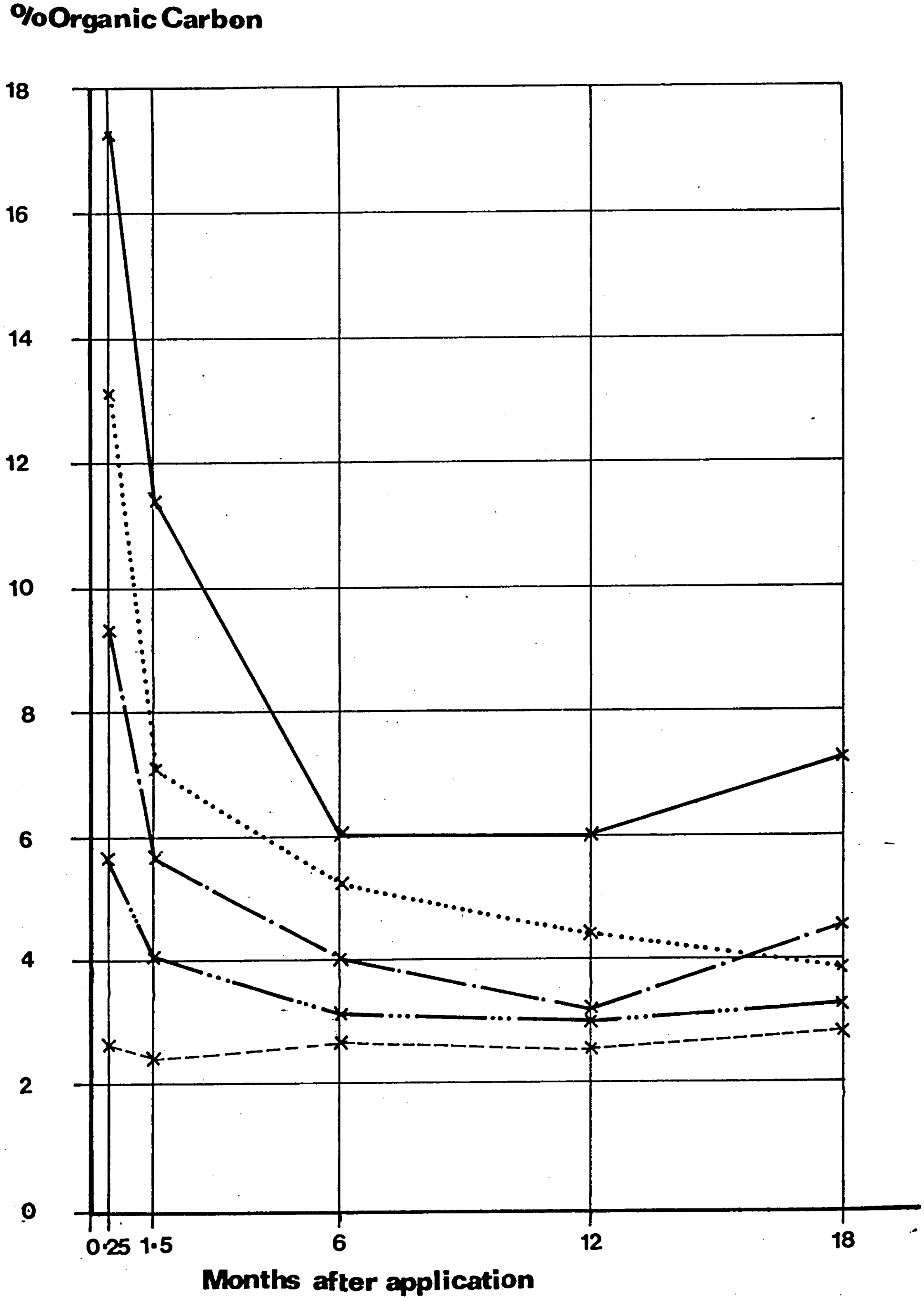


FIG. 9

Changes in soil Carbon content following refuse application at different dosage rates

KEY

= Soil Control

= 20t Refuse/hectare

= 32·5t Refuse/hectare

= 47·5t Refuse/hectare

= 65t Refuse/hectare

Dosage (tonnes/hectare)	b	r	p
20	- 0.33	- 0.80	0.10
32.5	- 0.48	- 0.72	n.s.
47.5	- 0.75	- 0.90	0.05
65	- 0.56	- 0.74	n.s.

b = regression coefficient of the natural logarithm of percentage carbon as a function of time (years).

r = correlation coefficient.

n.s. = not significant.

TABLE 32. Relationship between carbon decomposition and time at different dosages.

(1981) for a similar time period. After eighteen months all but the lowest dosage (20 tonnes/hectare) still showed values of organic carbon higher than the control soil. The slightly increased amounts at all dosages between twelve and eighteen months probably included plant material (roots, etc.) from the summer's growth period.

Table 32, indicates a pattern of increasing rate of decomposition up to the dosage of 47.5 tonnes/hectare and then a decrease at higher dosage rates which corresponds to that suggested by the change in ethanol/water soluble material (Table 30).

Results of organic nitrogen analyses (Table 33, Figure 10) show a slow but steady decrease although slight fluctuations appear to coincide with seasonal effects: the more rapid decreases occurred during the summer and autumn of the first and second year. After eighteen months all dosage treatments still showed levels of nitrogen above those of the control.

Humification

Incorporation of the Doncaster putrescibles into the soil increased the levels of humic acids (Table 34, Figure 11) and higher yields corresponded well with higher dosage rates. Fluctuations in humic acid levels occurred during biodegradation of the refuse but were small after the first six months and seemed to indicate a more or less stable state, similar to that seen for the additives trials. It seems, therefore, that the addition of the Doncaster putrescibles provides a long-term reserve of humic material in the soil, a suggestion which is supported by Table 35 in which the data is expressed as a percentage of the control. Humic acid levels were increased by up to 500% at the highest dosage as long as eighteen months after initial incorporation.

Soil pH

Table 36 presents the results of soil pH recorded at each sampling time. Values were always similar to those of the control, which would seem to indicate that application of the Doncaster putresibles even at the highest dosage applied here had little or no effect on soil pH.

Time (Months)	Dosage (tonnes/hectare)				
	Soil control	20	32.5	47.5	65
0.25	0.19 ± 0.01	0.31 ± 0.06	0 43 ± 0.11	0.53 ± 0.19	0.87 ± 0.16
1.5	0.19 ± 0.01	0.28 ± 0.03	0.41 ± 0.09	0.44 ± 0.11	0.81 ± 0.14
6	0.17 ± 0.01	0.25 ± 0.03	0.33 ± 0.09	0.41 ± 0.11	0.50 ± 0.09
12	0.17 ± 0.01	0.25 ± 0.02	0.27 ± 0.04	0.40 ± 0.09	0.55 ± 0.09
18	0.16 ± 0.01	0.20 ± 0.01	0.27 ± 0.03	0.28 ± 0.07	0.46 ± 0.20

TABLE 33. Percentage organic nitrogen (dry weight basis).

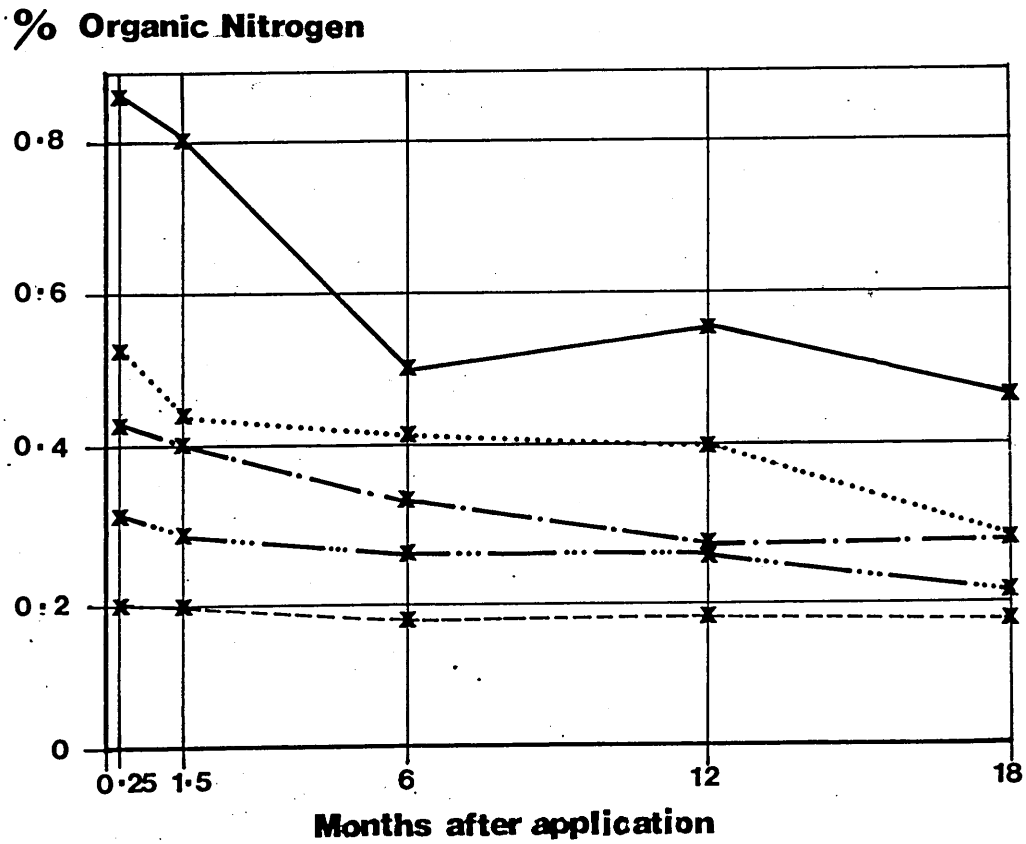


FIG.10

Changes in soil Nitrogen content following refuse application at different dosage rates

KEY

------	= Soil Control
—···—	= 20t/hectare
—·—	= 32·5t/hectare
·······	= 47·5t/hectare
——	= 65t/hectare

Time (Months)	Dosage (tonnes/hectare)				
	Soil control	20	32.5	47.5	65
0.25	0.50 ± 0.07	0.07 ± 0.18	1.36 ± 0.54	1.35 ± 0.29	2.52 ± 0.81
1.5	0.49 ± 0.05	0.79 ± 0.12	1.12 ± 0.57	1.43 ± 0.49	2.34 ± 0.71
6	0.49 ± 0.08	0.71 ± 0.07	1.05 ± 0.32	1.35 ± 0.27	1.57 ± 0.36
12	0.38 ± 0.06	0.70 ± 0.21	0.87 ± 0.24	1.24 ± 0.42	1.57 ± 0.29
18	0.38 ± 0.11	0.71 ± 0.28	1.01 ± 0.25	1.00 ± 0.23	1.92 ± 0.88

TABLE 34. Percentage humic acid content (dry weight basis).

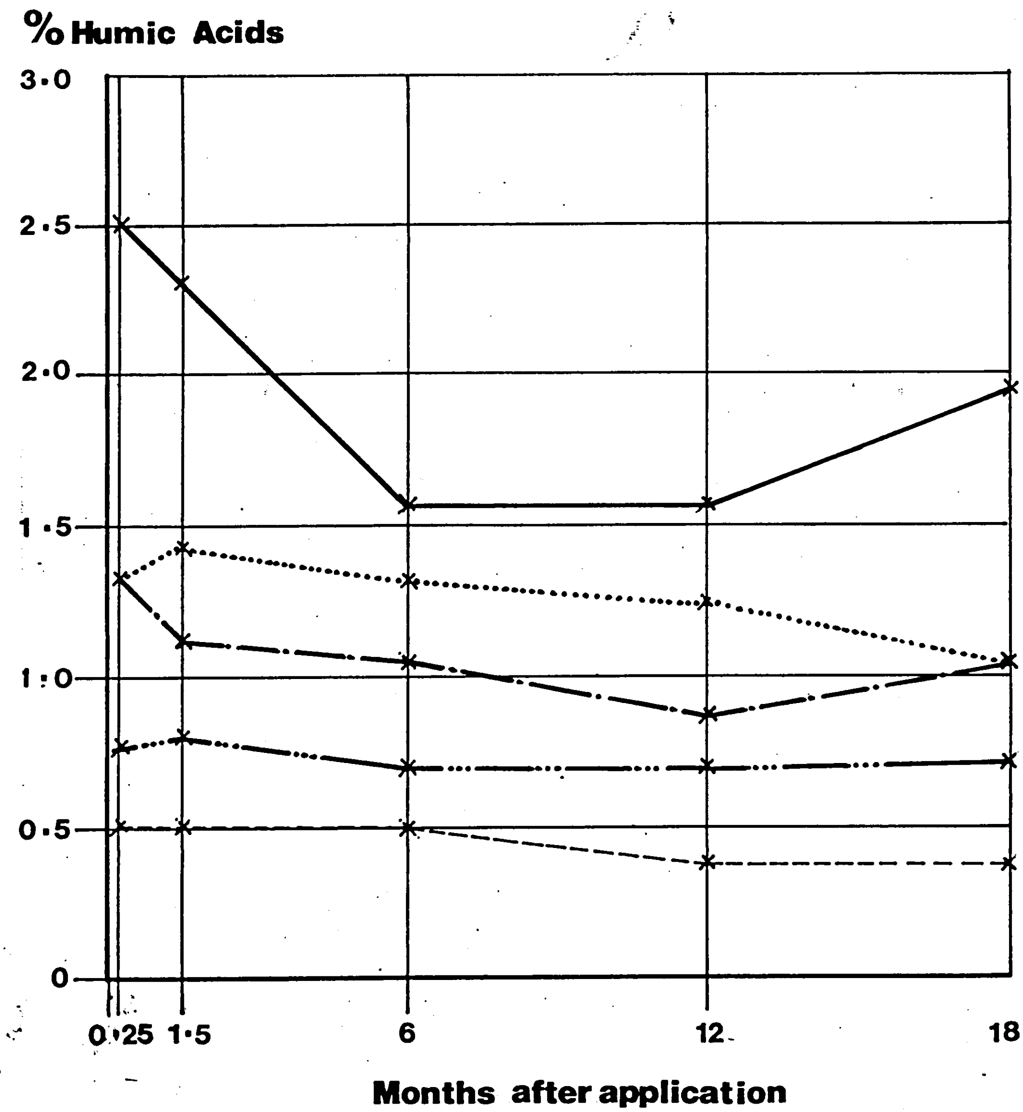


FIG.11
Changes in soil Humic Acid content following refuse application at different dosage rates

KEY

----	= **Soil Control**
—···—	= **20t Refuse/hectare**
—·—	= **32·5t Refuse/hectare**
·····	= **47·5t Refuse/hectare**
——	= **65t Refuse/hectare**

Time (Months)	Dosage (tonnes/hectare)			
	20	32.5	47.5	65
0.25	150	272	270	504
1.5	161.2	228.6	291.8	477.6
6	144.9	214.3	275.5	320.4
12	184.2	228.9	326.3	413.2
18	186.8	265.8	263.2	505.3

TABLE 35. Soil humic acid content expressed as a percentage of soil control at each sample time.

Time (Months)	Dosage (tonnes/hectare)				
	Soil control	20	32.5	47.5	65
0.25	7.72	7.43	7.32	7.50	7.88
1.5	7.60	7.43	7.96	7.89	7.93
6	8.13	8.01	7.87	7.61	7.61
12	7.92	8.03	7.87	7.93	7.81
18	7.73	7.80	7.79	7.82	7.61

TABLE 36. Soil pH.

Moisture Holding Properties

Percentage moisture in the plots was increased by all treatments one week after the refuse was incorporated, and was doubled at the highest dosage of 65 tonnes/hectare (Table 37). After eighteen months the increases, though smaller, were still notable.

Moisture holding capacity of the treated soil was measured on samples taken from the plots in March, 1982. The results are shown in Table 38.

Addition of the Doncaster refuse increased siginificantly the moisture held at 15 bars (permanent wilting point) at all dosages above and including 32.5 tonnes/hectare, but no significant differences were noted at field capacity(0.05 bars). From these two the 'available' water, or the overall storage capacity of the soil is seen to be decreased by the addition of the refuse at all dosage rates.

Trace elements

Measured levels of extractable zinc in soil samples from the increasing dosage plots, and total levels of zinc in shoots of ryegrass (*Lolium perenne*) are shown in Tables 39 and 40.

Addition of the Doncaster refuse increased levels of extractable zinc in the plots; the higher the dosage, the higher was the increase. The pattern of change over time was difficult to ascertain due to spatial variations with isolated 'patches' of high zinc but appeared to show very little change at low dosages, with a slight increase in extractable levels at higher dosages. These increased levels also reflect a concentration effect as biodegradation of the organic matter continues.

Plant tissue analysis again showed increasing levels of zinc corresponding to increasing refuse dosage rate. There appeared to be a concentration of the metal in the tissues during the first growing season, but uptake in the second year's growth (15 months) seemed to be slightly less than that in the first. Thus, although extractable levels in the soil had either increased slightly, or shown no change, plant uptake had not. There are two possible explanations for this;

Time (Months)	Dosage (tonnes/hectare)				
	Soil control	20	32.5	47.5	65
0.25	17.86 ± 0.39	23.69 ± 4.40	28.81 ± 5.23	32.10 ± 7.07	39.04 ± 5.61
1.5	16.13 ± 0.93	21.01 ± 3.58	25.51 ± 7.24	26.06 ± 7.24	32.23 ± 5.36
6	19.72 ± 0.60	23.33 ± 1.92	26.57 ± 3.65	28.54 ± 3.21	34.06 ± 5.67
12	12.63 ± 0.39	14.03 ± 1.17	16.75 ± 4.48	15.04 ± 1.79	16.63 ± 2.38
18	19.32 ± 1.43	23.06 ± 2.89	22.42 ± 4.65	27.03 ± 5.47	30.22 ± 3.93

TABLE 37. Percentage moisture content.

Treatment	Moisture held at 0.05 bars %	Moisture held at 15 bars %	A.W.C.
Soil Control	25.1	10.45	14.65
20 tonnes/ha.	24.3	10.8	13.5
32.5 tonnes/ha.	24.6	11.2	13.4
47.5 tonnes/ha.	25.9	11.75	14.15
65 tonnes/ha.	22.6	11.25	11.35
p	n.s.	0.05	n.s.
l.s.d.	-	0.45	-

A.W.C. = available water capacity.

l.s.d. = least significant difference.

n.s. = not significant.

TABLE 38. Moisture holding capacity of treated soil after eighteen months.

Time (Months)	Dosage (tonnes/hectare)				
	Soil control	20	32.5	47.5	65
0.25	16.0 - 17.5 (16.9)	15.5 - 36.5 (29.1)	29.5 - 62.5 (45.8)	43.5 - 82.5 (56.2)	60.0 - 129.0 (82.8)
1.5	15.8 - 18.5 (16.8)	25.5 - 39.0 (31.6)	27.0 - 220.0 (78.5)	30.5 - 58.0 (42.6)	80.5 - 166.5 (111.8)
12	8.0 - 16.0 (13.4)	22.0 - 40.0 (39.1)	17.0 - 38.0 (28.8)	53.0 - 425.0 (157.0)	87.0 - 200.0 (131.4)

TABLE 39. 0.1M HCl extractable zinc (μg/g) in field dosage plots (ranges plus means of 5 samples in brackets).

Time (Months)	Dosage (tonnes/hectare)				
	Soil control	20	32.5	47.5	65
3	33.75 - 47.5 (41.3)	36.25 - 45 (40.41)	36.25 - 68.75 (50.8)	61.25 - 77.5 (69.2)	65 - 83.15 (72.5)
6	32.5 - 37.5 (35.4)	50 - 62.5 (55.8)	55 - 70 (62.1)	62.5 - 75 (69.17)	75 - 87.5 (80)
15	30 - 40 (35.4)	25 - 61.3 (41.3)	37.5 - 41.3 (39.6)	40 - 130 (90)	47.5 - 56.3 (51.3)

TABLE 40. Total zinc content of ryegrass shoots (μg/g)

(ranges plus means of 3 samples in brackets) it could be that 0.1 M HCl "extractability" overestimated plant 'availability' (humification was causing increased binding of the element) or that the extractable zinc was being leached from the soil by rainfall.

Analyses of extractable copper in the soil and total copper in plant tissue are shown in Tables 41 and 42.

Levels of extractable copper in the soil were increased initially by the refuse additions in amounts corresponding to increasing dosage. After 12 months only 3 samples gave readings above the control indicating small pockets of contamination in the plots which, in general, were no different by this time from the control.

Plant tissue analysis again showed an accumulation of copper by the plants during the first growing season, and then a fall back to much lower levels by the next year. Copper, therefore, unlike zinc, appears to become rapidly bound in the soil and unavailable for plant uptake as biodegradation of the refuse continues.

Results of extractable lead in soil samples, and lead in plant tissue are given in Tables 43 and 44.

The results again indicated significant increases in extractable lead levels in the soil with increasing application rate of the refuse. This effect appeared to be relatively long-lasting and there was little change over the first twelve months, indicating the relative stability of lead in the soil. Plant uptake did not seem to be related, either to dosage rate, or extractable levels of lead in the soil although the fact that the levels in plant tissue were higher on the control and low dosages suggest that there was some plant uptake and accumulation. It is likely that the extractant used was unsuitable for this notoriously insoluble metal and gave very low values for soil extractable lead. Total lead levels would probably have given a clearer picture. Nevertheless, it is reasonable to suggest, since the augmented soil levels at increased dosage rates are not reflected in plant uptake, that lead is relatively strongly bound in the soil and has a low availability to plants. In addition,

Time (Months)	Dosage (tonnes/hectare)				
	Soil control	20	32.5	47.5	65
0.25	1.8 - 3.5 (2.6)	0 - 5.0 (3.5)	2.3 - 6.0 (4.8)	4.1 - 5.5 (4.9)	5.0 - 8.0 (6.8)
1.5	1.8 - 4.0 (3.1)	1.8 - 4.0 (3.3)	3.0 - 4.7 (4.1)	4.0 - 6.0 (4.9)	3.5 - 14.8 (6.8)
12	0 - 3.5 (2.3)	2.0 - 3.5 (2.6)	0 - 4.0 (1.6)	0 - 3.5 (2.2)	2.5 - 8.0 (4.0)

TABLE 41. 0.1M HCl extractable copper (μg/g) in field dosage plots

(ranges plus means of 5 samples given in brackets).

Time (Months)	Dosage (tonnes/hectare)				
	Soil control	20	32.5	47.5	65
3	3.5 - 5.5 (4.5)	3.0 - 6.3 (4.8)	3.0 - 5.0 (4.5)	5.0 - 5.5 (5.3)	3.0 - 7.0 (4.1)
6	2.5 - 4.5 (3.5)	5.5 - 9.3 (7.0)	7.0 - 10.0 (8.3)	6.3 - 12.5 (8.6)	9.8 - 12.5 (10.9)
15	2.5 - 4.5 (3.5)	2.5 - 6.5 (4.2)	4.3 - 5.5 (4.9)	5.0 - 6.5 (5.5)	4.3 - 10.5 (8.0)

TABLE 42. Total copper content of ryegrass shoots (μg/g)

(ranges plus means of 3 samples in brackets).

Time (Months)	Dosage (tonnes/hectare)				
	Soil control	20	32.5	47.5	65
0.25	3.0 - 5.0 (4.2)	2.0 - 12.0 (7.6)	7.0 - 60.0 (26.4)	9.0 - 15.0 (12.0)	11.0 - 24.0 (18.2)
1.5	4.0 - 5.0 (4.6)	4.0 - 10.0 (7.6)	5.0 - 14.0 (7.8)	6.0 - 22.0 (12.4)	15.0 - 26.0 (18.4)
12	5.0 - 6.0 (5.2)	6.0 - 9.0 (7.6)	7.0 - 9.0 (7.8)	10.0 - 18.0 (11.8)	8.0 - 28.0 (18.4)

TABLE 43. 0.1M HCl extractable lead (μg/g) in field dosage plots

(ranges plus means of 5 samples in brackets).

Time (Months)	Dosage (tonnes/hectare)				
	Soil control	20	32.5	47.5	65
3	6.8 - 10.5 (8.3)	6.8 - 7.5 (7.0)	5.8 - 7.5 (6.4)	6.8 - 13.8 (10.3)	5.8 - 6.8 (6.4)
6	18 - 20 (19)	20 - 36.8 (26.2)	19 - 31 (25.8)	20 - 21.8 (20.6)	18 - 29 (23.5)
15	6.8 - 7.5 (7.0)	5.8 - 9.5 (7.3)	6.8 - 10.5 (8.0)	6.8 - 10.5 (8.3)	6.8 - 17 (13.6)

TABLE 44. Total lead content of ryegrass shoots (μg/g)

(ranges plus means of 3 samples in brackets).

Tables 43 and 44 both indicate that lead remains in the soil over the first year and that there is no evidence of losses.

Results of cadmium analyses of soil from field dosage plots and plant tissue are shown in Tables 45 and 46.

As already noted in Section 3.1 cadmium levels in the soil on site were at the upper end of the range normally found in soils, but the addition of the Doncaster refuse, itself relatively low in cadmium, had little effect in increasing these levels, regardless of dosage. However, it should be stressed that the distribution was, as expected, very patchy and odd samples revealed high concentrations. This patchiness was also evident in the plant tissue, although in general there seems to be no difference between plant uptake on the controls and on the dosage plots.

Levels of water-extractable boron in the soil sampled from the field dosage plots are shown in Table 47. Shortage of time allowed only two sets of samples to be analysed for boron.

Increases in water-extractable boron, of up to five times the control level, resulted from the addition of the refuse separates to the plots. Again, there was evidence here of a good deal of patchiness within the treatments, with some samples showing very high concentrations. Higher dosages of the refuse resulted in higher levels of boron in the soil but the amounts were reduced by up to 30% after 12 months, mainly through leaching of this very soluble element.

Relatively few plant tissue analyses for boron concentration were carried out due to time limitations. Two replicates of ryegrass (*Lolium perenne*) taken in August 1980 showed that uptake of boron into the plant was closely related to soil concentration and thus to dosage rate although the concentration at 47.5 tonnes/hectare seemed anomalously low (Table 48). All the recorded concentrations, however, fell within the range normally found for plant materials.

4.1.2 Discussion of results

The aim of most empirical studies investigating land disposal

Time (Months)	Dosage (tonnes/hectare)				
	Soil control	20	32.5	47.5	65
0.25	1.7 - 2.0 (1.9)	1.7 - 2.3 (1.8)	1.7 - 3.3 (2.3)	1.7 - 2.3 (1.9)	2.0 - 3.7 (2.5)
1.5	1.5 - 2.3 (1.9)	1.7 - 2.0 (1.8)	1.7 - 2.3 (2.1)	1.7 - 2.3 (1.9)	2.0 - 3.3 (2.4)
12	1.6 - 1.8 (1.7)	1.6 - 2.0 (1.8)	1.6 - 2.0 (1.8)	1.9 - 2.4 (2.0)	2.0 - 2.2 (2.0)

TABLE 45. 0.1M HCl extractable cadmium (μg/g) in field dosage plots

(ranges plus means of 5 samples in brackets).

Time (Months)	Dosage (tonnes/hectare)				
	Soil control	20	32.5	47.5	65
3	0.23 - 0.26	0.41 - 0.68	0.16 - 0.30	0.20 - 0.24	0.20 - 0.25
6	0.36 - 0.38	0.34 - 0.45	0.37 - 0.53	0.34 - 0.55	0.34 - 0.49
15	0.46 - 0.51	0.42 - 0.49	0.53 - 0.80	0.36 - 0.79	0.24 - 0.76

TABLE 46. Total cadmium content of ryegrass shoots (μg/g) (2 samples analysed).

Time (Months)	Dosage (tonnes/hectare)				
	Soil control	20	32.5	47.5	65
0.25	0.6 - 2.8 (1.5)	1.5 - 3.4 (2.3)	3.5 - 6.0 (4.7)	3.5 - 8.8 (5.7)	6.0 - 9.1 (7.5)
12	0.8 - 2.4 (1.6)	0.8 - 3.2 (2.0)	1.0 - 3.9 (3.0)	3.2 - 4.6 (3.9)	1.1 - 8.4 (5.2)

TABLE 47. Water-extractable boron (μg/g) in field dosage plots (ranges plus means of 5 samples in brackets).

Time (Months)	Dosage (tonnes/hectare)				
	Soil Control	20	32.5	47.5	65
12	12.5	18.9	30.2	23.0	58.0

TABLE 48. Boron concentration of ryegrass shoots (μg/g)

of organic waste materials has been to find the 'optimum' application rate. Although overload in terms of oxygen demand of the organics has never been established (Loehr, 1979), problems at high dosages are usually associated with high levels of nitrate, metal toxicities or impracticable 'bulk'. Nevertheless, it is likely that the rate of dosage of organic wastes added to soils will have a critical effect on the speed at which biodegradation occurs and this will relate to the carbon:nitrogen ratio of both the soil and the waste. However, Stewart and Webber (1976) quoted research showing that application rate had no effect on the decomposition rate of either cattle manure or digested sludge in soil. The results reported here, on the other hand (Table 30 and 32), suggest that the response is a quadratic one, with decomposition rate decreasing at high dosage. This agrees with the findings of Haghiri _et_. _al_. (1978) who are quoted by Khaleel _et_._al_. (1981) in their review of investigations into the effect on soil organic carbon content of organic waste applications. They found that at a low loading rate (15.5 tonnes of carbon/hectare) a greater proportion of all decomposition took place during the first year, whereas at higher loading rates (49.9 and 99.9 tonnes of carbon/ hectare) the proportion of total decomposition which had occurred during the same period was not as high. This implies that stabilisation of the wastes takes a longer time at higher rates of application. Taking the dosage rate and carbon content of the Doncaster refuse, the loading rates of organic carbon in this investigation were 0, 6.6, 10.7, 15.6 and 21.3 tonnes of carbon per hectare. It is probable, therefore, that the refuse separates in the plots treated with 20 tonnes/hectare are virtually stable at the end of the experiment, the net increase over the control by this time being only 0.5% compared with 3.05% at the start. At the highest dosage, however, the carbon content was still 3 times that of the control at the end of the investigation and carbon decomposition was far from complete. The longer-term additive experiments showed very little change in carbon content relative to the control, in any of the plots after the first twelve months.

As noted earlier, nitrogen is normally considered to be the most important limiting factor with respect to the rate of biodegradation of highly carbonaceous wastes (Stewart and Webber, 1976). Table 33 showed the steady decrease in organic nitrogen over time for all treatments following its utilisation by microorganisms. Whether or not the slower rate of decomposition at the highest dosage can be explained in terms of the immobilisation of nitrogen by microorganisms was investigated in laboratory experiments (Sections 4.2 and 4.3).

Table 49 suggests that the carbon:nitrogen ratios on the dosage plots are below the critical level for nitrogen mineralisation of 25 : 1 as proposed by Lanning and Williams (1979) although the initial ratio at the top dosage seems to be rather low and this figure should be considered with caution.

Lanning and Williams (1979), investigating the decomposition of buried litter in china clay sand waste, found that decomposition rates were generally inversely related to C:N ratio. The results of decomposition rates of Doncaster refuse separates find this not to be the case and suggest the rate to be positively related to C:N ratio. This is probably due to the fact that the C:N ratios recorded in the trial plots here were never high enough to cause significant immobilisation of nitrogen which might have seriously limited the rate of decomposition.

Humification

There appears to be little published work on the effect of increasing the application rate of organic materials to soils on humic-acid production and build-up. It would seem likely that the more material applied, the greater will be the production of stable organic matter in the humus fraction. This is indicated by the data in Table 35. However, it is known that application of fresh organic wastes stimulates humus decomposition in soils (Terry, Nelson and Sommers, 1979; Hartenstein, 1981) which would suggest that higher applications may increase the rate of decomposition, which could counteract the previously mentioned build-up. Referring to Table 34 this would appear to be the case with the greatest percentage reduction in humic acids over the 18 month period occurring at the highest dosage. Cultivation of land can also increase humus decomposition, and this probably accounts for the reduction in % humic acids which occurred on the control plots.

Time (Months)	Dosage (tonnes/hectare)				
	Soil control	20	32.5	47.5	65
0.25	13.89	18.35	21.72	24.81	19.91
1.5	12.95	14.82	13.70	16.16	14.08
6	15.76	12.80	12.18	12.88	12.08
12	14.52	12.38	12.20	10.16	11.04
18	17.69	16.65	17.0	13.75	15.89

TABLE 49. Carbon/Nitrogen ratios (Dosage plots).

Soil pH and moisture effects

Application of increasing dosages of Doncaster putrescibles had little effect on soil pH. It may have been expected to have had a liming effect, particularly at the highest dosage, as has been found by several investigators who added refuse compost to soil. Mays *et*.*al*. (1973) found that soil pH had increased from pH 5.4 to pH 6.2 two years after an application of 82 tonnes/hectare of compost from municipal refuse and sewage sludge. A similar increase was reported by Duggan and Wiles (1976). It is suggested that whilst composts may increase the pH of acid soils, the effect is minimal on neutral soils. They may, however, serve to maintain soil pH of neutral soils in comparison with other nitrogenous fertilisers which could depress pH as nitrification proceeds. This would have obvious benefits where waste materials high in toxic metals are applied to land, since availability of metals is normally increased at greater acidity.

The results tabulated in Table 38 showed that higher application rates of the Doncaster refuse increased the amount of water retained at high moisture tensions (15 bars), but the overall 'available' water capacity was reduced. This agrees with work by Webber (1978) who found that shredded solid waste increased the amount of moisture retained by the surface soil in field plots at permanent wilting point (15 bars) but the moisture holding characteristics (measured in terms of available water capacity) of the undisturbed samples from field plots three years after application were not significantly influenced by additions of waste.

Volk and Ullery (1973) found that shredded refuse increased moisture retained at field capacity (0.05 bars), but again that available water was not increased. Investigations into land application of municipal composts have also shown that the moisture holding capability at field capacity is increased by higher application rates (Mays *et*. *al*., 1973) although no mention is made here on the effect of waste additions upon 'available' water capacity.

Clearly, the effect of waste materials on soil physical properties, and water retention, require further explanation. According to Khaleel *et*. *al*. (1981), the water holding capacity of soils is controlled primarily by (i) the number of pores and pore-size distribution of soils; and (ii) the specific surface area of soils. As a result of decreased bulk density (following waste application), the pore-size distribution is altered and the relative number of small pores increases, especially for coarse textured soils (Volk and Ullery, 1973). Since the tension which causes a particular pore to drain depends upon the diameter of the pore, greater tension is required to drain small pores than large pores. Thus the increased water holding capacity at lower tensions (e.g. field capacity) is primarily the result of an increase in the number of small pores.

At higher tensions close to permanent wilting point, nearly all pores are filled with air and the moisture content is determined largely by the specific surface area and the thickness of water films on these surfaces. Clayey soils have a larger surface area than sandy soils and thus retain more water at higher tensions. The addition of organic matter increases specific surface area further, resulting in increased water holding capacity at higher tensions. This effect was seen in this investigation. Soil texture, therefore, may go some way to explaining why some researchers have reported increased water retention at high tensions and some at low tensions following waste additions.

If, however, increases in organic content cause an increase in moisture content at both field capacity and wilting point the net result is that the amount of available water may not be greatly affected since available water capacity is defined as the difference between moisture contents at field capacity and wilting point. Furthermore, an increase in soil organic matter results in a decrease in soil bulk density. The decreased bulk density of the waste-incorporated soil tends to counterbalance any increased available water capacity on a weight basis, resulting in only small increases on a volume basis.

<u>Trace elements</u>

Little reference has yet been made to the detrimental effects which may result from the land application of the Donacaster refuse putrescibles. Accumulation of toxic metals could become a long term soil management problem at high rates of application. However,

Table 50 indicates that although augmented levels of zinc, copper, lead, and cadmium were found in the refuse treated soil, no sample from the site was found to contain concentrations which exceeded the D.O.E. guidelines (1980) for 'acceptable' levels in areas to be used as public open spaces, or amenity grasslands.

Only one sample was found to contain zinc concentration near the 'acceptable' limit (at 425 μg/g) but, apart from this, the highest recorded concentration was less than 200 μg/g. This emphasises not only the variable nature of the refuse separates themselves and the resulting patchiness of contamination, but also the need for careful monitoring of land to which the refuse will be, or has been applied. The copper, lead and cadmium levels are well within the guidelines, but the main area of potential contamination is from boron although this ion is highly soluble in soil and it is rapidly leached out. Any evidence of phytotoxicity on refuse treated areas is likely to be the result of high concentrations of boron.

The behaviour of these five elements during the biodegradation of the refuse (Tables 39 - 48) is similar to that reported earlier (Section 3.1.2). A detailed discussion of all the elements studied, except boron, appears in that section and will not be repeated here.

Boron was found to pose the greatest threat to potential soil contamination of all the elements studied, in both the investigations involving the co-disposal of refuse separates, and digested sewage sluge, and those involving increased application rates of refuse. Normal phytotoxic symptoms of boron may include small, wrinkled or cup-shaped leaves with curled edges and chlorotic spots (Cottrell, 1975). Although none of these symptoms was specifically noted in the vegetation on the field plots, high levels of boron were almost certainly responsible for the poor establishment of barley (*Hordeum vulgare*), a notably intolerant plant, in the 80 and 100% refuse treatments in the growth room experiment (Section 3.3). Ryegrass (*Lolium perenne*) is known to exhibit a relatively good degree of tolerance to high boron concentrations in the soil (Bradshaw and Chadwick, 1980).

		Zinc	Copper	Lead	Cadmium	Boron
Normal range (1)	Total	10 - 300	2 - 100	2 - 200	0.1 - 2	0.2 - 5
	extractable	- - 40	0.1 - 30	1 - 10	0.1 - 2	0.2 - 5
Maximum 'acceptable' levels (2)	Total	n.r.	n.r.	1500 - 2000	12 - 15	n.r.
	extractable	280 - 560	140 - 280	250	n.r.	6
Highest recorded levels in this investigation.	Total	n.r.	n.r.	n.r.	n.r.	n.r.
	extractable	425	14.8	28	3.7	9.1

n.r. = not recorded.

(1) = Allen, 1974.

(2) = D.O.E., 1980.

TABLE 50. Potential toxicity of certain elements present in the Doncaster refuse separates (all figures as µg/g dry soil).

Boron toxicity was found to be a serious problem by A.D.A.S. workers during their experiments at High Wycombe investigating the growth potential of pulverised domestic refuse with a boron content of 9 - 10 μg/g. They found that by mixing the refuse with soil, boron levels were brought down to about 4 μg/g and this provided a more acceptable growth medium. (personal communication)

Cottrell (1975) also found that boron posed the greatest threat of toxicity following the application of large quantities (up to 1000 tonnes/hectare) of solid municipal waste to a sandy soil. She found that the soil boron content increased up to 60-fold initially and although it decreased to 0.8 μg/g or less within the first season in the soil with the lower waste treatments (250 tonnes/hectare), it remained greater than 2 μg/g after two years in the soil which received the highest applications. Uptake of boron by wheat and fescue reached toxic levels, but uptake by alfalfa increased only slightly.

Soil contamination and phytotoxicity by boron has also been found associated with refuse compost applied to land. (Purves and Mackenzie, 1973). They noted augmented levels of water-extractable boron in soils to which 50 and 100 tonnes/hectare of municipal compost had been added, and that a relatively small increase in soil content led to a substantial increase in boron content of lettuce, dwarf beans, potatoes and peas. They suggest that most municipal composts which have phytotoxic properties owe these to a high content of water-soluble boron, which plants have no means of excluding.

4.1.3 Conclusion

Rate of decomposition of the refuse putrescibles exhibited a quadratic response with increasing dosage; the highest rate corresponding to a dosage of 47.5 tonnes/hectare. Carbon:nitrogen ratios at all dosages and all sample times were below the critical levels for mineralisation and nitrogen limitation of biodegradation was not in evidence. Long-term stabilisation of the wastes was measured by the extent of humification, and showed that although there was a greater build up of humic material at the higher dosages, this was counteracted by high rates of humic acid breakdown at these dosages

Effects of dosage rate on soil acidity was minimal and although the water holding capacity of the soil was increased by greater amounts at higher dosages, at 15 bars suction (permanent wilting point) no change was noted at any dosage in available water capacity. It seems that changes in the water holding characteristics of soils to which waste materials have been added are very much related to inherent soil physical properties such as texture, and also to properties altered by the wastes themselves such as bulk density.

All the toxic metals analysed were within acceptable limits (D.O.E., 1980) but boron concentrations were above recommended levels at all but the lowest dosage immediately after application. Plant uptake of boron was directly related to amounts within the soil, but never reached phytotoxic levels, due presumably to the relatively rapid leaching of this element from the soil during the growing season.

4.2 Soil/refuse mixtures as media for plant growth

4.2.1 Results of field growth analysis

Figures 12 and 13 are graphical representations of the field growth experiment conducted immediately after refuse incorporation in July, 1980, and using perennial ryegrass (*Lolium perenne*) as the trial plant. The data from which these were derived appears in Appendix C. Figure 12 shows that in terms of dry weight increase all treatments showed significantly higher increases than the control after the initial stages of growth. The highest dosage treatment showed the longest lag period in growth before producing an eventual yield well above the control. The middle ranges of dosage rate appeared to give the greatest yields but differences were not significant, as indicated by the plotted 95% confidence limits. Figure 13 indicates that the relative growth rate fell more steeply over time with increasing dosage, i.e. growth was more rapid at higher dosages during the early stages of growth, but in the final stage this dropped to a level below the control.

Plant tissue was analysed for nitrogen at each sample time; the results are presented in Table 51. According to Dancer et. al.(1979), the nitrogen concentration of plant tissues is one of the best indicators of the amount of available nitrogen in the soil at any one time. Tissue nitrogen here appeared to increase with increasing dosage at each sample time up to the highest dosage, at which point it fell slightly. The only exception to this was at the final harvest (December) at which time there was no decrease at 65 tonnes/hectare. This pattern suggests that there was a lack of availability of nitrogen at this high dosage in the first four months, reflecting some inhibition of mineralisation. The data shows that by December, 1980, mineralisation was producing larger amounts of available nitrogen at the highest dosage than at the lower rates of application.

In order to obtain a longer-term assessment of the productivity of the treated plots, ryegrass biomass was estimated after thirteen months using the same dry weight per unit volume method as was used in the field growth trials (Table 52).

All dosage treatments except 20 tonnes/hectare remained signific-

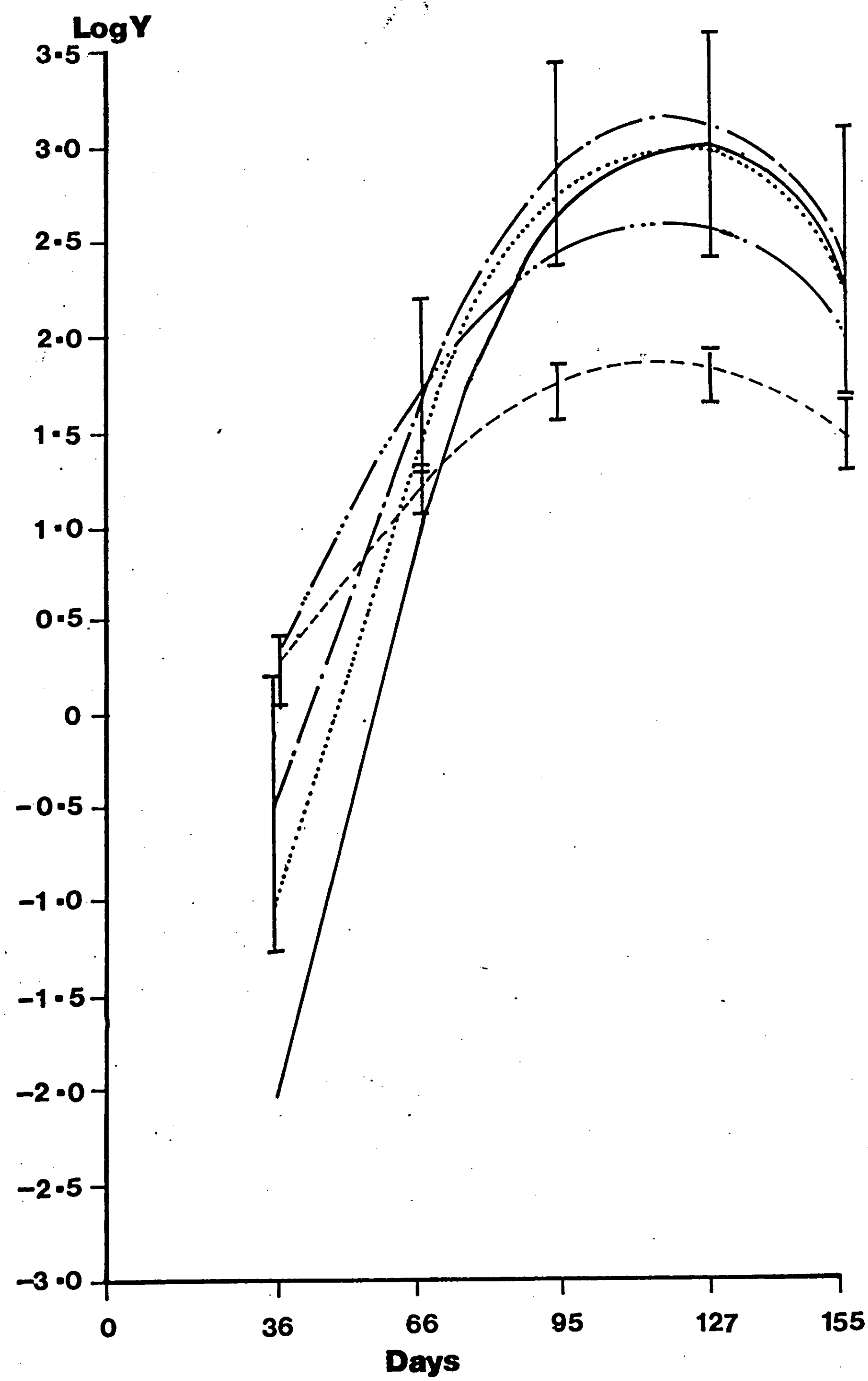


FIG.12
Plant dry weight increase following refuse addition at different dosage rates

KEY

----	Soil Control (+ Confidence Limits)
—··—	20 t Refuse/hectare
—·—	32·5t Refuse/hectare (+ Confidence Limits)
·······	47·5 t Refuse/hectare
——	65 t Refuse/hectare

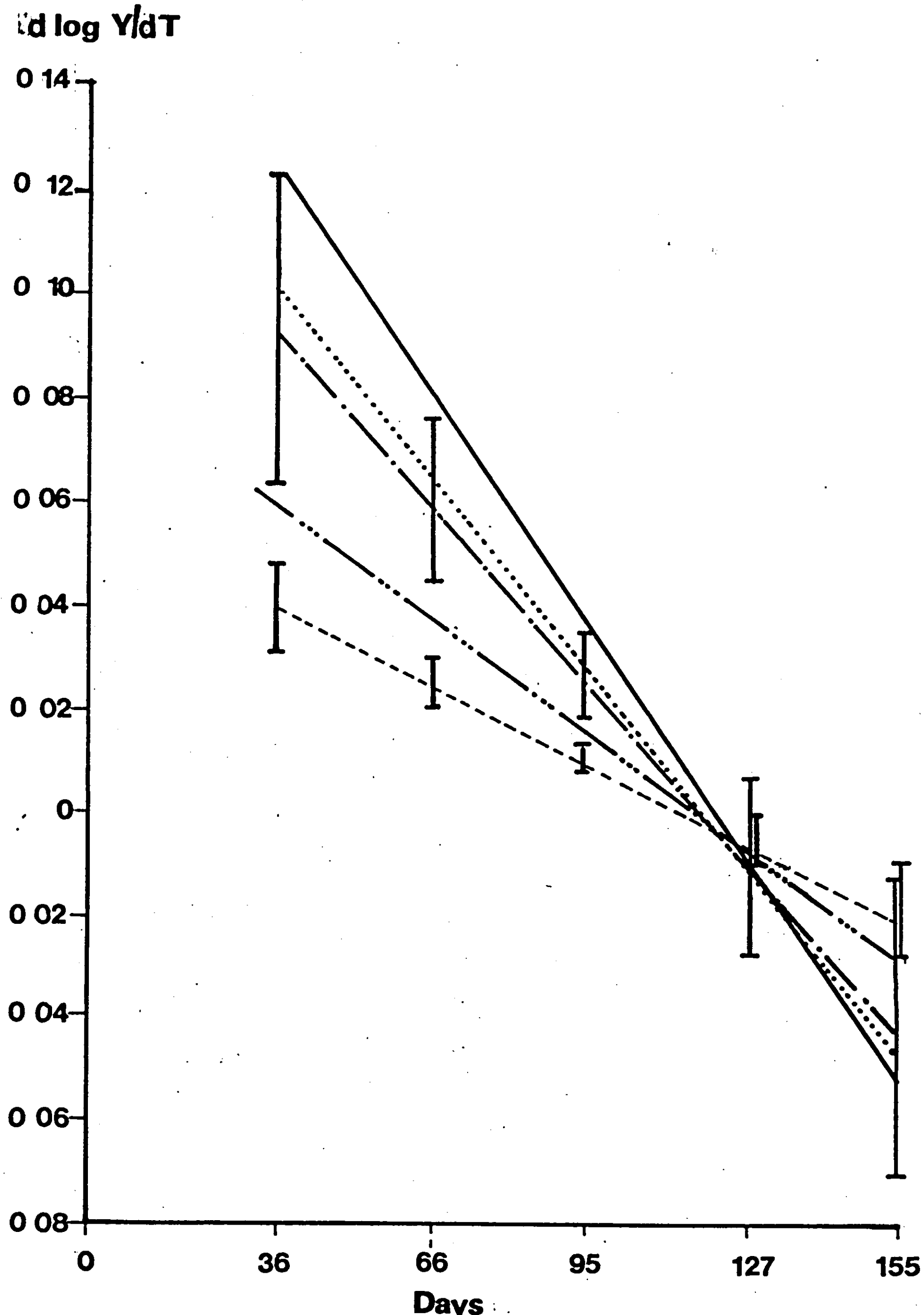


FIG 13
Relative growth rate
KEY

Soil Control (+confidence limits)
20t/hectare
32·5t/hectare (+confidence limits)
47·5t/hectare
65t/hectare

	Dosage (tonnes/hectare)				
	Soil Control	20	32.5	47.5	65
August 1980	3.33 ± 0.15	4.78 ± 0.46	5.72 ± 0.13	5.68 ± 0.23	5.12 ± 0.02
September	2.83 ± 0.21	3.77 ± 0.37	4.88 ± 0.57	5.50 ± 0.28	5.18 ± 0.06
October	3.08 ± 0.27	3.73 ± 0.25	3.38 ± 0.16	4.40 ± 0.50	4.38 ± 0.27
November	2.57 ± 0.21	3.03 ± 0.44	4.53 ± 0.29	4.67 ± 0.62	4.02 ± 0.19
December	2.95 ± 0.35	3.93 ± 0.22	4.18 ± 0.17	4.42 ± 0.13	4.68 ± 0.15

TABLE 51. Percentage nitrogen in *Lolium perenne*.

Soil control	Dosage tonnes/hectare				p	l.s.d.
	20	32.5	47.5	65		
9.88	12.04	16.52	18.34	14.73	0.01	5.1

l.s.d. = least significant difference.

TABLE 52. Dry weight (g.) of *Lolium perenne* per 25 cm^2 quadrat, September, 1981.

antly higher than the control. The pattern of decreasing productivity at the top dosage was still in evidence although the differences between the top three dosages were not significant (l.s.d. at $p \leqslant 0.05$ was 5.1)

4.2.2 Results of growth room analysis

The more closely controlled growth experiment was conducted in a growth cabinet using barley (*Hordeum vulgare*) and the results of dry weight increase and relative growth rate are shown in Figures 14 and 15. These again show that the soil/refuse mixtures produced significantly higher plant yields than the control, and that the highest yield was in the middle range (60% refuse); this time, however, it was significantly higher than with the other dosages. Relative growth rate (Fig.15) was maintained at a higher level than the control in all treatments except the 100% refuse which showed a distinctive pattern of increasing relative growth rate, representing a more prolonged first stage of the typical growth curve.

Nitrogen content of the plant tissues was analysed and recorded at each harvest time (Table 53). There appeared to be a general trend of increasing tissue nitrogen corresponding with increasing dosage. There was no indication here that production of plant available nitrogen (as shown by amounts taken up into the plant tissue) was limited at high dosage, even in the pots containing 100% refuse.

Recorded levels of inorganic nitrogen, as shown by Tables 54 and 55, and Figures 16 and 17, would seem to support the suggestion that, certainly under controlled growth room conditions the production of nitrate nitrogen was only inhibited in the first few days when recorded levels in the pots containing the refuse separates were below control levels.

At the start of the experiment most of the inorganic nitrogen was nitrate-nitrogen in the soil controls and 20% refuse treatments, and ammonium nitrogen in the higher refuse treatments. The data indicates a general trend of mineralisation and ammonification. However, during the first few days ammonium-nitrogen increased in the 60% and 80% refuse treatments and much was subsequently lost by volatilisation

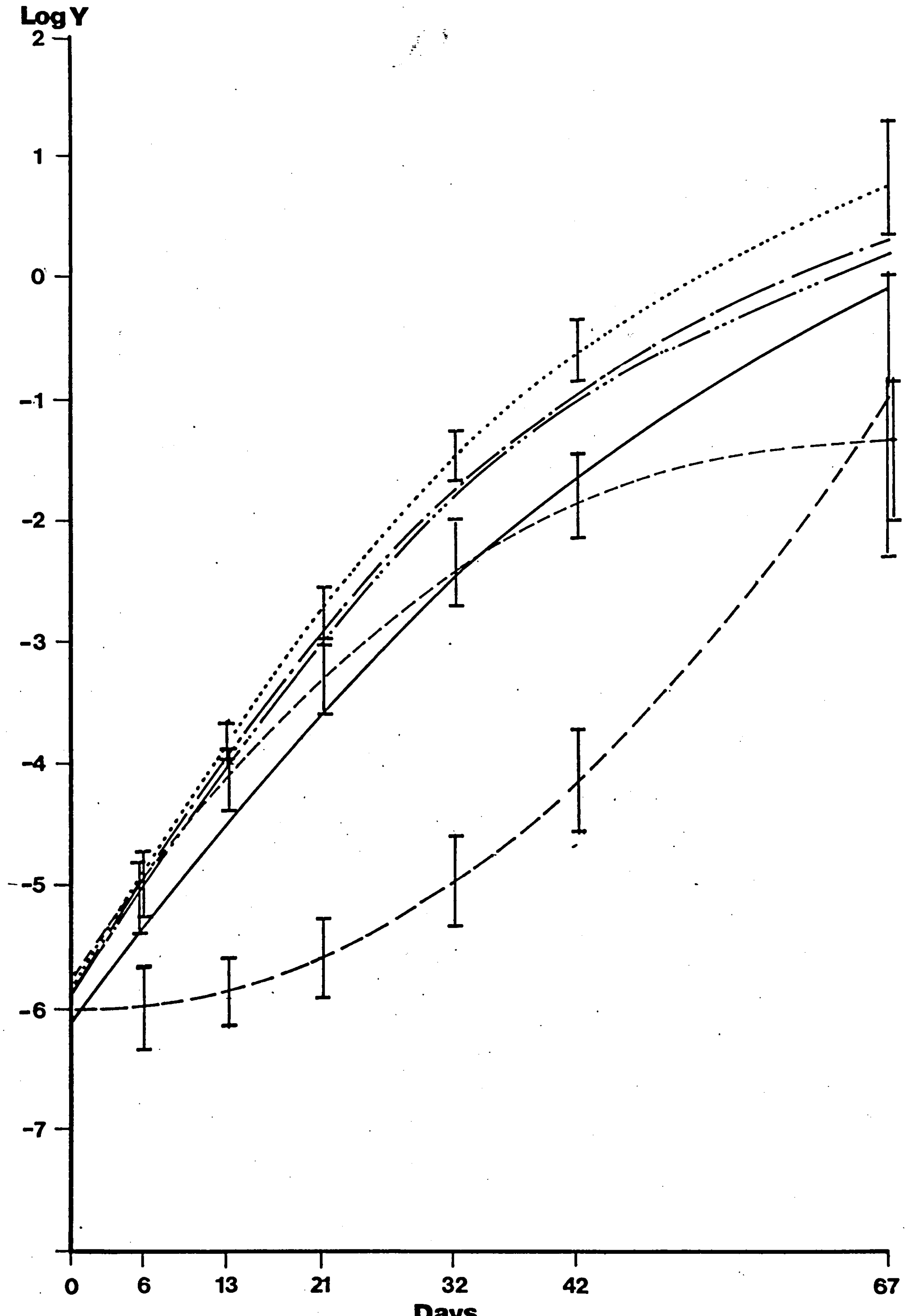


FIG. 14
Plant dry weight increase at different refuse dosages (pot experiment)

KEY

-----	= Soil Control (+ CONFIDENCE LIMITS)	······	= 60% Refuse (+ CONFIDENCE LIMITS)
—··—	= 20% Refuse	——	= 80% Refuse
—·—	= 40% Refuse	— — —	= 100% Refuse (+ CONFIDENCE LIMITS)

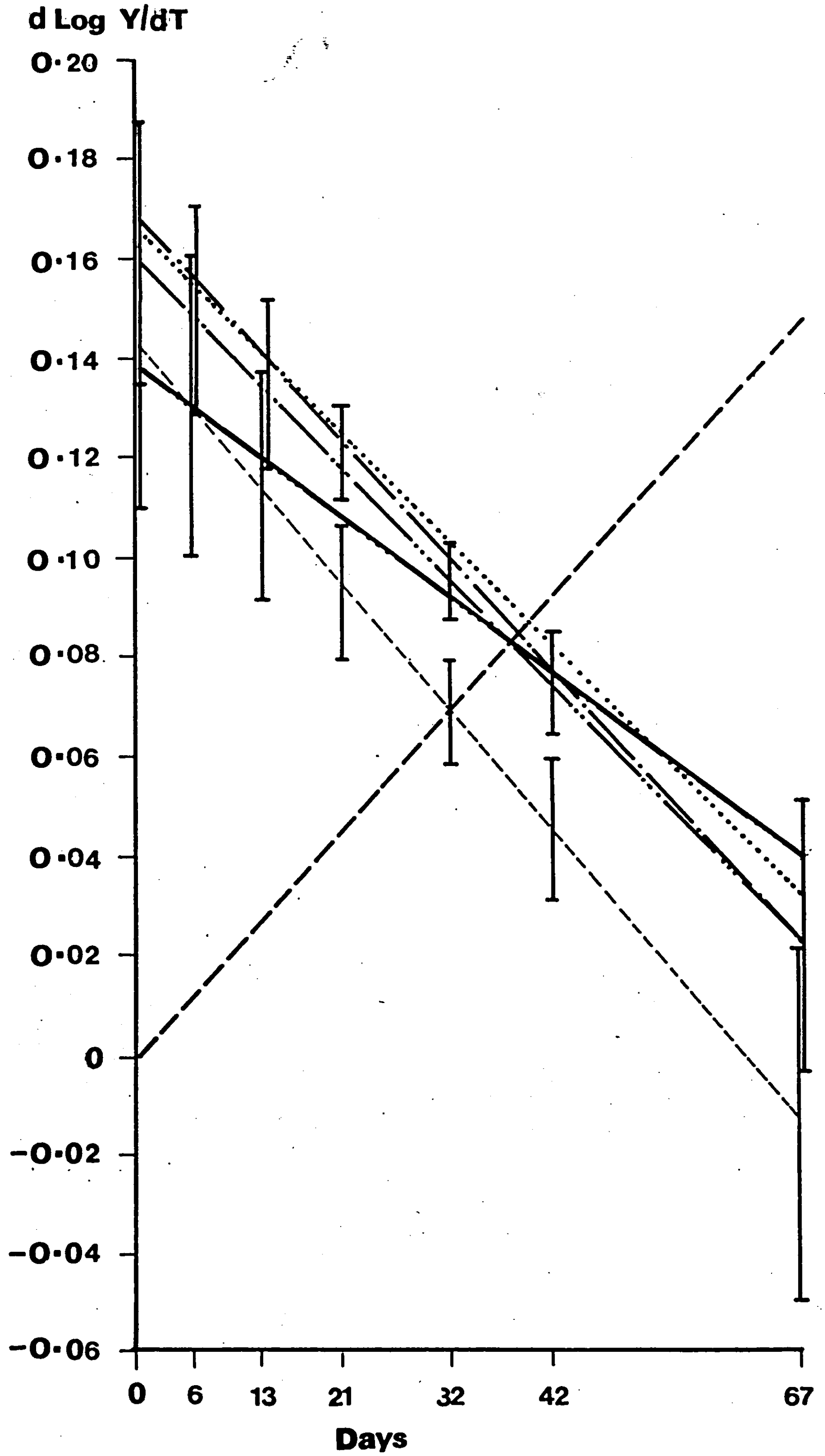


FIG.15
Relative growth rate (pot experiment)

KEY

- - - - -	= Soil Control (+ Confidence Limits)	· · · · · ·	= 60% Refuse
—···—	= 20% Refuse (+ Confidence Limits)	———	= 80% Refuse
—·—	= 40% Refuse	— — —	= 100% Refuse

Time (Days)	Control 100% soil	Treatment				
		20% refuse	40% refuse	60% refuse	80% refuse	100% refuse
6	7.5	6.0	6.5	8.0	8.0	9.0
13	5.0	4.1	4.6	5.4	6.5	6.5
21	4.4	4.6	4.1	4.5	5.4	9.0
32	3.2	4.8	4.7	5.0	5.3	8.0
42	3.0	3.7	3.9	3.8	5.0	8.0
62	3.7	4.0	3.9	4.8	3.9	4.5

TABLE 53. Percentage nitrogen in *Hordeum vulgare*.

Time (Days)	Treatment					
	100% soil	20% refuse	40% refuse	60% refuse	80% refuse	100% refuse
1	28.5 ± 1.2	29.2 ± 7.4	24.5 ± 1.3	23.5 ± 2.7	17.2 ± 1.5	6.1 ± 8.6
6	23.4 ± 3.8	22.7 ± 9.0	16.2 ± 4.4	19.2 ± 4.9	20.9 ± 3.9	18.2 ± 11.0
21	22.8 ± 6.4	25.3 ± 2.1	22.2 ± 0.8	22.4 ± 5.6	38.9 ± 20.3	39.1 ± 7.2
67	32.0 ± 2.0	83.4 ± 14.1	168.2 ± 54.2	248.6 ± 66.0	296.9 ± 62.6	333.5 ± 71.7

TABLE 54. Soil nitrate-nitrogen (μg/g), pot experiment.

Time (Days)	Treatment					
	100% soil	20% refuse	40% refuse	60% refuse	80% refuse	100% refuse
1	14.6 ± 2.0	22.4 ± 4.4	134.4 ± 125.2	52.6 ± 7.4	508.8 ± 296.6	2320.6 ± 429.7
6	14.0 ± 1.9	22.7 ± 3.6	44.4 ± 31.3	99.6 ± 40.8	609.2 ± 348.2	1335.6 ± 487.5
21	15.2 ± 2.3	19.9 ± 0.7	28.0 ± 3.9	35.3 ± 5.4	357.9 ± 187.0	559.0 ± 325.7
67.	18.9 ± 5.1	16.2 ± 1.6	27.7 ± 13.0	33.9 ± 16.2	32.4 ± 9.2	89.5 ± 63.7

TABLE 55. Soil ammonium-nitrogen (μg/g), pot experiment.

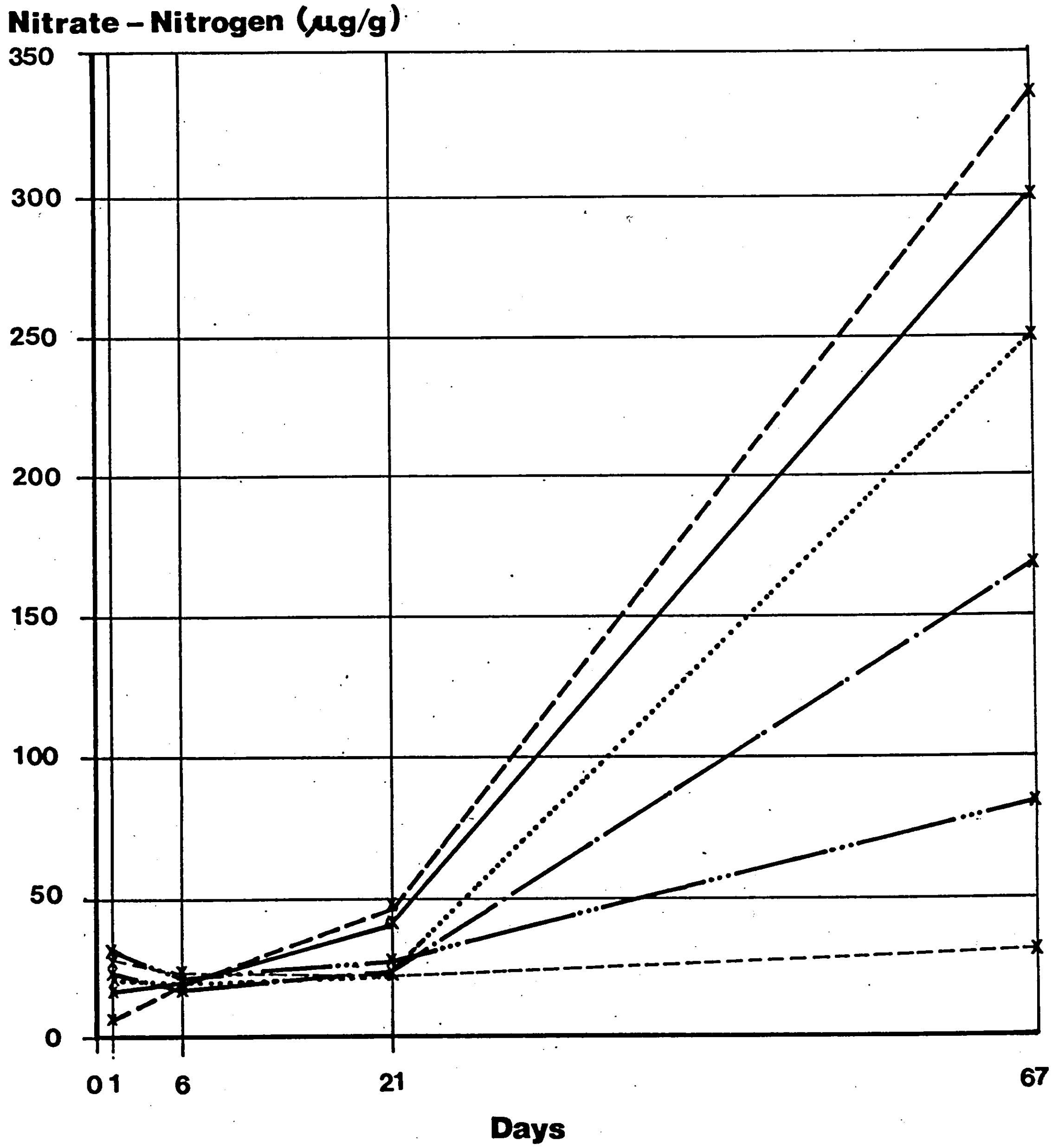


FIG.16

Changes in soil Nitrate-Nitrogen (pot experiment)

KEY

- - - - -	= Soil Control
—···—	= 20 % Refuse
—·—	= 40 % Refuse
······	= 60 % Refuse
——	= 80 % Refuse
— — —	= 100% Refuse

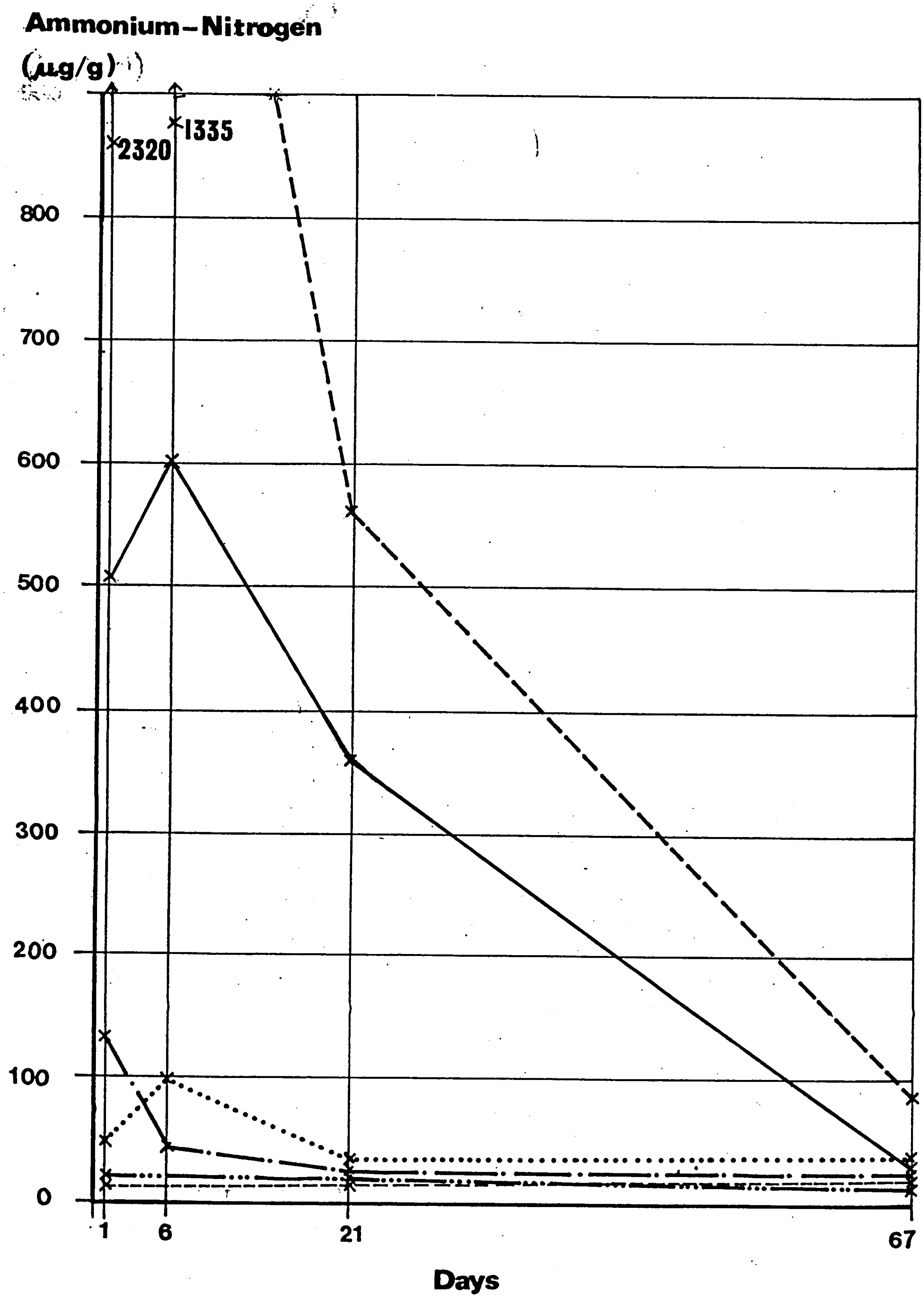


FIG.17
Changes in soil Ammonium-Nitrogen (pot experiment)
KEY

-------	= Soil Control	·······	= 60% Refuse
—···—	= 20% Refuse	———	= 80% Refuse
—·—·—	= 40% Refuse	— — —	= 100% Refuse

(the smell of ammonia was noticeable from the higher dosage pots) as shown in Table 55 and Figure 17. At the same time there appeared to be a lag period before nitrification (conversion of ammonium-to nitrate-nitrogen) became apparent after about 20 days. A similar lag was found in the additives experiment for the soil plus refuse treatment although this was not seen in the well-digested and stable sewage sludge. (Section 3.3)

It should be pointed out here that grinding of material may increase levels of mineralised nitrogen by improving the accessibility of organic matter to microorganisms (Bremner, 1965). The levels of inorganic nitrogen recorded in the growth room experiment, may, therefore, have been artificially high.

4.2.3 Discussion of results

If the Doncaster refuse separates are to be used as a soil amendment in land reclamation schemes, it is important to investigate their manurial value in terms of vegetational yield at different application rates. Previous work has reported adequate crop yields using various kinds of organic wastes as agricultural soil amendments. In general, these findings indicate that the yield response to organic wastes is quadratic, declining at excessively high rates of application (Sims and Boswell, 1980). The results from this study support these findings as indicated by the graphic representation of the response surface analysis (the data is presented in Appendix D). Figures 18 and 19 show the canopy shape of the response yield to time and dosage in the laboratory growth experiment, and to dosage squared in the field experiment where the response to dosage was not significant. This shape suggested a slight decline in yield at the highest dosages, although the effect became smaller over time. The laboratory experiment showed the differences in relative growth rate more clearly than the field experiment. Figure 21 shows that response in terms of growth rate was highest in the middle of the dosage range during the early stages of biodegradation but the decreasing convexity of the curve indicates that the effect became less distinct over time.

Sims and Boswell (1980) suggest factors which may contribute to this decline at high dosages such as inhibition of germination,

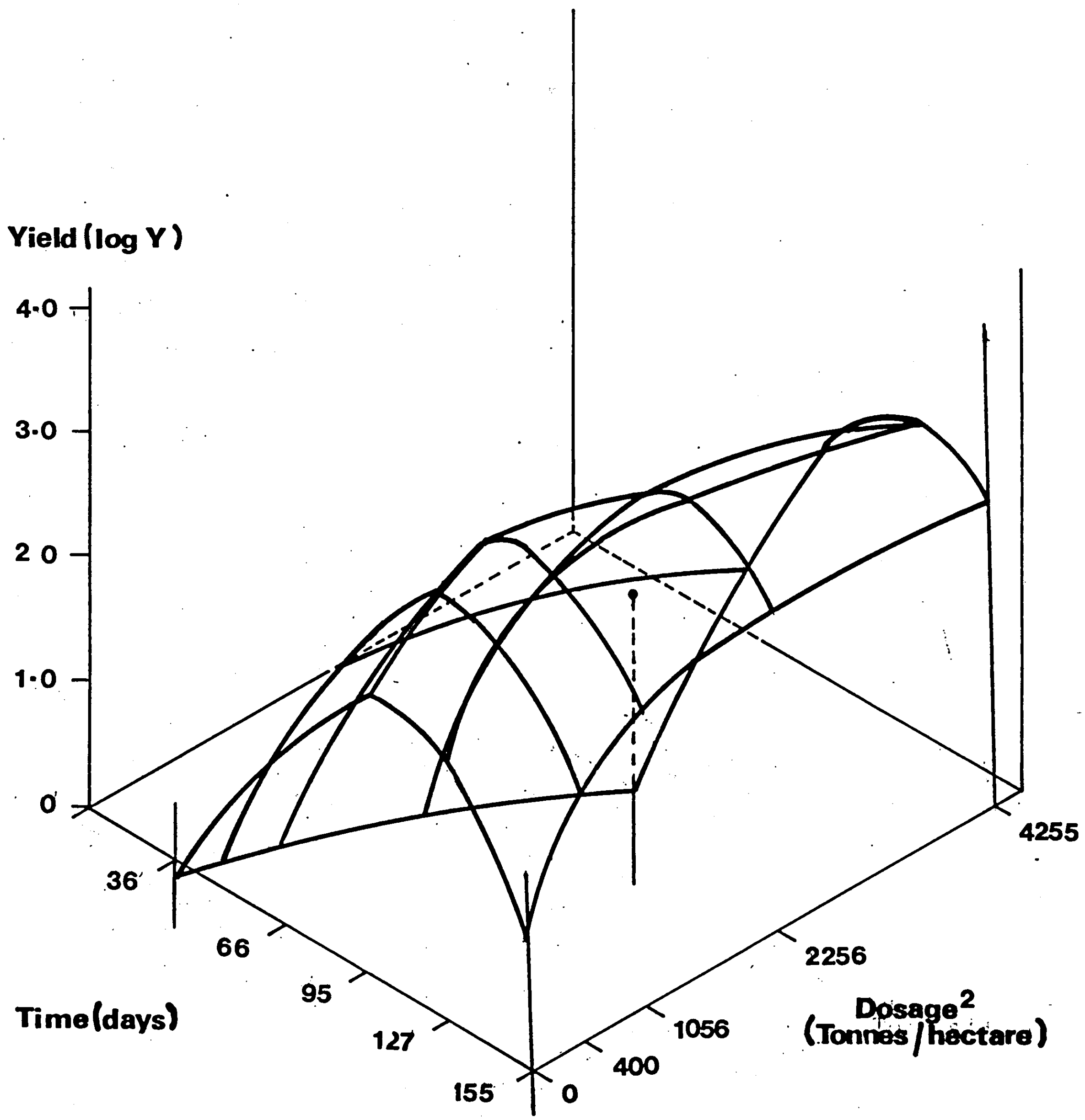


FIG. 18

Response - surface analysis. Yield as a function of time & dosage squared (field experiment)

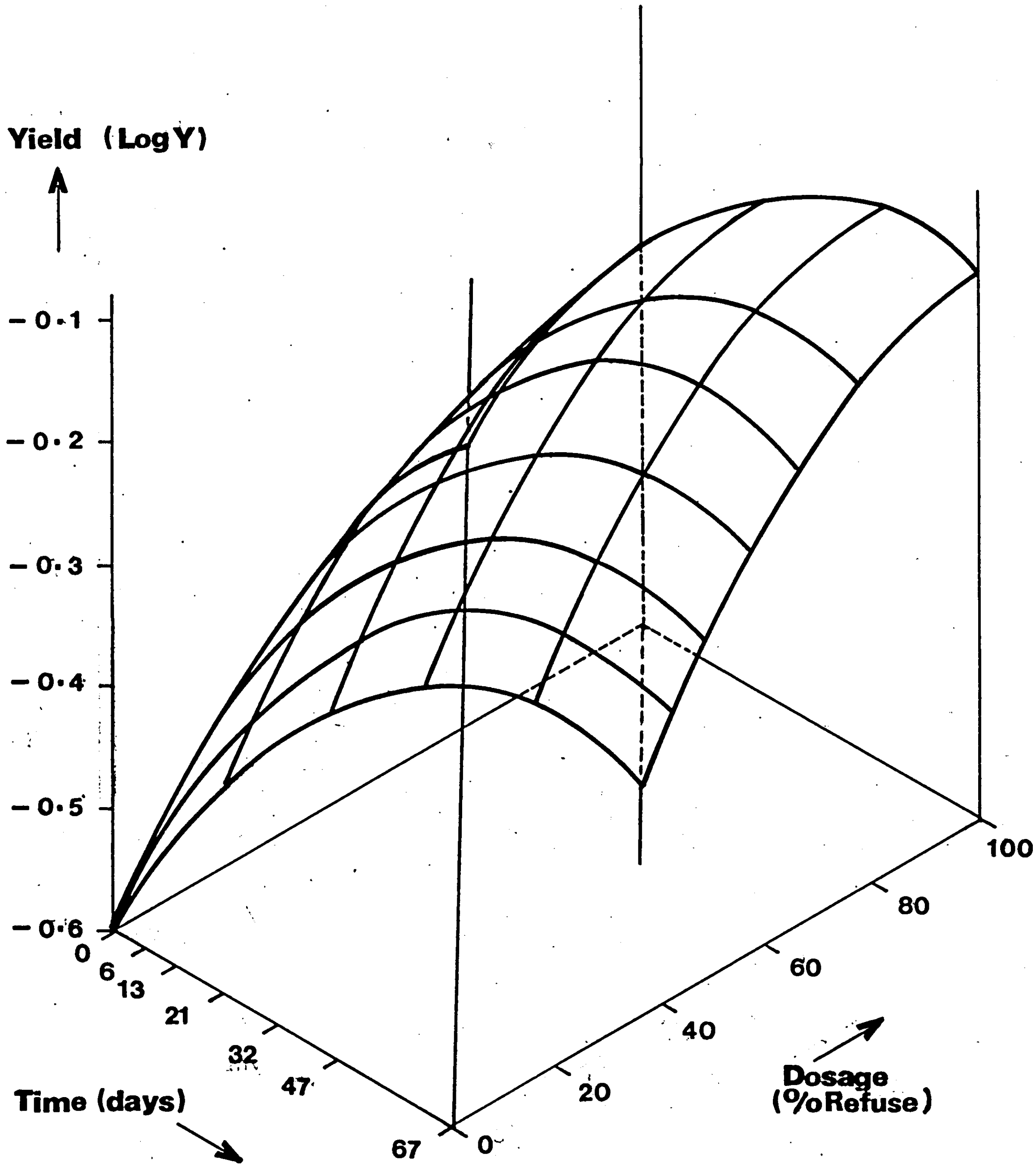


FIG.19

Response – surface analysis. Yield as a function of time & refuse dosage (pot experiment)

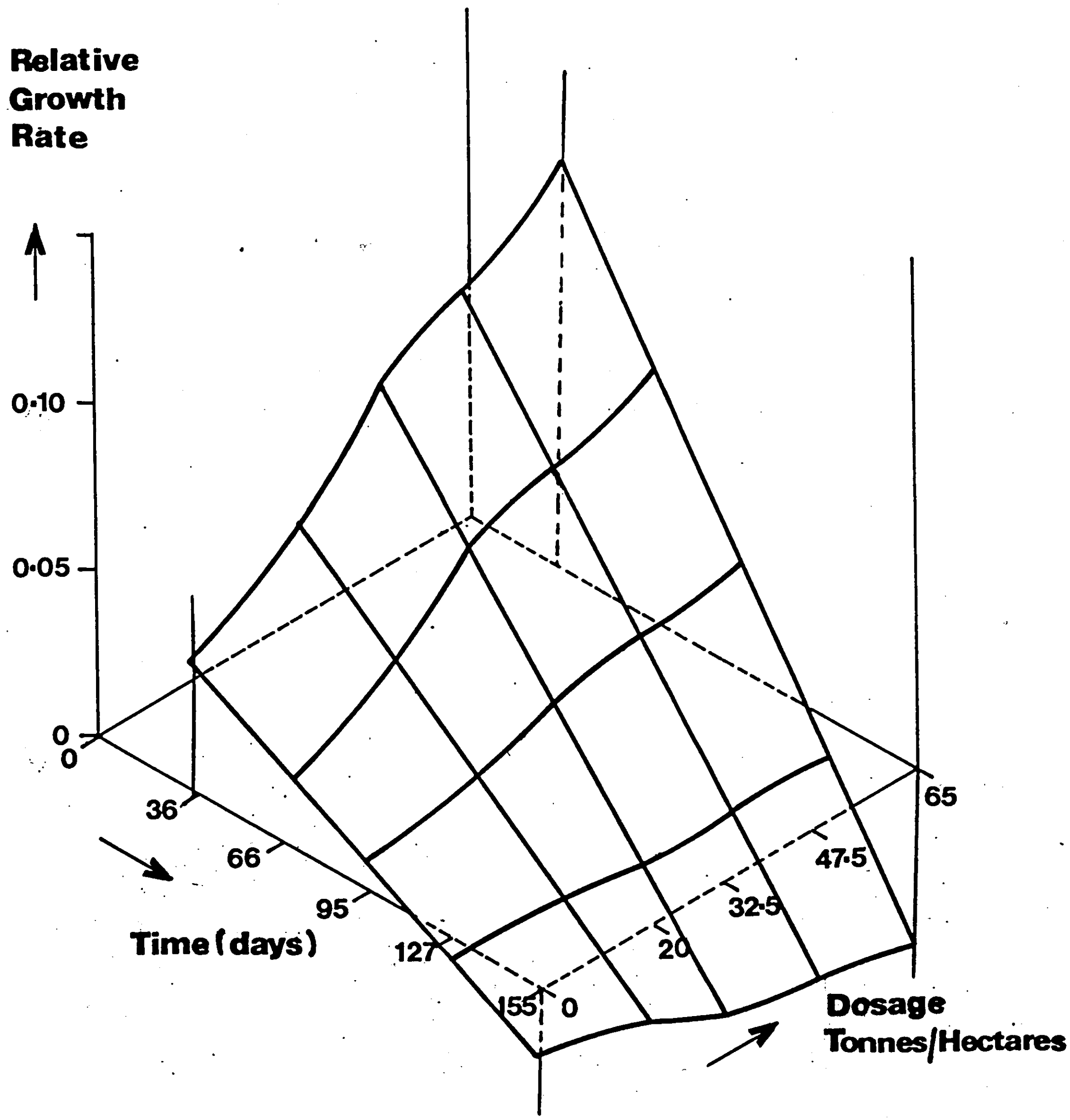


FIG.20

Response surface analysis. Relative growth rate as a function of time & dosage (field experiment)

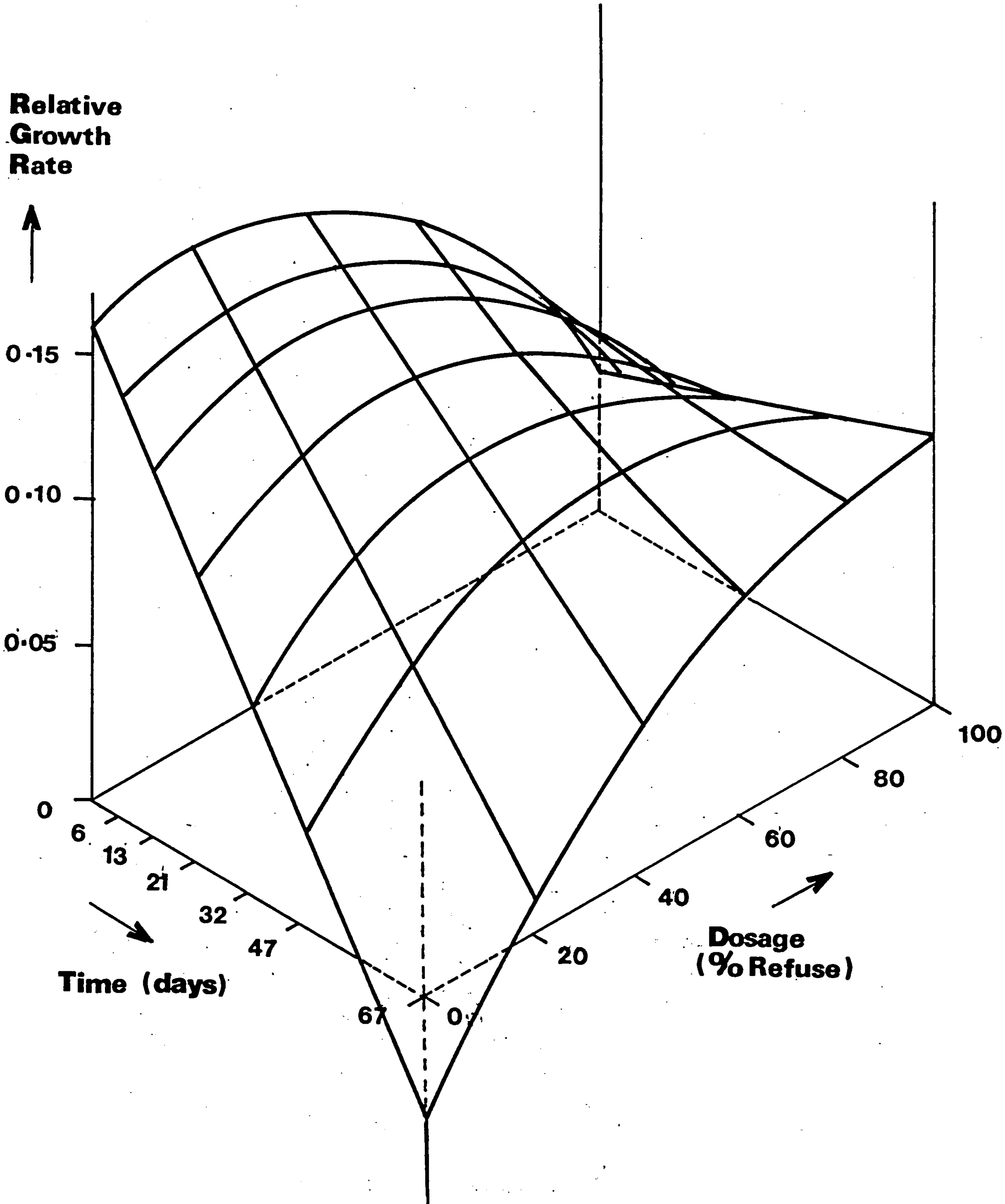


FIG. 21

Response – surface analysis. Relative growth rate as a function of time & dosage (pot experiment).

phytotoxicity of heavy metals and immobilisation of available nitrogen by high C:N wastes. The first of these was likely to be partly responsible for the slightly reduced yield found in this investigation at the highest dosage in the field trials, although once established the seedling growth rapidly equalled, and then exceeded that of the control. Phytotoxicity, due mainly to high levels of boron and other soluble salts was possibly important, particularly in the very harsh 100% refuse environment included as the extreme case in the laboratory growth experiment. It is also suggested that high levels of ammonium-nitrogen (over 2000 μg/g) could also have caused inhibition of seedling growth during the first few days in this particular treatment.

Reduced yields at high application rates of organic waste have often been attributed to lack of available nitrogen. Mays et. al. (1973) reported that the use of inorganic nitrogen plus compost resulted in greater sorghum yields than compost or nitrogen alone, suggesting the need for nitrogen supplementation of refuse compost. This suggestion was supported by Duggan and Wiles (1976) who applied varying rates (0 - 488 tonnes/hectare) of municipal waste compost fortified with sewage sludge to corn for five years. Yields on compost-treated plots were higher with increasing rates of application up to 100 tons/acre (250 tonnes/hectare); above this level yields decreased and the plants showed nitrogen-deficiency. This was attributed to the large C:N ratio of the compost.

The amount and form of potentially available nitrogen in the soil depends upon the extent of mineralisation. This involves the microbial decomposition of the organic materials with release of ammonium-nitrogen (ammonification) which is then converted to nitrite and then to nitrate (nitrification). Nitrification is more sensitive to environmental influence than ammonification and ammonium-nitrogen may accumulate under conditions of low temperature, low pH and high soil moisture content (Haynes and Goh, 1978). Significant amounts of ammonia may be lost from waste materials due to volatilisation if they are merely spread on the soil, but incorporation should reduce this (Stewart and Webber, 1976) and enhance nitrification (Webber and Doyle, 1975). Only the 100% refuse treatment in the laboratory showed any significant losses by volatilisation, indicating that incorporation of the refuse into the soil did indeed reduce these losses. The ammonia set free in the decomposition was probably absorbed by the clay in the soil and then nitrified.

Analysis of percentage nitrogen in the herbage collected from the field trials (Table 51) suggested that decreased availability of nitrogen was only slightly evident during the first four months of biodegradation and then only at the highest dosage rate. Mention should be made here of the possibility that mineralisation of added nitrogen was inhibited by the ryegrass itself. Magdoff and Amadon (1980) found that corn yields responded better to added nitrogen rich sludge (with increased levels of phosphorous, potassium and humic substances) than to fertiliser-nitrogen. Ryegrass on the other hand, responded better to fertiliser-nitrogen and the authors suggested that the ryegrass inhibited the mineralisation of the added sludge. This fits in with the accepted view that mineralisation rates are lower under grass than under arable cropping, due probably to the greater production of high carbon/nitrogen organic substances by roots in the rhizosphere (Harmsen and Kolenbrander, 1965).

The growth room experiment revealed an overall trend of nitrogen mineralisation in the soil. Again, however, there was evidence of denitrification, or at least a lack of nitrification, at the highest dosages during the early stages of biodegradation. Inhibition of nitrification can occur at temperatures above 45^{o}C. (Harmsen and Kolenbrander, 1965) but recorded soil temperatures never exceeded 30^{o}C. so that temperature inhibition was unlikely to have been significant here. Denitrification could be expected to be an important mode of nitrate loss in soil receiving high dosage rates of organic refuse. The refuse would supply the energy source required for denitrification; it would increase respiration and moisture content and thus induce anaerobic conditions at various microsites within the soil.

Nitrate losses from the soil can be attributed to plant uptake, volatilisation, denitrification, leaching and immobilisation. High dosages of organic wastes have been shown to result in a greater water retention (Webber, 1978) which would reduce leaching thereby causing a comparable reduction in nitrate loss by this means. Microbial immobilisation would result in levels of nitrate being reduced to below the control levels, as found by Rothwell and Hartenstein (1969) following the addition of refuse compost to Arrendondo fine sand. Immobilisation could, therefore, partly account for the reduced nitrate levels during the first week of the growth room experiment at dosages of 100% and 80% refuse where C : N ratios were greater than 25 : 1. Denitrification was evident in both field and growth room experiments during the early stages at high dosage; in the growth room, volatilisation of ammonia seemed to be a significant method of loss.

In general, therefore, despite spatial and seasonal variability, it would appear that soil incorporation of the Doncaster refuse separates at all but excessively high dosages did not cause significant immobilisation of nitrogen, although there may have been some inhibition of nitrification at dosages above approximately 50 tonnes/hectare. This was short lived, but as such could partly explain the observed effect on plant growth.

The results of the plant growth experiment at different application rates of Doncaster refuse separates seem, therefore, to agree with the findings of Webber (1978). He found that yields of corn (*Zea mays*) were higher in the first crops following the addition of fresh shredded domestic refuse at 188 tonnes/hectare (oven-dried basis) than in subsequent years. This result was not anticipated in view of the high C : N ratio of 65 : 1 of the shredded solid waste. He also found an apparent lack of net immobilisation of nitrogen, which he attributed to the initial soil nitrogen level of 0.24%.

Soil seems to act as a buffer to ameliorate the extent of nitrogen immobilisation in highly carbonaceous wastes. It also buffers the effect of other factors such as toxicity of soluble salts in domestic refuse which have been shown to have a detrimental effect on plant growth (unpublished work carried out by M.A.F.F. (A.D.A.S.) using pulverised refuse at High Wycombe, Buckinghamshire - personal communication.

In general, therefore, it would seem that the problem of nitrogen immobilisation following the addition of non-composted domestic refuse in soil has perhaps been overstated. Rees (1980) demonstrated that microbial growth in refuse landfills is not limited by nitrogen availability. If this is so, it is unlikely that it will be a limiting factor when refuse is mixed with soil.

Having established that soil availability of nitrogen can only partly explain the observed pattern of plant growth, other suggestions need to be made. Inhibition of germination and high levels of boron, and ammonium-nitrogen have already been mentioned. It is also possible that by mixing refuse with soil at high dosages shortages of certain trace elements may have occurred. It has been shown, for example, that nitrogen fertiliser addition can sometimes cause immobilisation of sulphur in the soil by upsetting the nitrogen:sulphur balance (Kowalenko and Lowe, 1978). Analyses of selected samples from the field trials in this investigation for a wide range of plant nutrients (Appendix B) showed that magnesium may be in short supply on the plots. However, Cottrell (1975) found that with the exception of nitrogen, none of the major plant nutrients (phosphorous, sulphur, calcium, magnesium, potassium) limited fescue or alfalfa growth following the addition of refuse to crop-growing land. The content of each of these elements in the plants grown on waste-treated soil was found to be similar to plants grown on untreated control plots. It is clear that there is room for further investigative work on this subject.

4.2.4 Conclusion

Plant response to increasing the application rate of the refuse separates followed a quadratic curve, with both relative growth rate and yield declining at rates above 50 tonnes/hectare. A similar pattern occurred in the laboratory experiment and may in part be due to inhibition of nitrification at high dosage. These effects disappeared within a relatively short time. Other problems which were associated with the early stages of growth at high dosage were possible boron toxicity and high levels of ammonium-nitrogen.

4.3 Mineralisation of nitrogen in the soil/refuse mixtures

4.3.1 Results of incubation experiment

Initial levels of nitrate-nitrogen in the field samples, and amounts mineralised over the four week test period, are shown in Tables 56 (October 1980) and 57 (October 1981) and Figure 22. During

		Dosage (tonnes/hectare)					
	Soil Control	20	32.5	47.5	65	p	l.s.d.
Initial NO_3 - N	11.0	44.4	64.2	72.2	126.5	0.001	29.5
NO_3 - N mineralised	67.4	116.9	143.6	152.9	126.3	n.s.	-
NH_4 - N mineralised	31.9	15.9	5.4	3.7	0	0.05	16.9
Total mineralised	99.3	132.8	149.0	156.6	126.3	n.s.	-

l.s.d. = least significant difference.

n.s. = not significant.

TABLE 56. Mineralised nitrogen (μg/g) during 28 days incubation, October 1980.

	Soil Control	Dosage (tonnes/hectare) 20	32.5	47.5	65	p	l.s.d.
Initial NO_3 - N	13.2	12.5	26.5	24	30.7	0.05	14.5
NO_3 - N mineralised	28.7	56.4	109.4	99.7	148.7	0.01	56.1
NH_4 - N mineralised	0	0	0	0	0	n.s.	-
Total mineralised	28.7	56.4	109.4	99.7	148.7	0.01	56.1

l.s.d. = least significant difference.

n.s. = not significant.

TABLE 57. Mineralised nitrogen (μg/g) during 28 days incubation, October 1981.

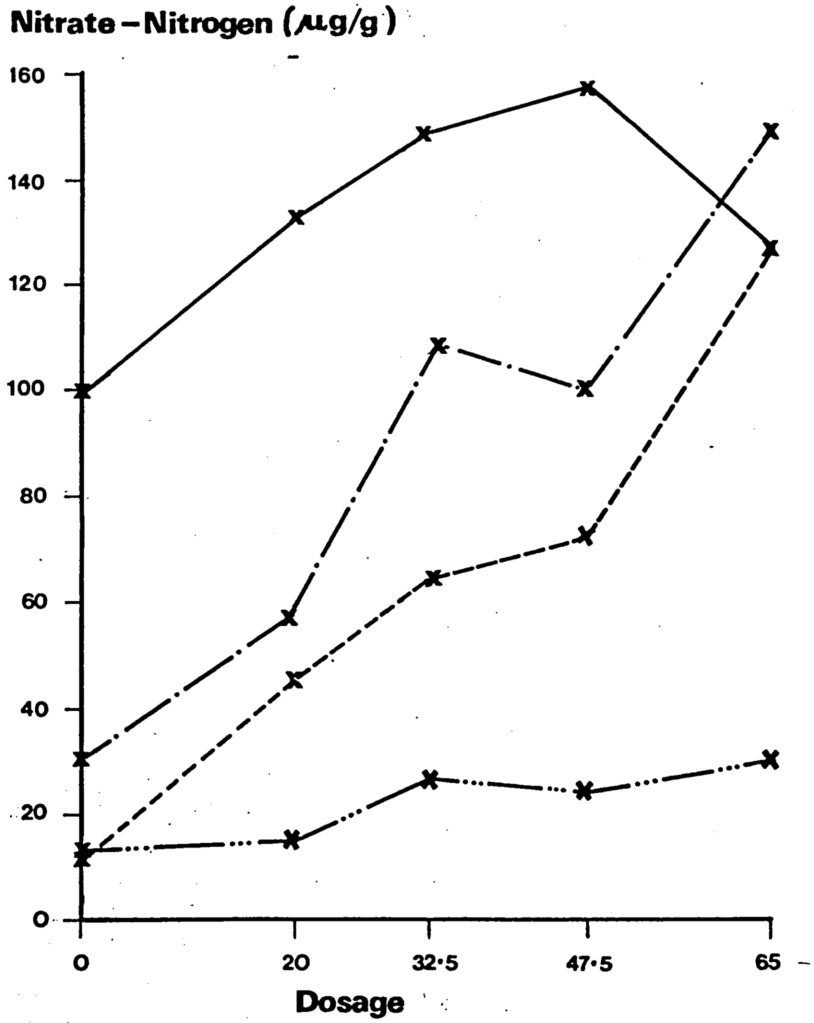


FIG. 22
Soil Nitrate – Nitrogen levels and Mineralised Nitrogen during 4 weeks incubation

KEY

———	=	**Mineralised – N (1980)**
—·—	=	**Mineralised – N (1981)**
-----	=	**Soil NO_3 – N (OCT 1980)**
—···—	=	**Soil NO_3 – N (OCT 1981)**

the first year (Table 56) soil nitrate-nitrogen at all dosages was significantly different from the control ($p < 0.001$) and increased with increasing dosage. Variability within the treatments was high so that the differences in the amount of nitrate-nitrogen produced by mineralisation over four weeks at the different dosages were not significant ($p > 0.05$). However, the data does seem to suggest that there was an increase in nitrate-nitrogen production with increasing dosage followed by a fall off at the top dosage. The fact that the amount of ammonium-nitrogen decreases with increasing dosage implies that it is the mineralisation of organic-nitrogen to ammonium-nitrogen rather than nitrification which is inhibited at the highest dosage.

Fourteen months after refuse incorporation (Table 57) only the highest dosage of 65 tonnes/hectare had a soil nitrate-nitrogen content different from the control ($p < 0.05$). Levels of mineralised nitrogen after incubation generally increased with increasing dosage but differences between dosages of 32.5, 47.5 and 65 tonnes/hectare again were not significant ($p > 0.05$). There was no evidence of either immobilisation, or inhibition of nitrification (levels of ammonium-nitrogen were negligible after incubation) by this time.

4.3.2 Discussion of results

The implications from these experiments fit in closely with those discussed in Section 4.2.3. Three months after initial incorporation of the refuse separates mineralisation of nitrogen was inhibited at high dosage, although the high degree of variability would suggest that even here there were pockets where nitrate-nitrogen production was rapid. Twelve months later there appeared to be increased availability of nitrogen related to increasing dosage, although variability was again too high to allow for conclusive interpretation of results. It is an inherent difficulty of incubation experiments such as the one described to maintain constant moisture levels in samples which are respiring at different rates. Where moisture differences do occur, these will have a critical influence on the activity of nitrifying microorganisms and thus distort the results.

4.3.3 Conclusion

During the first few months of biodegradation, mineralisation of organic nitrogen was inhibited above the dosage of 47.5 tonnes/hectare. After twelve months no inhibition was noted and production of plant available nitrogen increased with increasing dosage.

CHAPTER 5

CONCLUSIONS AND RECOMMENDATIONS

5.1 Soil as an effective means of disposal

Incorporation of the non-composted Doncaster waste putrescibles into surface soil provided an effective means of disposal. An exponential pattern of decomposition resulting from soil microbiological processes stabilised the wastes within twelve months after initial incorporation.

Earlier investigations on the direct soil disposal of unsorted domestic refuse have concentrated on the need to add a source of nitrogen to aid the breakdown of the highly carbonaceous waste; traditionally digested sewage sludge has been used for this purpose. The putrescible waste separates used in this investigation were, however, found to have a carbon:nitrogen ratio of 32.2:1 compared with a ratio of 65:1 for unsorted refuse (King et. al., 1974; 1977; Cottrell, 1975). Coupled with the buffering effect of the nitrogen in the soil, this meant that nitrogen shortage did not limit biodegradation except only in the very short term. No difference in rate of breakdown was found between refuse separates applied to the soil, and that applied in combination with sewage sludge at a total application rate of 20 tonnes/hectare - a spreading of approximately 5 cm depth. The two waste materials could, however, be successfully co-disposed.

Dosages of up to 65 tonnes/hectare were effectively disposed of in the soil. Rate of breakdown, however, showed a quadratic response with dosage, increasing with organic loading up to a dosage rate of 47.5 tonnes/hectare and thereafter declining. At the highest dosage applied in this investigation, organic carbon levels were three times those in the control soil after 18 months decomposition.

Although growth room and laboratory incubation experiments indicated some degree of nitrogen immobilisation during the early stages of decomposition at high dosage, there was no evidence that this significantly retarded either the rate of decomposition of the separates or the release of plant available nitrogen in the longer term. It is, therefore, suggested that the need for nitrogen supplementation following the addition of domestic refuse to soil has been overstated, and that the nitrogen content of the soil itself will act as a buffer against significant microbial immobilisation processes.

5.2 Effects on the soil - beneficial and detrimental

Overall fertiliser benefits to the soil of the Doncaster putrescibles was assessed in terms of plant growth. The refuse provided increased levels of most plant nutrients (Appendix B). Yield and growth rate were increased on the refuse amended soil, but showed a quadratic response to increasing dosage rising to a maximum and then declining at very high rates. This response was, however, reduced over time corresponding closely with early immobilisation of available nitrogen and then later nitrification. Other suggested reasons for the early retarded growth at high dosage are phytotoxicity caused by high concentrations of ammonium-nitrogen and boron, both of which reduced rapidly with time.

Although plant yields on the refuse amended soils were higher than those on the untreated soil, the overall manurial value of the putrescibles is relatively low in comparison with other fertilisers, although they did compare favourably with the digested sewage sludge used in this investigation.

Emphasis in the experimentation was laid upon nitrogen, although the refuse also supplies other fertiliser elements (Appendix B). The total nitrogen input from the refuse separates, which contained 1.02% nitrogen can be calculated from the different dosages (Table 58). The table also shows the amount of nitrogen which will be supplied to the plants by mineralisation, based on the percentage loss of nitrogen during the first year (from Table 33).

Normal fertiliser application varies from about 80 kg Nitrogen/ hectare for upland districts to 120 kg Nitrogen/hectare in dairying areas (based on data collected in 1977, M.A.F.F. 1980) although grass-lands can utilise up to 525 kg Nitrogen/hectare applied in sewage sludge (D.O.E. 1977b). Thus it would appear that, depending upon the dosage applied , the Doncaster separates could provide some, or even all of the nitrogen required for productive grassland areas. However, this assumes that all the organic nitrogen lost in the first

	Dosage tonnes/hectare			
	20	32.5	47.5	65
Total applied N (kg/ha)	204	332	485	663
Mineralisation Rate	19%	37%	25%	37%
N supplied to plants (kg/N/ha/yr)	39	123	122	239

TABLE 58 . Nitrogen supplied by Doncaster refuse separates at different dosages.

12 months is mineralised to plant available forms, and there are no losses. In reality, it has been shown that some will be lost by volatilisation early in biodegradation, and leaching losses may also be significant.

It has been suggested that the Doncaster putrescibles could provide a low cost "manure" to derelict land or nutrient poor sites. According to Jefferies et. al. (1981) mine and mineral spoils reclaimed to grassland commonly require applications of 50 - 200 kg Nitrogen/hectare/year to sustain growth. So long as the dosage was above approximately 30 tonnes/hectare the Doncaster refuse could provide this, but only in the first year. Repeated applications would be needed after this since the mineralisation of nitrogen declines after the first twelve months. This, of course, would introduce a problem of build up of toxic material, referred to later.

Application rates of sewage sludge are often limited by potential pollution from percolation of nitrate into groundwater. The highest recorded level of nitrate-nitrogen on any of the refuse treated plots was 126 µg/g which would equate to 282.2 kg NO_3/hectare based on the commonly used assumption that one hectare of soil, to a depth of 15 cm weighs 2.24×10^6 kg (Fitter, 1974). Further investigations would be necessary to determine whether this concentration in the soil is reflected in increased groundwater nitrate.

Two and a half years after incorporation of the refuse separates certain beneficial effects on soil physical properties were still evident. Water retention at 15 bars (permanent wilting point) was increased by both the refuse and sewage sludge, more significantly at higher dosages, but there was no change in available water capacity. No quantification of other physical properties was undertaken but a visual appraisal of soil profiles after two and a half years showed an obvious improvement in soil structure (Plates 4 and 5)

Humification of the organic matter was an erratic process with a gradual build up of humic acids on all treated plots relative to the control, and a concurrent degradation of some of the humic material, particularly at high dosages. Darker areas of humic material can be clearly seen in Plates 7 and 8.

PLATE 4. Soil control.

Note densely-packed angular blocky aggregates.

PLATE 5. Soil treated with refuse separates at a dosage rate of 47.5 tonnes/hectare. Note well-structured profile with rounded loosely-packed aggregates.

PLATE 6. Soil treated with refuse separates at a dosage rate of 20 tonnes/hectare (Treatment 2).

PLATE 7. Soil treated with digested sewage sludge at a dosage rate of 20 tonnes/hectare.

Note darker areas of humic material.

PLATE 8. Soil treated with refuse separates at a dosage rate of 47.5 tonnes/hectare. Note darker areas of humic material and persistence of non-biodegradable material.

There were no apparent symptoms of phytotoxicity in the field plots but soil analyses revealed that high levels of boron could be a problem immediately after incorporation. This was leached out fairly quickly and levels were safe after twelve months, although isolated samples still showed high concentration after this time at the top dosage of 65 tonnes/hectare.

Zinc, copper, lead and cadmium were all at acceptable levels for areas to be used as amenity grassland and public open space. Agricultural usage of the waste-treated land is not envisaged in South Yorkshire. High levels of lead and cadmium were found in isolated patches, emphasising the heterogeneity of the refuse and the need for strict monitoring of the refuse treated land. High cadmium concentrations were found associated with the sewage sludge treated plots. Copper and lead were found to be unavailable to plants in the form in which they occurred in the refuse, but zinc, cadmium and boron were all taken up into the plant tissue. Phytotoxicity was not a problem, although it is thought that high levels of boron caused some inhibition of germination in the dosage plots which were seeded immediately after incorporation of the refuse. Zinc and cadmium availability was reduced over time either by reversion to less available forms, or by binding with the more stable organic matter produced by biodegradation.

The occurrence of pathogenic organisms in organic wastes is another potential hazard arising from the soil disposal of the Doncaster refuse putrescibles. It is considered that the high temperatures obtained when composting refuse (up to 65°C) will kill most pathogenic organisms, so that municipal compost is normally innocuous in this sense, but there is little information available on the survival of pathogens in refuse applied directly to the land. Hartenstein (1981), however, believes that humification in the soil may be as important as high temperatures in compost piles in the destruction of pathogens. Wright (1974) pointed out that sewage sludge requires sterilisation to destroy pathogens before they can be applied to soil; without this most pathogens will only survive in the soil for several days to a few months, although some (e.g. the eggs of intestinal worms such as *Ascaris lumbricoides* can survive for a number of years (Epstein and Chaney, 1978)).

A major drawback to the soil disposal of the Doncaster refuse is the persistence of non-biodegradable materials for which initial separation at the recycling plant is difficult. After fifteen months there were still relatively large amounts of glass, stones, plastic and coal clinker evident both at the surface and at depth on some of the plots (Plates 5 and 8). Apart from their deleterious appearance in the soil after most of the organic material has degraded, they would be considered a safety hazard in public amenity areas. It would appear, therefore, that the constraints to this method of disposal are not environmental, but depend on the ability of the recycling process to effect complete removal of these inert materials.

5.3 Recommendations for management of disposal sites

Successful disposal of the refuse putrescibles will depend on knowledge of the site, and careful management of it during and after application. The following suggestions could be used as guidelines.

1) The refuse separates should be characterised with respect to elemental composition prior to land disposal.

2) Potential toxic metals and soluble salts should be monitored at suitable intervals for at least three years after incorporation and Government safety guidelines should be observed. If 'total' or 'available' metal concentrations are at unacceptable levels ameliorative measures such as liming, deep cultivation or irrigation (to reduce the soluble salts) could be adopted.

3) Appropriate choice of plant species for landscaping treated areas will be those affording a reasonable tolerance to metals and soluble salts and possibly those which can utilise ammonium-nitrogen. Most of the popular landscaping grasses would be suitable and those which flourish in acidic or wet environments, e.g. *Festuca ovina*.

4) The refuse treated soil should ideally be left for three to six months before being seeded to allow for possible problems

of salt phytotoxicity to subside.

5) Dosage rates are likely to be governed largely by amounts of refuse requiring disposal and area of available land. They should, however, take account of the fact that decomposition may be retarded at dosages above approximately 50 tonnes/hectare. Costs of mowing and maintenance should also be considered since fairly low dosages will enhance plant growth by a significant amount. Slow growing species may be more suitable if this consideration is important.

6) Repeated applications (e.g. at annual intervals) could be made, but are likely to be restrained by toxic metal build up, not by organic overload.

These recommendations are based on the findings of this investigation but it must be pointed out that the nature of the material itself is still changing with engineering developments at the recycling plant. In addition, the behaviour of the separates may differ slightly from one site to another since any soil will have a unique set of characteristics with which it is equipped, or otherwise, to provide a safe and effective medium for waste degradation. Sommers et. al. (1979), however, found that soil properties were not significant factors in determining the extent of decomposition after sewage sludge incorporation into five different soil types.

5.4 Suggestions for further work

The present work has suggested several areas requiring further investigation, on both a practical and theoretical level. At the fairly low dosage of 20 tonnes/hectare it was found that measurement of rates and patterns of breakdown was complicated by the increased amounts of soil organic matter resulting from the decay of extra plant material. Those plots which supported the greatest plant biomass tended to be the ones which were expected to show the highest rates of organic matter decomposition and nitrogen mineralisation. More information on rates and patterns of biodegradation could be obtained by litter-bag techniques, respirometry or the use of ^{14}C-labelled material which would permit direct measurement of $^{14}CO_2$ evolved as an index of degradation.

Although results suggest that most heavy metals contained in wastes are rather immobile in soils, long-term studies are needed to gain an understanding of their behaviour with time and the extent to which the metals complex with added organic matter, in a fresh, or stabilised state. More greenhouse and laboratory work is needed to correlate heavy metal uptake with soil extraction procedures; this could lead to more consistency in usage. Water percolation studies in the field and the use of lysimeters could follow movement through the soil, not only of heavy metals, but also of nitrate produced during the decomposition of the refuse putrescibles.

The influence of refuse additions on soil physical properties, the major fertiliser elements, and plant yield has been fairly well established but there is evidence that compost additions may result in reduced crop content of certain micronutrients such as magnesium (de Haan, 1981) and manganese (Hortensteine and Rothwell, 1973; Gray and Biddlestone, 1980; de Haan, 1981). Since results here show that magnesium levels in the plots treated with the Doncaster refuse are decreased (Appendix B) further work is needed to establish whether the increased growth resulting from augmented levels of macronutrients, may cause some micronutrient deficiencies.

There is a need to assess the growth potential of the refuse-amended soil for a variety of plant species, particularly those which would be commonly used in landscaping schemes. The Doncaster refuse putrescibles have possibilities as a low grade fertiliser and stabilising agent on unproductive land contaminated with other waste materials such as colliery shale, china clay waste and pulverised fuel ash. Future work could investigate the behaviour of the putrescibles when incorporated into these materials although certain problems are envisaged, e.g. high boron content of pulverised fuel ash, low nitrogen on china-clay waste, and high metal solubility if the colliery shale is acidic. Associated with this, research would also be needed into the effect of the refuse application on the physical stability of the receiving medium, surface runoff and erosion-sediment transport.

5.5 Acceptability of direct soil disposal as a waste management scheme

One remaining area of discussion is whether the disposal method outlined in this thesis will be a viable proposition in South Yorkshire, or in any other Waste Disposal Authority. It will only be viable if it is economically sound.

There seem to have been two major problems associated with the practice of direct soil disposal of domestic refuse in the past. The first is the difficulty in obtaining a consistent product after separation of plastics and bulking items so as to permit proper incorporation into the soil. The second is the non-breakdown of items for which initial separation is difficult. The sophisticated recycling plant at Doncaster does produce a material which is reasonably consistent and workable, and in its present form seems well suited to soil disposal. The putrescible fraction is considered to be a 'non-useful' by-product from the recycling plant so that separation costs are negligible. The second problem, that of recalcitrant and ugly material in the waste, remains an engineering problem.

Composting of the refuse prior to incorporation would have the advantages of producing a less bulky, more pathogen-free more workable and publically acceptable material. However, since neither bulk nor workability were found to be a problem, and pathogen removal is likely to be completed along with the humification of the material, composting may be an unnecessary step here, merely adding time and increasing disposal costs.

Land use charges would be low or non-existent if the same local authority wished to dispose of the waste and also upgrade areas of derelict land, although transport costs may be unattractive in comparison to those required to reach landfill sites.

Some other features of this waste disposal system may limit its flexibility. Firstly, it cannot be applied on frozen or snow-covered areas in winter. Secondly, it cannot be applied on days when the soil is too wet to permit the movement of heavy equipment distributing the waste. Thirdly, if large tonnages are to be disposed of, the land requirements may well exceed amounts which are available. Soil incorporation of separated waste could, therefore, provide an auxilliary system of disposal, but it is unlikely to be a complete alternative to present methods.

" We must learn, or relearn, the habit of regarding waste material as potentially valuable resources. The world cannot afford the luxury of a throwaway society. "

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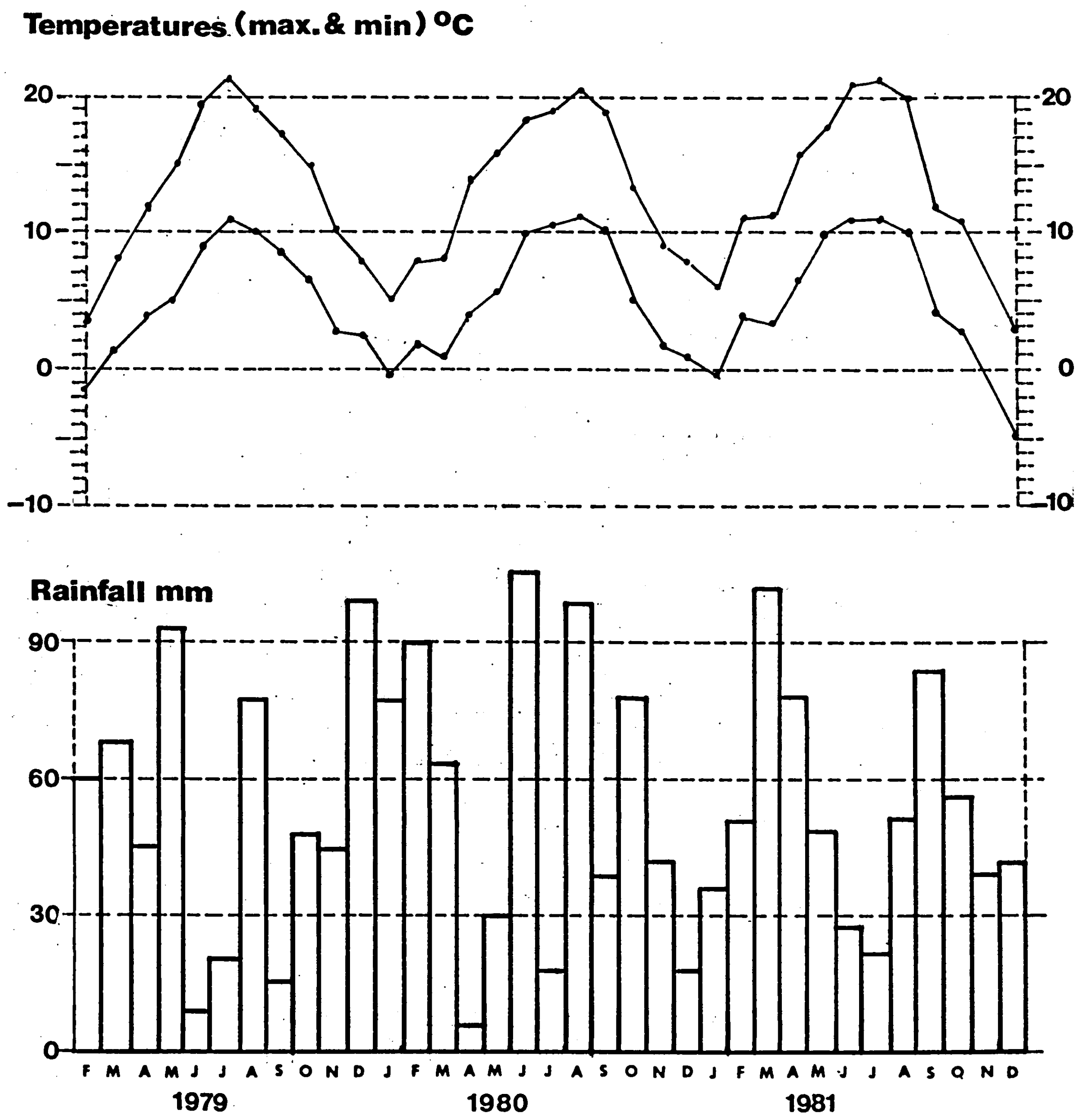

Mean monthly temperatures (maximum & minimum) and total monthly precipitation Finningley Meteorological Station

<u>Details of snowfall : Winter 1978/79</u> (recorded at Finningley Meteorological Station)

December 1978	8 days of snow or sleet 3 days on which snow lay on the ground at 0900 greatest depth : 7 cm on December 31st.
January 1979	14 days of snow or sleet 21 days on which snow lay on the ground at 0900 greatest depth : 8 cm on January 1st and 2nd.
February 1979	9 days of snow or sleet 12 days on which snow lay on the ground at 0900 greatest depth : 8 cm on February 16th - 19th.
March 1979	12 days of snow or sleet 5 days on which snow lay on the ground at 0900 greatest depth : 6 cm on March 17th.
April 1979	4 days of snow or sleet 0 days on which snow lay on the ground at 0900.

Snow lay continuously from 19th to 31st January, and from 12th to 21st February.

Sample		Available (ppm)					Total (ppm)					
	pH	P	K	Mg	NO_3-N	B	Cu	Ni	Zn	Cd	Pb	Cond.* µS
Soil (control)	7.8	17	127	443	9	0.88	24	22	120	1.5	55	1968
Refuse alone	6.6	256	2945	314	37	6.82	127	29	210	0.9	276	5979
20 tonnes/hectare	7.6	22	200	209	9	1.38	56	23	155	1.7	67	2009
20 tonnes/hectare x 2	7.8	28	399	206	12	1.44	56	24	150	1.4	68	2046
65 tonnes/hectare	7.4	94	1125	310	150	5.88	136	43	410	1.9	462	2960

* Conductivity (µS) measured in saturated $CaSO_4$

<u>Analyses of soil/refuse samples from field plots</u> (courtesy of MAFF (ADAS), Reading).

APPENDIX C

PLANT GROWTH DATA

Time (Days)	Soil Control						20% Refuse					
	Log* Y	Lower Conf. Limit	Upper Conf. Limit	DlogY/DT	Lower Conf. Limit	Upper Conf. Limit	Log* Y	Lower Conf. Limit	Upper Conf. Limit	DlogY/DT	Lower Conf. Limit	Upper Conf. Limit
0	-5.7753	-6.2376	-5.3130	0.1445	0.1091	0.1799	-5.8975	-6.2454	-5.5496	0.1602	0.1335	0.1869
6	-4.9509	-5.2798	-4.6219	0.1303	0.1007	0.1599	-4.9734	-5.2210	-4.7259	0.1478	0.1255	0.1702
13	-4.0967	-4.3680	-3.8254	0.1137	0.0907	0.1368	-3.9889	-4.1930	-3.7847	0.1335	0.1161	0.1508
21	-3.2625	-3.5627	-2.9623	0.0948	0.0786	0.1110	-2.9870	-3.2129	-2.7612	0.1170	0.1048	0.1292
32	-2.3628	-2.7163	-2.0093	0.0688	0.0581	0.0794	-1.8244	-2.0904	-1.5584	0.0944	0.0864	0.1024
42	-1.7934	-2.1545	-1.4323	0.0451	0.0311	0.0591	-0.9834	-1.2551	-0.7117	0.0738	0.0633	0.0844
67	-1.4055	-1.9944	-0.8165	-0.0141	-0.0503	0.0222	0.2196	-0.2236	0.6628	0.0224	-0.0049	0.0495

* Fitted log Y is derived from four replicates.

Effect of Dosage Rate (laboratory experiment). Fitted values for yield (log Y) and relative growth rate ($\frac{D \log Y}{DT}$), plus 95% confidence limits.

PLANT GROWTH DATA

Time (Days)	40% Refuse						60% Refuse					
	Log* Y	Lower Conf. Limit	Upper Conf. Limit	$\frac{DlogY}{DT}$	Lower Conf. Limit	Upper Conf. Limit	Log* Y	Lower Conf. Limit	Upper Conf. Limit	$\frac{DlogY}{DT}$	Lower Conf. Limit	Upper Conf. Limit
0	-6.0023	-6.3875	-5.6171	0.1669	0.1369	0.1968	-5.8650	-6.1888	-5.5411	0.1667	0.1416	0.1918
6	-5.0402	-5.3139	-4.7666	0.1538	0.1289	0.1788	-4.9011	-5.1311	-4.6711	0.1546	0.1337	0.1755
13	-4.0167	-4.2433	-3.7900	0.1386	0.1193	0.1579	-3.8685	-4.0590	-3.6780	0.1404	0.1242	0.1566
21	-2.9772	-3.2282	-2.7263	0.1212	0.1077	0.1348	-2.8097	-3.0207	-2.5988	0.1243	0.1129	0.1356
32	-1.7751	-2.0692	-1.4810	0.0973	0.0879	0.1068	-1.5651	-1.8123	-1.3179	0.1020	0.0941	0.1100
42	-0.9103	-1.2137	-0.6069	0.0756	0.0625	0.0887	-0.6458	-0.9008	-0.3908	0.0818	0.0708	0.0928
67	0.3011	-0.2605	0.8628	0.0213	-0.0114	0.0540	0.7682	0.2960	1.2403	0.0313	0.0038	0.0588

* Fitted log Y is derived from four replicates.

Effect of Dosage Rate (laboratory experiment). Fitted values for yield (log Y) and relative growth rate ($\frac{D\ logY}{DT}$), plus 95% confidence limits.

Time (Days)	80% Refuse						100% Refuse					
	Log* Y	Lower Conf. Limit	Upper Conf. Limit	$\frac{D \log Y}{DT}$	Lower Conf. Limit	Upper Conf. Limit	Log* Y	Lower Conf. Limit	Upper Conf. Limit	$\frac{D \log Y}{DT}$	Lower Conf. Limit	Upper Conf. Limit
0	-6.1106	-6.6321	-5.5892	0.1395	0.0987	0.1803	-6.0077	-6.4887	-5.5263	-0.0034	-0.442	0.0374
6	-5.2999	-5.6686	-4.9311	0.1307	0.0966	0.1649	-5.9866	-6.3229	-5.6504	0.0104	-0.0228	0.0435
13	-4.4208	-4.7276	-4.1141	0.1204	0.0938	0.1471	-5.8576	-6.1453	-5.5699	0.0265	0.0016	0.0513
21	-3.5044	-3.8509	-3.1579	0.1087	0.0899	0.1274	-5.5725	-5.8981	-5.2468	0.0448	0.0276	0.0621
32	-2.3979	-2.8114	-1.9845	0.0925	0.0804	0.1046	-4.9406	-5.3174	-4.5637	0.0701	0.0537	0.0864
42	-1.5464	-1.9706	-1.1222	0.0778	0.0621	0.0935	-4.1251	-4.5453	-3.7049	0.0930	0.0675	0.1185
67	-0.0605	-0.7226	0.6016	0.0411	-0.0001	0.0822	-1.0825	-2.2270	0.0620	0.1504	0.0928	0.2080

* Fitted log Y is derived from four replicates.

Effect of Dosage Rate (laboratory experiment). Fitted values for yield (log Y) and relative growth rate ($\frac{D \log Y}{DT}$), plus 95% confidence limits.

PLANT GROWTH DATA

Time (Days)	Soil Control						20 tonnes/hectare					
	Log* Y	Lower Conf. Limit	Upper Conf. Limit	$\frac{DlogY}{DT}$	Lower Conf. Limit	Upper Conf. Limit	Log* Y	Lower Conf. Limit	Upper Conf. Limit	$\frac{DlogY}{DT}$	Lower Conf. Limit	Upper Conf. Limit
36	0.2430	0.0392	0.4469	0.0395	0.0313	0.0477	0.3478	-0.1396	0.8352	0.0576	0.0380	0.0771
66	1.2046	1.0722	1.3371	0.0246	0.0201	0.0291	1.7483	1.4316	2.0650	0.0358	0.0250	0.0466
95	1.7104	1.5582	1.8625	0.0103	0.0080	0.0125	2.4816	2.1178	2.8454	0.0148	0.0093	0.0202
127	1.7849	1.6537	1.9162	-0.0056	-0.0103	-0.0009	2.5827	2.2689	2.8966	-0.0084	-0.0197	0.0028
155	1.4340	1.2326	1.6355	-0.0195	-0.0276	-0.113	2.0619	1.5801	2.5437	0.0288	-0.0482	-0.0093

* Fitted log Y is derived from three replicates.

Effect of Dosage Rate (field experiment). Fitted values for yield and relative growth rate ($\frac{DlogY}{DT}$), plus 95% confidence limits.

PLANT GROWTH DATA

Time (Days)	32.5 tonnes/hectare						47.5 tonnes/hectare					
	Log* Y	Lower Conf. Limit	Upper Conf. Limit	$\frac{DlogY}{DT}$	Lower Conf. Limit	Upper Conf. Limit	Log* Y	Lower Conf. Limit	Upper Conf. Limit	$\frac{DlogY}{DT}$	Lower Conf. Limit	Upper Conf. Limit
36	-0.5316	-1.2568	0.1936	0.0915	0.0624	0.1205	-1.0115	-1.8694	-0.1537	0.0997	0.0654	0.1341
66	1.7062	1.2350	2.1774	0.0577	0.0416	0.0738	1.4343	0.8770	1.9917	0.0733	0.0443	0.0823
95	2.9063	2.3650	3.4476	0.0251	0.0169	0.0332	2.7601	2.1198	3.4004	0.0281	0.0185	0.0377
127	3.1318	2.6648	3.5988	-0.0110	-0.0277	0.0058	3.0381	2.4857	3.5905	-0.0107	-0.0306	0.0091
155	2.3835	1.6667	3.1003	-0.0425	-0.0714	-0.0136	2.2615	1.4136	3.1095	-0.0447	-0.0789	-0.0105

* Fitted log Y is derived from three replicates.

Effect of Dosage Rate (field experiment). Fitted values for yield and relative growth rate ($\frac{DlogY}{DT}$) plus 95% confidence limits.

Time (Days)	75 tonnes/hectare					
	log* Y	Lower Conf. Limit	Upper Conf. Limit	$\frac{DlogY}{DT}$	Lower Conf. Limit	Upper Conf. Limit
36	-2.0458	-2.5695	-1.5221	0.1229	0.1019	0.1439
66	0.9760	0.6367	1.3172	0.0786	0.0670	0.0903
95	2.6378	2.2469	3.0287	0.0359	0.0300	0.0418
127	3.0317	2.6945	3.3689	-0.0113	-0.0234	0.0008
155	2.1381	1.6205	2.6558	-0.0526	-0.0734	-0.0317

* Fitted log Y is derived from three replicates.

Effect of Dosage Rate (field experiment). Fitted values for yield and relative growth rate ($\frac{DlogY}{DT}$), plus 95% confidence limits.

Time (Days)	Soil Control						Soil + Refuse					
	Log* Y	Lower Conf. Limit	Upper Conf. Limit	$\frac{DlogY}{DT}$	Lower Conf. Limit	Upper Conf. Limit	Log* Y	Lower Conf. Limit	Upper Conf. Limit	$\frac{DlogY}{DT}$	Lower Conf. Limit	Upper Conf. Limit
1	-5.8965	-6.1716	-5.6213	0.1372	0.1160	0.1583	-5.7696	-6.1171	-5.4221	0.1253	0.0986	0.1520
5	-5.3656	-5.5879	-5.1433	0.1283	0.1093	0.1472	-5.2812	-5.5620	-5.0005	0.1189	0.0950	0.1428
16	-4.0888	-4.2728	-3.9049	0.1039	0.0908	0.1169	-4.0703	-4.3027	-3.8380	0.1013	0.0848	0.1178
29	-2.9261	-3.1596	-2.6927	0.0750	0.0678	0.0822	-2.8890	-3.1839	-2.5941	0.0805	0.0714	0.0895
43	-2.0935	-2.3415	-1.8456	0.0439	0.0372	0.0507	-1.9194	-2.2326	-1.6063	0.0580	0.0495	0.0666
71	-1.7335	-2.0417	-1.4252	-0.0182	-0.0387	0.0023	-0.9217	-1.3110	-0.5323	0.0132	-0.0127	0.0391

* Fitted log Y is derived from five replicates.

Effect of Additives (Growth Room experiment). Fitted values for yield (log Y) and Relative Growth Rate ($\frac{DlogY}{DT}$).

Time (Days)	Soil + Sewage Sludge						Soil + Refuse + Sewage Sludge					
	log* Y	Lower Conf. Limit	Upper Conf. Limit	DlogY/DT	Lower Conf. Limit	Upper Conf. Limit	log* Y	Lower Conf. Limit	Upper Conf. Limit	DlogY/DT	Lower Conf. Limit	Upper Conf. Limit
1	-5.7426	-6.0833	-5.4019	0.1541	0.1279	0.1802	-5.8694	-6.1063	-5.6326	0.1545	0.1363	0.1727
5	-5.1453	-5.4206	-4.8700	0.1446	0.1211	0.1680	-5.2692	-5.4606	-5.0779	0.1456	0.1293	0.1619
16	-3.6986	-3.9264	-3.4708	0.1185	0.1023	0.1347	-3.8025	-3.9609	-3.6442	0.1211	0.1098	0.1323
29	-2.3591	-2.6483	-2.0700	0.0876	0.0787	0.0965	-2.4170	-2.6180	-2.2160	0.0921	0.0859	0.0982
43	-1.3653	-1.6723	-1.0582	0.0544	0.0460	0.0628	-1.3464	-1.5598	-1.1329	0.0609	0.0550	0.0667
71	-0.7732	-1.1550	-0.3915	-0.0121	-0.0375	0.0133	-0.5162	-0.7816	-0.2508	0.0016	-0.0192	0.0161

* Fitted log Y is derived from five replicates.

Effect of Additives (Growth Room experiment). Fitted values for yield (log Y) and Relative Growth Rate ($\frac{DlogY}{DT}$).

PLANT GROWTH DATA - Response Surface Analysis

Y computed Relative Growth Rate (D logY/DT)	X1 Time (Days)	X2 Dosage (Tonnes/hectare)
0.0393	36	0
0.0242	66	0
0.0097	95	0
- 0.0063	127	0
- 0.0204	155	0
0.0643	36	20
0.0403	66	20
0.0172	95	20
- 0.0084	127	20
- 0.0307	155	20
0.0812	36	32.5
0.0513	66	32.5
0.0223	95	32.5
- 0.0096	127	32.5
- 0.0375	155	32.5
0.1025	36	47.5
0.0651	66	47.5
0.0290	95	47.5
- 0.0109	127	47.5
- 0.0457	155	47.5
0.1240	36	65
0.0792	66	65
0.0359	95	65
- 0.0119	127	65
- 0.0537	155	65

Effect of Dosage Rate (field experiment). Relative growth rate as a function of time and dosage.

PLANT GROWTH DATA - Response Surface Analysis

Y computed Yield (log Y)	X1 Time (Days)	X2 Dosage^2 (Tonnes/hectare)
- 0.1495	36	0
1.4753	66	0
2.1845	95	0
1.9842	127	0
0.9632	155	0
- 0.0918	36	400
1.7617	66	400
2.6921	95	400
2.7359	127	400
1.9284	155	400
- 0.2943	36	1056.25
1.7117	66	1056.25
2.7895	95	1056.25
2.9960	127	1056.25
2.3308	155	1056.25
- 0.8185	36	2256.25
1.3782	66	2256.25
2.6403	95	2256.25
3.0501	127	2256.25
2.5629	155	2256.25
- 1.6439	36	4225.0
0.7435	66	4225.0
2.1898	95	4225.0
2.8030	127	4225.0
2.4937	155	4225.0

<u>Effect of Dosage Rate (field experiment)</u>. Yield as a function of time and dosage squared.

PLANT GROWTH DATA - Response Surface Analysis

Y computed Relative Growth Rate (D logY/DT)	X1 Time (Days)	X2 Dosage (% Refuse)
0.0472	0	100
0.0503	6	100
0.0538	13	100
0.0578	21	100
0.0634	32	100
0.0684	42	100
0.0810	67	100
0.0996	0	80
0.0983	6	80
0.0968	13	80
0.0951	21	80
0.0927	32	80
0.0905	42	80
0.0851	67	80
0.1373	0	60
0.1316	6	60
0.1251	13	60
0.1176	21	60
0.1072	32	60
0.0979	42	60
0.0744	67	60
0.1602	0	40
0.1502	6	40
0.1386	13	40
0.1253	21	40
0.1071	32	40
0.0905	42	40
0.0490	67	40
0.1683	0	20
0.1541	6	20
0.1374	13	20
0.1184	21	20
0.0922	32	20
0.0684	42	20
0.0089	67	20
0.1618	0	0
0.1432	6	0
0.1215	13	0
0.0967	21	0
0.0625	32	0
0.0315	42	0
0.0460	67	0

Effect of Dosage Rate (growth room experiment). Relative growth rate as a function of time and dosage.

PLANT GROWTH DATA - Response Surface Analysis

Y computed Yield (Log Y)	X1 Time (Days)	X2 Dosage (% Refuse)
- 7.3060	0	100
- 6.5411	6	100
- 5.7078	13	100
- 4.8334	21	100
- 3.7667	32	100
- 2.9334	42	100
- 1.4183	67	100
- 6.1820	0	80
- 5.4226	6	80
- 4.5958	13	80
- 3.7287	21	80
- 2.6722	32	80
- 1.8482	42	80
- 0.3561	67	80
- 5.4921	0	60
- 4.7383	6	60
- 3.9179	13	60
- 3.0582	21	60
- 2.0119	32	60
- 1.1970	42	60
0.2720	67	60
- 5.2363	0	40
- 4.4880	6	40
- 3.6741	13	40
- 2.8219	21	40
- 1.7857	32	40
- 0.9801	42	40
0.4659	67	40
- 5.4147	0	20
- 4.6720	6	20
- 3.8645	13	20
- 3.0196	21	20
- 1.9936	32	20
- 1.1972	42	20
0.2256	67	20
- 6.0273	0	0
- 5.2901	6	0
- 4.4891	13	0
- 3.6516	21	0
- 2.6357	32	0
- 1.8485	42	0
- 0.4488	67	0

<u>Effect of Dosage Rate (growth room experiment)</u>. Yield as a function of time and dosage.

	1	2	3	4	5	6	7	8
A	1	2	3	4	5	6	7	8
B	1	3	5	7	2	4	6	8
C	2	4	6	8	1	3	5	7
D	8	6	4	2	7	5	3	1
E	3	2	1	7	4	6	5	8
F	7	5	3	8	2	4	6	1
G	6	7	8	1	2	3	4	5
H	4	1	7	6	2	5	8	3
I	3	6	8	7	4	5	1	2
J	8	7	6	5	4	3	2	1

Randomised block layout of additives field trials.

(Treatment numbers given in boxes)