THESIS

Mehmet Ögüt

The Graduate School
University of Kentucky
1997

CARBON AND NITROGEN MINERALIZATION IN SLUDGE AND FLY ASH-LIMESTONE SCRUBBER RESIDUE (LSR) TREATED SOIL

THESIS

A thesis submitted in partial fulfillment
of the requirements for the degree of
Master of Science
at the University of Kentucky

By

Mehmet Öğüt

Lexington, Kentucky

Director: Dr. M. Coyne, Soil and Plant Science

Lexington, Kentucky

1997

MASTER'S THESIS RELEASE

I authorize the University of Kentucky Libraries to reproduce this thesis in whole or in part for purposes of research.

I do not authorize the University of Kentucky Libraries to reproduce this thesis in whole or in part for purposes of research.

(Include the statement above which applies.)

Signed: ______________________________

Date: ______________________________

ACKNOWLEDGMENTS

CARBON AND NITROGEN MINERALIZATION IN SLUDGE AND FLY ASH-LIMESTONE SCRUBBER RESIDUE (LSR) TREATED SOIL

By

Mehmet Ögüt

Director of Thesis

Director of Graduate Studies

ABSTRACT OF THESIS

CARBON AND NITROGEN MINERALIZATION IN SLUDGE AND FLY ASH-LIMESTONE SCRUBBER RESIDUE (LSR) TREATED SOIL

Carbon and nitrogen mineralization were measured in soil treated with sludge and fly ash-LSR, at rates up to 270 Mg ha^{-1}. The more sludge, the greater the cumulative C and N mineralization. A regression model indicated that a 154 Mg ha^{-1} sludge rate maximized cumulative C mineralization. No model was developed for N mineralization because of considerable unrecovered N attributable to NH_3 volatilization and denitrification. Nitrite accumulated up to 151 mg kg^{-1} soil. Toxicity of NH_3 was a plausible suggestion for delayed nitrification. Additional sludge probably increased the rate and extent of NH_3 volatilization and suppressed nitrifiers for longer periods. The more sludge, the smaller the % sludge organic C and N mineralization. This was attributed to sludge salinity. The sludge rate maximizing % sludge organic C mineralized was 15 Mg ha^{-1}. Fly ash-LSR at rates up to 135 Mg ha^{-1} didn't affect C and N mineralization probably because it had low salinity and a neutral pH.

04-29-1997

RULES FOR THE USE OF THESES

Unpublished theses submitted for the Master's degree and deposited in the University of Kentucky Library are as a rule of inspection, but are to be used only with due regard to the rights of the authors. Bibliographical references may be noted, but quotations or summaries of parts may be published only with the permission of the author, and with the usual scholarly acknowledgments.

Extensive copying or publication of the theses in whole or in part also requires the consent of the Dean of the Graduate School of the University of Kentucky.

A library that borrows this theses for use by its patrons is expected to secure signature of each user.

Name	Date

ACKNOWLEDGMENTS

I am thankful to god for giving me all the necessitates to finish the following work. I owe a great deal of appreciation to my thesis chair, Dr. M. Coyne, for his guidance and patience during this work. He thought me how to think, write, and talk scientifically. I also would like to thank my other committee members, Dr. W. Thom, and Dr. J. Grove, for their critical review of my work. Next, I wish to thank my friends, S. Tarhan and M. Ataman for their help during soil sample preparation. Thanks go to the workers of Agronomy Department at the University of Kentucky for providing a friendly environment and technical supports.

I couldn't finish this work without the supports of my family and friends. Special thanks go to my lovely wife, Fatma who has helped me get trough this work by providing a peaceful life and assisting in laboratory works. Finally, I would like to thank my sponsor, Gaziosmanpasa University, for providing the financial aids needed to finish this work.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER ONE: INTRODUCTION

Production and Disposal of Sewage Sludge

Prior to 1972, wastewater from municipalities was routinely discharged to surface waters, creating water pollution. In 1972, to abate water pollution, the Federal Water Pollution Control Act Amendment required municipalities in the U.S. to treat wastewater before discharging it. The number of wastewater treatment plants in the US has increased since then. This has caused an increase in treated wastewater residues, namely sewage sludge. For example, annual sludge production in the U.S. increased from 6 million Mg in 1983 to 8.5 million Mg in 1990 (EPA, 1984). It is projected to be 12 million Mg by 2000 as the population grows and more advanced sewage treatment techniques are developed (Hasbach, 1991).

The large quantities of sludge produced by wastewater treatment plants must be disposed of properly. Approximately 25 % of the sludge production in the U.S. is landfilled, 50 % is incinerated or stored in lagoons, and 25 % is land applied (Hasbach, 1991). The major limitations to landfilling sludge are the lack of landfill space and the high costs for landfill maintenance. Most landfills will be at capacity in 2 to 3 years (Hasbach, 1991). As these landfills are filled, new sites need to be identified. For these reasons, many states discourage landfilling of sludge. Missouri, Ohio, Oregon, and Washington limit sludge landfilling to 10 % of sludge production. Some states, such as New Jersey and North Carolina, prohibit sludge landfilling (Goldstein, 1991). These states landfill some of their sludge elsewhere, which increases sludge transportation costs.

Although sludge incineration recovers some energy, it may not be a viable long-term solution for sludge management due to stringent regulations on air pollution control. Incineration destroys all pathogens and degrades many toxic organics in sludge. But, incineration doesn't destroy metals in sludge. Instead, it concentrates them in the ash that must be disposed of eventually. The limitations to lagoon storage of sludge, like landfills, include lack of lagoon space and lagoon maintenance costs.

Many states encourage application of sludge to land as a means of sludge disposal. The Environmental Protection Agency (EPA) encourages sludge use as a fertilizer in agriculture, and numerous studies have been conducted to maximize the beneficial effects of sludge on soil properties and crop growth.

These studies demonstrate that sludge can be applied to forests, drastically disturbed lands, and agricultural lands to improve soil physical characteristics such as aggregate stability (Epstein, 1975; Kladivko and Nelson, 1979; Pagliai et. al., 1981) and water retention (Epstein, 1975; Epstein et al., 1976; Gupta et al., 1977), and chemical characteristics such as plant nutrient availability (Harding et al., 1984; Hinedi et al., 1988; Kelling et al., 1977; McLaughlin and Champion, 1987) and cation exchange capacity (Epstein et al., 1976; Kladivko and Nelson, 1979). For these reasons, sludge application onto lands has increased many-fold. A mean of 37 % of the total sludge production in 21 states is land applied (Goldstein, 1991). Colorado, Florida, Oregon, and Washington, for example, land apply over 70 % of sludge.

Production and Disposal of Fly Ash-Limestone Scrubber Residue (LSR)

In the 1970's, regulations to control both water and air pollution were enacted due to increased public awareness of environmental problems. The Clean Air Act of 1977 requires coal burning power plants to outfit each of their smokestacks with a sulfur dioxide (SO_2) scrubbing unit. The unit captures SO_2 with a solid such as limestone dust before it exits to the atmosphere. Captured particles and fly ash (coal combustion residue) fall underneath the boiler. The mixture of the two is referred to as fly ash-Limestone Scrubber Residue (LSR). The annual U.S. production of fly ash-LSR was approximately 16 million Mg in 1984 and is expected to increase to 50 million Mg by 2000 (EPA, 1988).

There are two major methods of fly ash-LSR disposal: settling ponds and landfills. Settling ponds are surface impoundments that receive fly ash-LSR in slurry form. Landfills receive dry fly ash-LSR. Approximately 70 % of the power plants in the U.S. use one or a combination of the two methods (EPA, 1988). The major adverse impact of fly ash-LSR disposal is leaching of toxicants into soils and groundwater. For this reason, states have increased regulation of disposal sites and require groundwater monitoring. These bring additional costs to fly ash-LSR disposal.

Applying fly ash-LSR to agricultural lands has been investigated as a means of improving soil physical properties such as texture (Chang et al., 1977) , aeration (Aitken and Bell, 1985; Chang et al., 1977), and water holding capacity (Chang et al., 1977). Fly ash-LSR also increases the pH of acid soils (Elseewi and Page, 1984; Petruzzelli et al., 1987; Plank et al., 1975). It may also relieve nutrient deficiencies including those of boron, calcium, molybdenum and sulfur (Mulford and Martens, 1971; Plank and Martens, 1974; Wallace et al., 1980; Wallace and Wallace, 1986).

Compared to sludge treated soil, fly ash-LSR treated soil has been the subject of few studies on C and N mineralization (Arthur et al., 1984; Cervelli et al., 1987; Pichtel, 1990). Fly ash-LSR, unlike sludge, has little mineralizable C and N. Fly ash-LSR usually decreases total C and N mineralization. The studies suggest that the decreases in total C and N mineralization are due to high heavy metal content and/or pH in fly ash-LSR (Arthur et al., 1984; Cervelli et al., 1987; Pichtel, 1990). However, these hypotheses aren't well tested.

CHAPTER TWO: LITERATURE REVIEW

C Mineralization in Sludge or Fly Ash-LSR Amended Soil

Sludge application increases microbial numbers and enzyme activities in soil because there are readily available forms of organic C in sludge (Coppola, 1983; Hattori, 1988; Varanka et al., 1976). For example, Hattori and Mukai (1986) determined that the protein content of 6 sludges ranged from 25 to 60 % of the total organic matter. As soil microorganisms consume readily available substrate, C mineralization rates decline (Hsieh et al., 1981a). Incremental decreases in C mineralization rates with time indicate that soil microorganisms utilize organic sludge compounds according to their degradability. In other words, resistant organic C is decomposed at slower rates than labile compounds. The mineralization rates in sludge-amended soil drop to levels of non-amended soil within 4 to 6 weeks, and stay constant independent of time (Hattori, 1988; Hsieh et al., 1981a; Pichtel, 1990).

Although patterns of C mineralization in sludge-treated soils are similar, total C mineralization varies greatly. The factors that influence total C mineralization are: 1. Type of sludge treatment; 2. Sludge organic C content and composition; 3. Soil texture; 4. Soil pH; 5. Soil water potential; 6. Temperature; 7. Sludge application rate.

Carbon dioxide (CO_2) evolution measurements from sludge treated soil under controlled conditions are a standard technique used to determine total C mineralization. Some studies trap CO_2 evolved from amended soil in an alkaline solution such as sodium hydroxide (NaOH) and measure trapped CO_2 by titration. Other studies measure CO_2 evolution directly by gas chromatography.

Independent of the technique employed, the duration of incubation in these studies varies greatly. Much of the CO_2 evolution occurs during the first 4 weeks (Miller, 1974). Terry et al. (1979), for example reported that about 90 % of the total CO_2 produced from sludge in 19 weeks evolved in the first 4 weeks of an incubation. To remove the time effect on C mineralization, only CO_2 evolved in the first 4 weeks of incubations was plotted in Figures 1 to 11 for this review of the literature.

Wastewater undergoes primary treatment which removes settleable solids, and secondary treatment which removes fine suspended solids (EPA, 1984). Volatile sludge solids are reduced by stabilization processes thereafter. Major stabilization methods include anaerobic digestion, aerobic digestion, and composting. Anaerobic digestion, the most common sludge treatment, retains the sludge anaerobically at 20 ^{0}C and converts 40 to 50 % of the volatile solids in sludge to methane and carbon dioxide (Demuyncle et al., 1985). Aerobic digestion, the mechanical mixing of sludge with air or oxygen to maintain aerobic conditions, converts 40 to 50 % of the volatile solids in sludge primarily into carbon dioxide (EPA, 1984). The composting process commonly mixes sludge with a bulking agent such as wood chips, bark, or straw, which decomposes aerobically for several weeks. The objective of this process is to decompose organic matter in sludge to stable, humus-like substances, so that the compost can be marketed as a slow release fertilizer.

A survey of 150 sludge treatment plants in the U.S. showed that the organic C contents of aerobically and anaerobically digested sludges are similar (Sommers, 1977). Most sludges contain approximately 20 to 30 % organic C, independent of sludge type. Higgins et al. (1982) treated a raw sludge by different sludge stabilization methods. They determined that cellulose and lignin make up a greater percentage of total solids in anaerobic sludge than in aerobic sludge (Table 1).

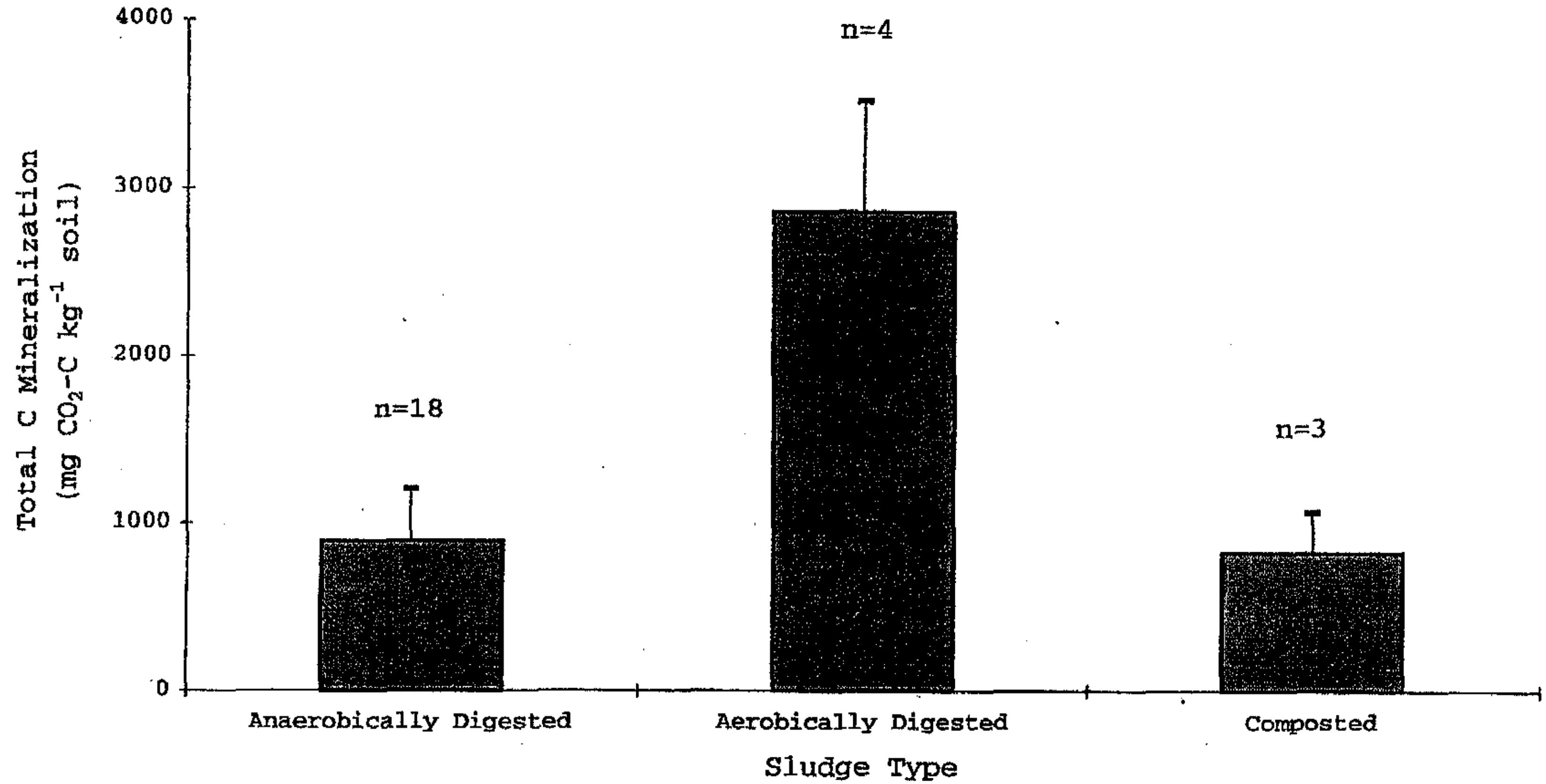
Lignin resists degradation (Bartlett and Norman, 1938) and slows the decomposition rate of cellulose with which it is intermixed (Fuller and Norman, 1943). This may explain why total C mineralization is higher in aerobic sludges than anaerobic sludges for a given time (Figure 1). Hsieh et al. (1981a) found that anaerobically digested sludge had 8 times

Table 1. Effect of stabilization method on final organic composition of a sludge[†].

Constituent	% of Total Solids		
	Aerobically Digested	Anaerobically Digested	Composted
Total Grease (Ether Soluble)	4.9	4.6	5.4
Amino Acids	14.5	15.1	13.6
Cellulose	12.9	15.7	7.2
Lignin	9.5	12.1	10.5

[†] Data from Higgins et al., 1982.

Figure 1. The relationship between total C mineralization and sludge type[†].



[†] Data from Agbim et al. (1977); Boyle and Paul (1989); Hattori and Mukai (1986); Hsieh et al. (1981a); Lerch et al. (1992); Neilson and Pepper (1990); Pichtel and Hayes (1990); Terry et al. (1979); Tester et al. (1977); Wiseman and Zibilske (1988). Error bar and n represent 1 standard deviation and number of observations, respectively.

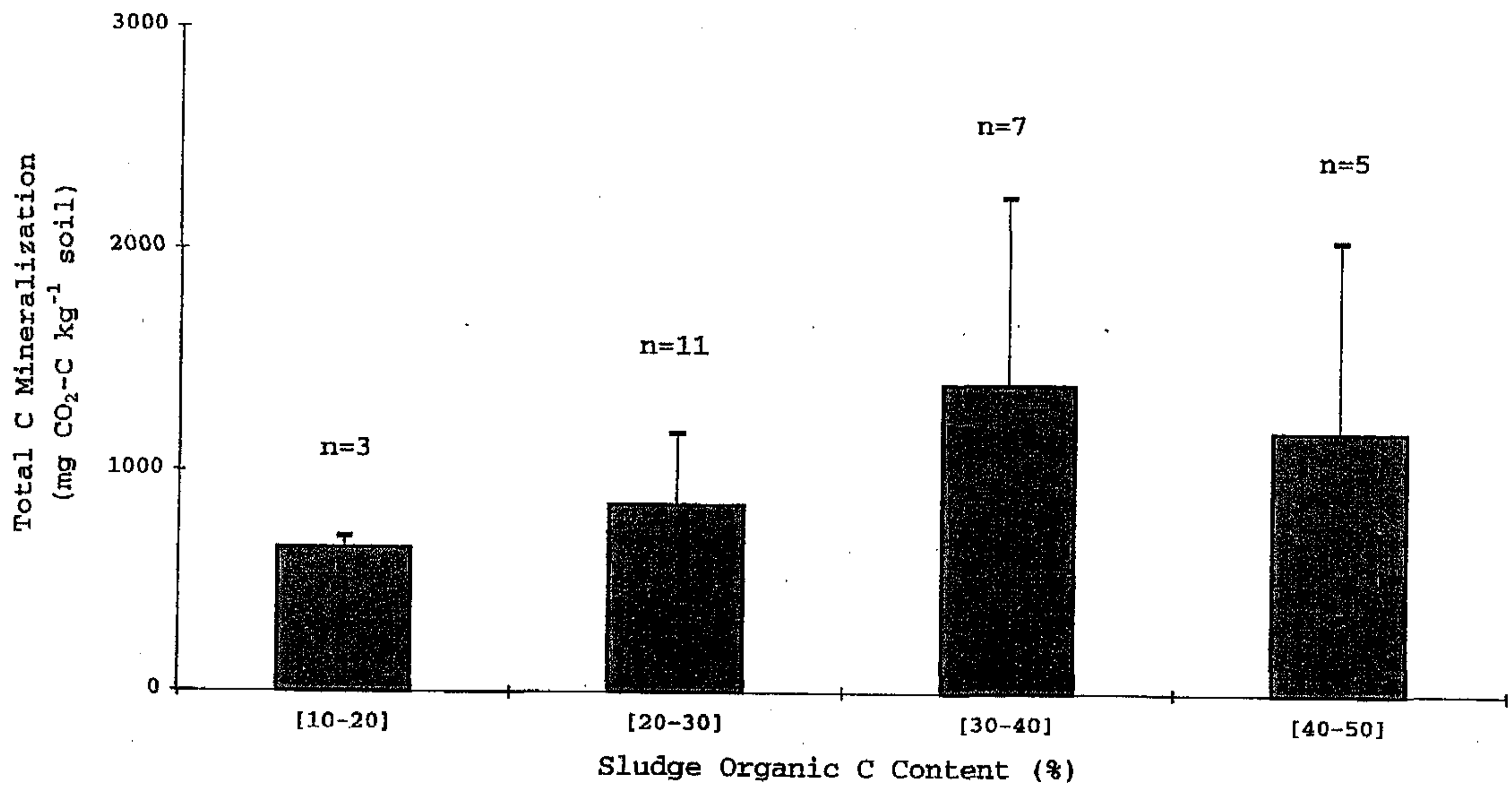
less total C mineralized than aerobically digested sludge because it had 1.5 times more lignin.

The ratio of bulking agent to sludge in compost is 3:1 in most cases (EPA, 1984). Composting increases the organic C content of the final product since bulking agents contain more organic C than sludge. For example, carbon contents of bark and wood are 32 and 39 % of the dry matter, respectively (Agbim et al., 1977). Much of the organic C in a compost is resistant to microbial degradation since the bulking agent is poorly degradable. For example, the lignin content of wood typically ranges from 25 to 50 % of the dry matter (Lawson and Still, 1957). This results in less total C mineralization in composted sludges than in aerobic sludges (Figure 1).

Total C mineralization tends to increase as sludge organic C content increases (Figure 2). However, sludge organic C composition is more important than sludge organic C content in determining total C mineralization (Boyle and Paul, 1989; Hattori and Mukai, 1986; Hsieh et al., 1981a; Lerch et al., 1992). As labile organic matter in sludge increases, total C mineralization in sludge treated soil increases.

Simple sugars and free amino acids are the most available constituents of sludge (Rebhun and Manka, 1971). These constituents are mineralized early in the incubation

Figure 2. The relationship between total C mineralization and sludge organic C content†.



† Data from Agbim et al. (1977); Boyle and Paul (1989); Hattori and Mukai (1986); Hsieh et al. (1981a); Lerch et al. (1992); Neilson and Pepper (1990); Pichtel and Hayes (1990); Terry et al. (1979); Tester et al. (1977); Wiseman and Zibilske (1988). Error bar and n represent 1 standard deviation and number of observations, respectively.

and contribute much to respiration (Schaumberg et al., 1980). Hattori and Mukai (1986) determined that there was a relationship between total C mineralization and water-soluble sugar contents in 5 sludges (Figure 3).

Lerch et al. (1992) determined that there was high correlation between total C mineralization and extractable proteins in 7 sludges (Figure 4). The correlation coefficient was approximately 0.95 for both H_2O and 1 M NaOH soluble proteins. Protease activity of sludge amended soil was also positively related to total C mineralization since most soil microorganisms are proteolytic (Hattori, 1988).

Total C mineralization is inversely related to sludge lignin content (Hattori and Mukai, 1986). Since lignin in sludge contributes to soil organic matter (Epstein et al., 1976; Gupta et al., 1977; Hohla et al., 1978; Kladivko and Nelson, 1979; Lunt, 1959), one can assume that as sludge contribution to soil organic matter increases, total C mineralization from the sludge amended soil should decrease. Data from C mineralization studies show this. Anaerobically digested sludges (Gupta et al., 1977; Kladivko and Nelson, 1979) increase soil organic matter more than aerobically digested sludges (Epstein et al., 1976; Lunt, 1959) anaerobically digested sludges also have lower total C mineralization than aerobically digested sludges (Figure 1).

Figure 3. The relationship between total C mineralization and water soluble sugar contents in 5 sludges (Hattori and Mukai, 1986).

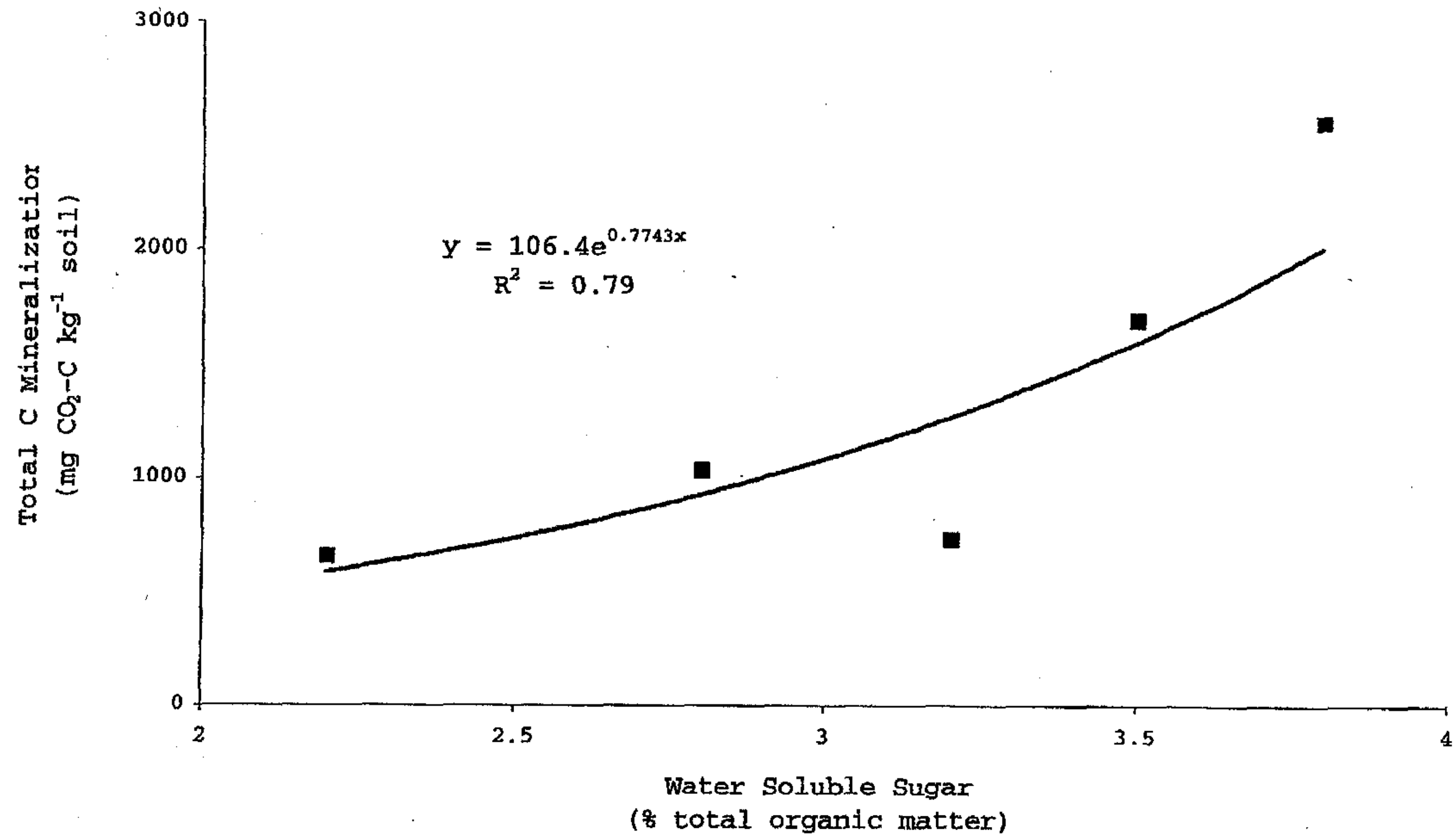
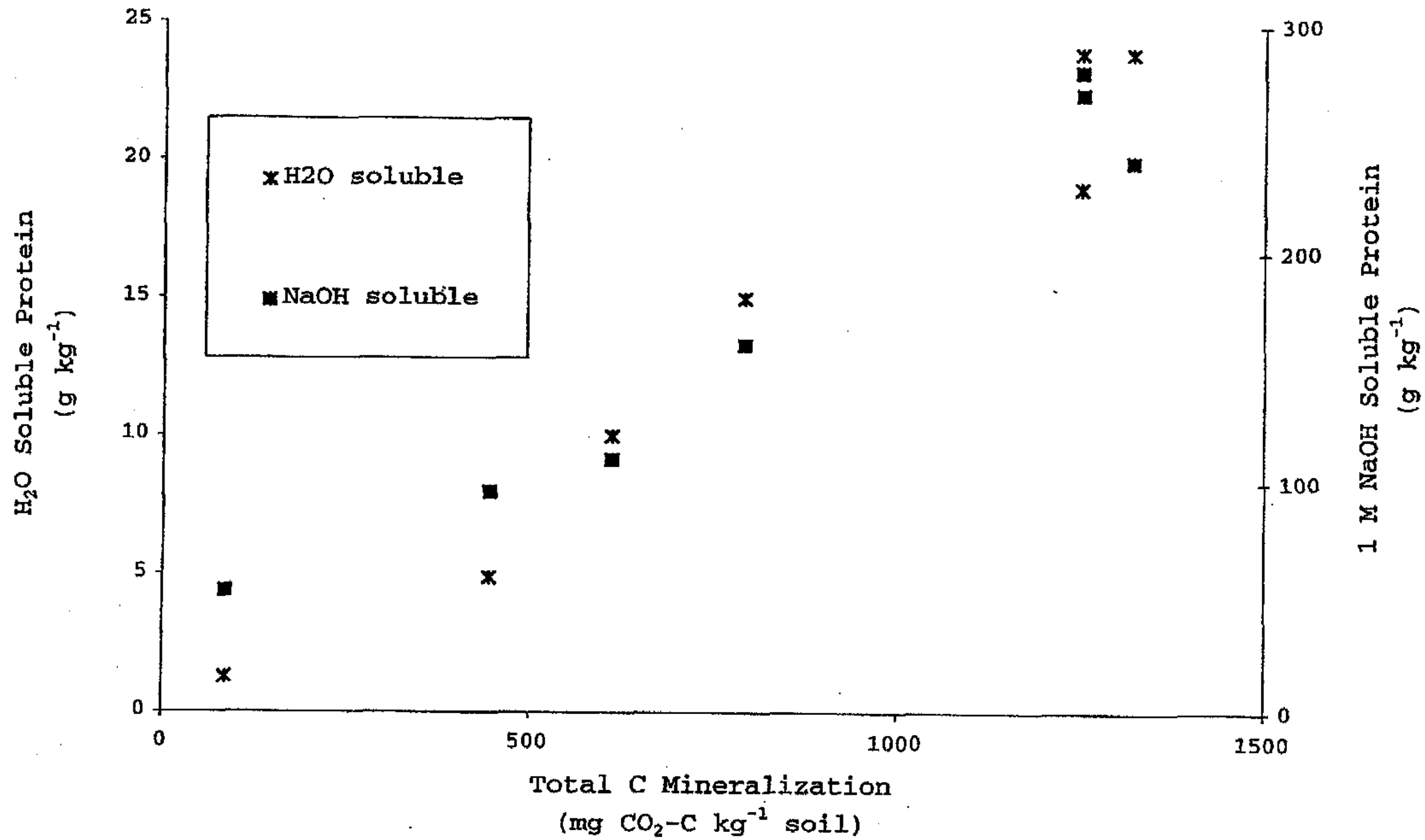


Figure 4. The relationship between total C mineralization and protein contents in 7 sludges (Lerch et al., 1992).

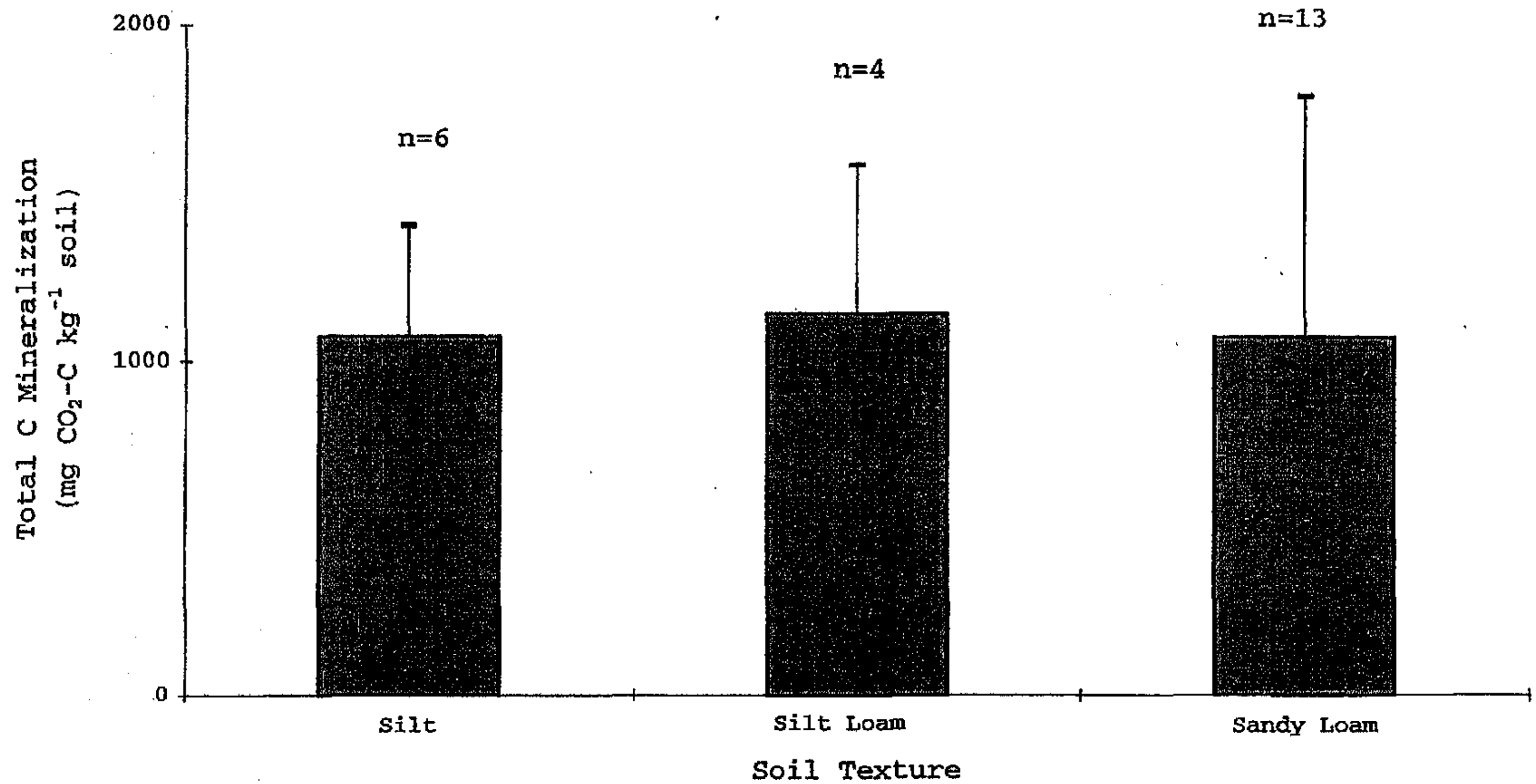


Soil texture seems to have little effect on total C mineralization in sludge treated soils (Figure 5). However, the variability of soil properties in various studies is too great to demonstrate this conclusively. Soil texture effects on C mineralization can be best explained by looking at the results from studies that used soils differing greatly in soil texture, but not in other soil properties.

Tester et al. (1977) determined that when amended with sludge, a silt loam soil mineralized approximately 15 % more C than a sand and a sandy loam soil. Terry et al. (1979) found that total C mineralization values in 3 different soils amended with a sludge were similar after a month. But after a year, total C mineralization values were approximately 20 % higher in silt loam soil than in sandy loam soil. Relatively high total C mineralization in silt loam soils may be a function of water regime and aeration in such soils. These soils tend to have more available water for a given moisture content.

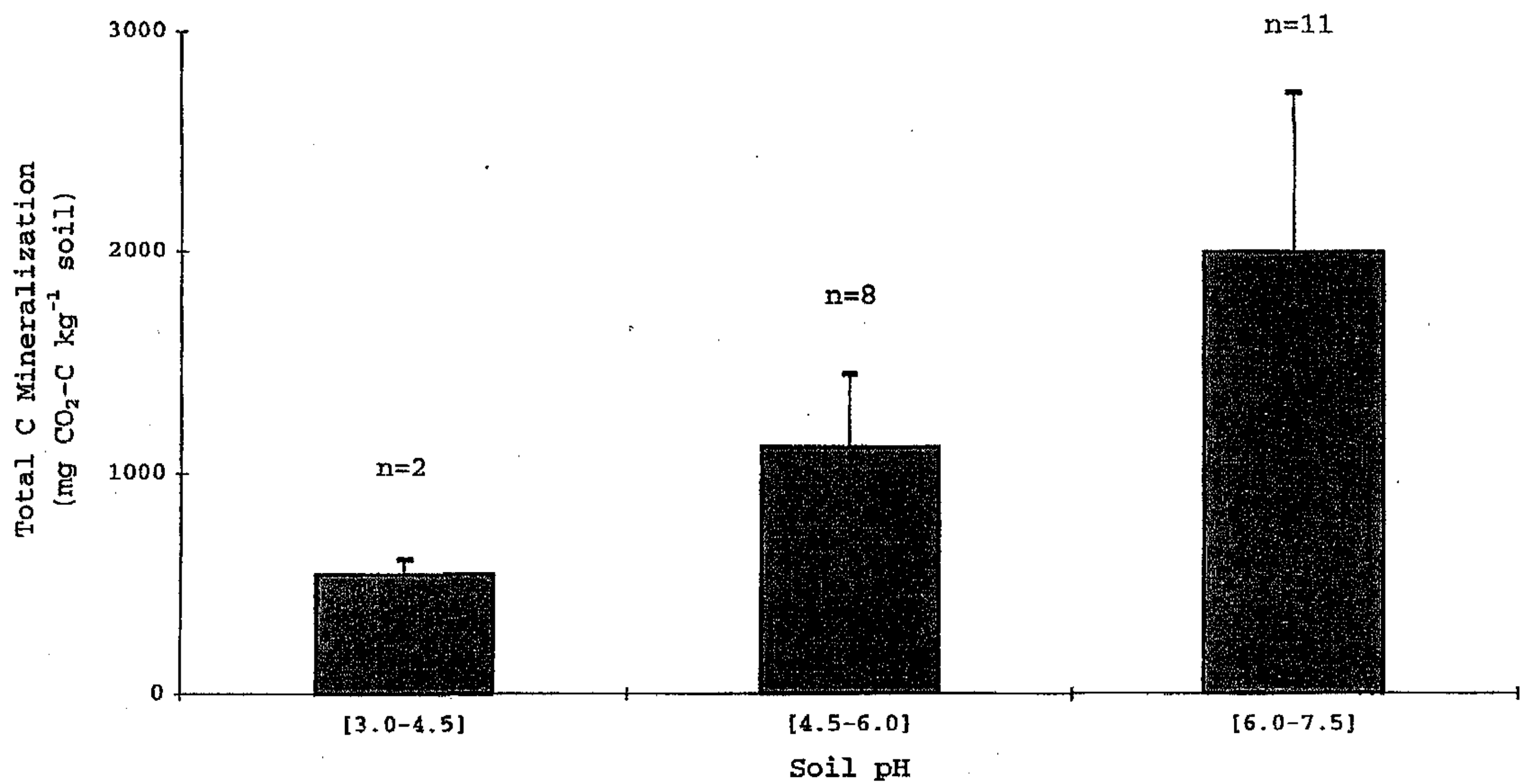
Total C mineralization in sludge amended soil tends to be highest when soil pH is neutral (Figure 6). As soil pH decreases, total C mineralization decreases. The optimum pH for most heterotrophic microorganisms ranges from 6.5 to 7.5. Bacteria, major mineralizers in soil, are more susceptible to low pH than fungi.

Figure 5. The relationship between total C mineralization and soil texture†.



† Data from Agbim et al. (1977); Boyle and Paul (1989); Hattori and Mukai (1986); Hsieh et al. (1981a); Lerch et al. (1992); Neilson and Pepper (1990); Pichtel and Hayes (1990); Terry et al. (1979); Tester et al. (1977); Wiseman and Zibilske (1988). Error bar and n represent 1 standard deviation and number of observations, respectively.

Figure 6. The relationship between total C mineralization and soil pH[†].



[†] Data from Agbim et al. (1977); Boyle and Paul (1989); Hattori and Mukai (1986); Hsieh et al. (1981a); Lerch et al. (1992); Neilson and Pepper (1990); Pichtel and Hayes (1990); Terry et al. (1979); Tester et al. (1977); Wiseman and Zibilske (1988). Error bar and n represent 1 standard deviation and number of observations, respectively.

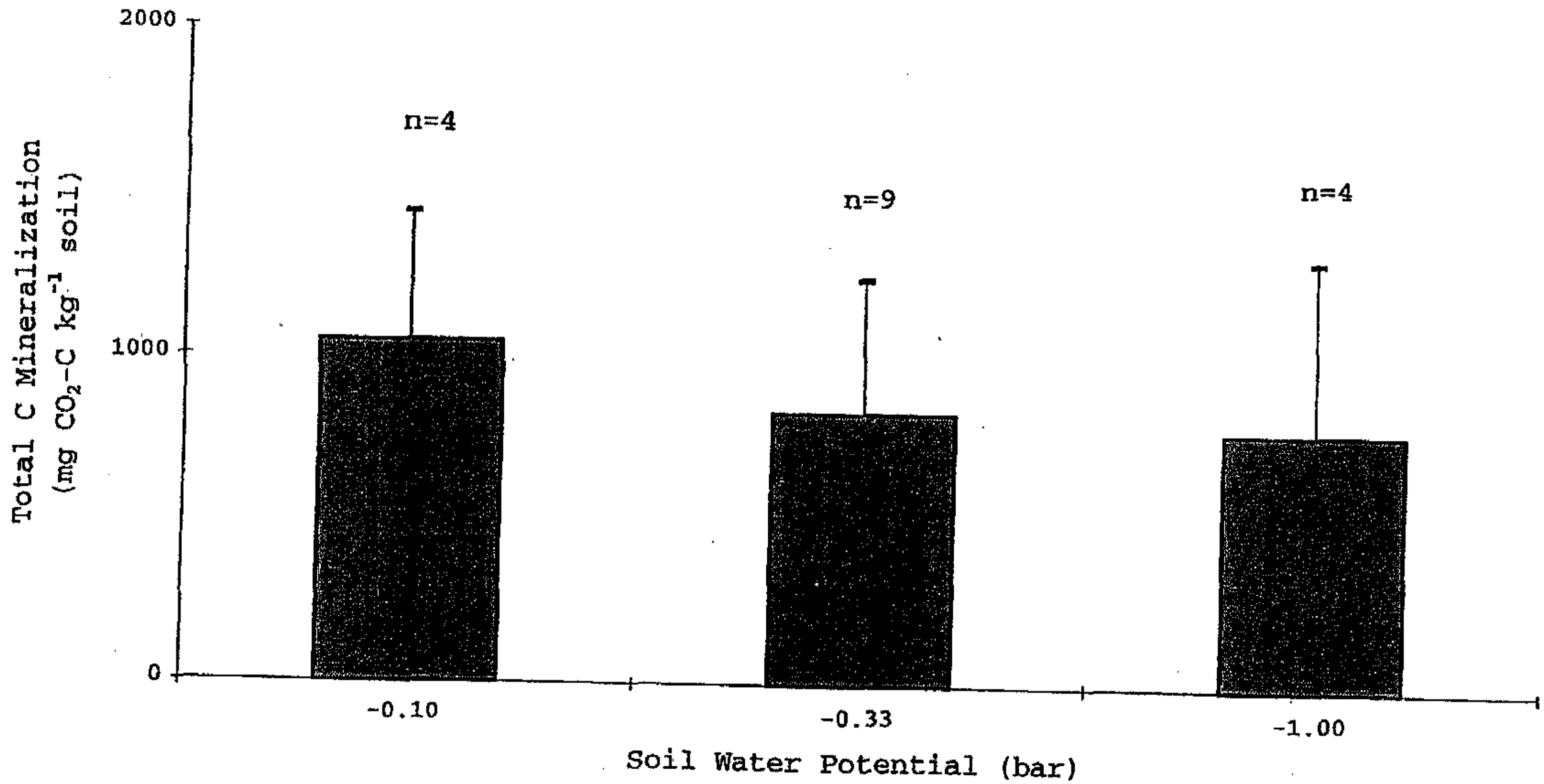
The pH of sludge-amended soil is controlled by the initial soil and sludge pH. When an acid soil is treated with acid sludges, total C mineralization values are near 0 (Hattori and Mukai, 1986). Alkaline sludges neutralize acidity arising from the initial soil pH and nitrification during incubation (Hattori and Mukai, 1986; Terry et al., 1979). Alkaline sludge amendments to acid soils usually increase total C mineralization by creating a favorable pH for microbial growth.

Soil water affects important factors for microbial growth such as the nature and amount of soluble materials and aeration. When soil water potential is too low, the amount and movement of dissolved compounds in soil are insufficient. When soil is saturated, aeration becomes a problem.

Total C mineralization in sludge-amended soil varies little when soil water potential ranges from -0.01 bar to -1.00 bar (Figure 7). Hsieh et al. (1981a) reported that total C mineralization values were similar when soil water potential ranged from -0.06 to -0.33 bar.

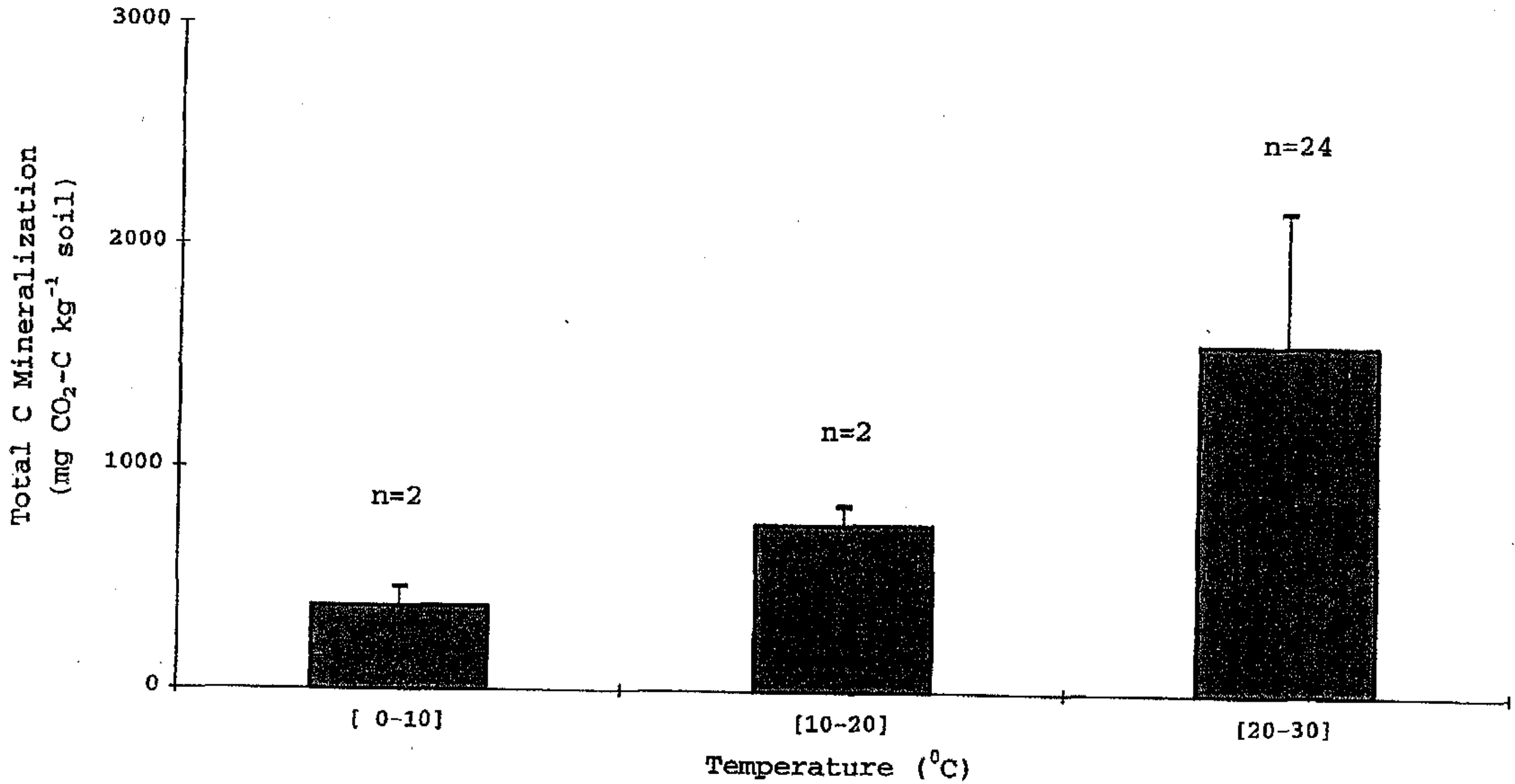
Total C mineralization increases as temperature increases (Figure 8). The temperature quotient Q_{10} (the ratio of reaction rates at two temperatures separated by 10 ^{0}C) for sludge C mineralization is approximately 2.0 at temperatures ranging from 15 ^{0}C to 30 ^{0}C (Hsieh et al.,

Figure 7. The relationship between total C mineralization and soil water potential†.



† Data from Agbim et al. (1977); Boyle and Paul (1989); Hattori and Mukai (1986); Hsieh et al. (1981a); Lerch et al. (1992); Neilson and Pepper (1990); Pichtel and Hayes (1990); Terry et al. (1979); Tester et al. (1977); Wiseman and Zibilske (1988). Error bar and n represent 1 standard deviation and number of observations, respectively.

Figure 8. The relationship between total C mineralization and temperature[†].



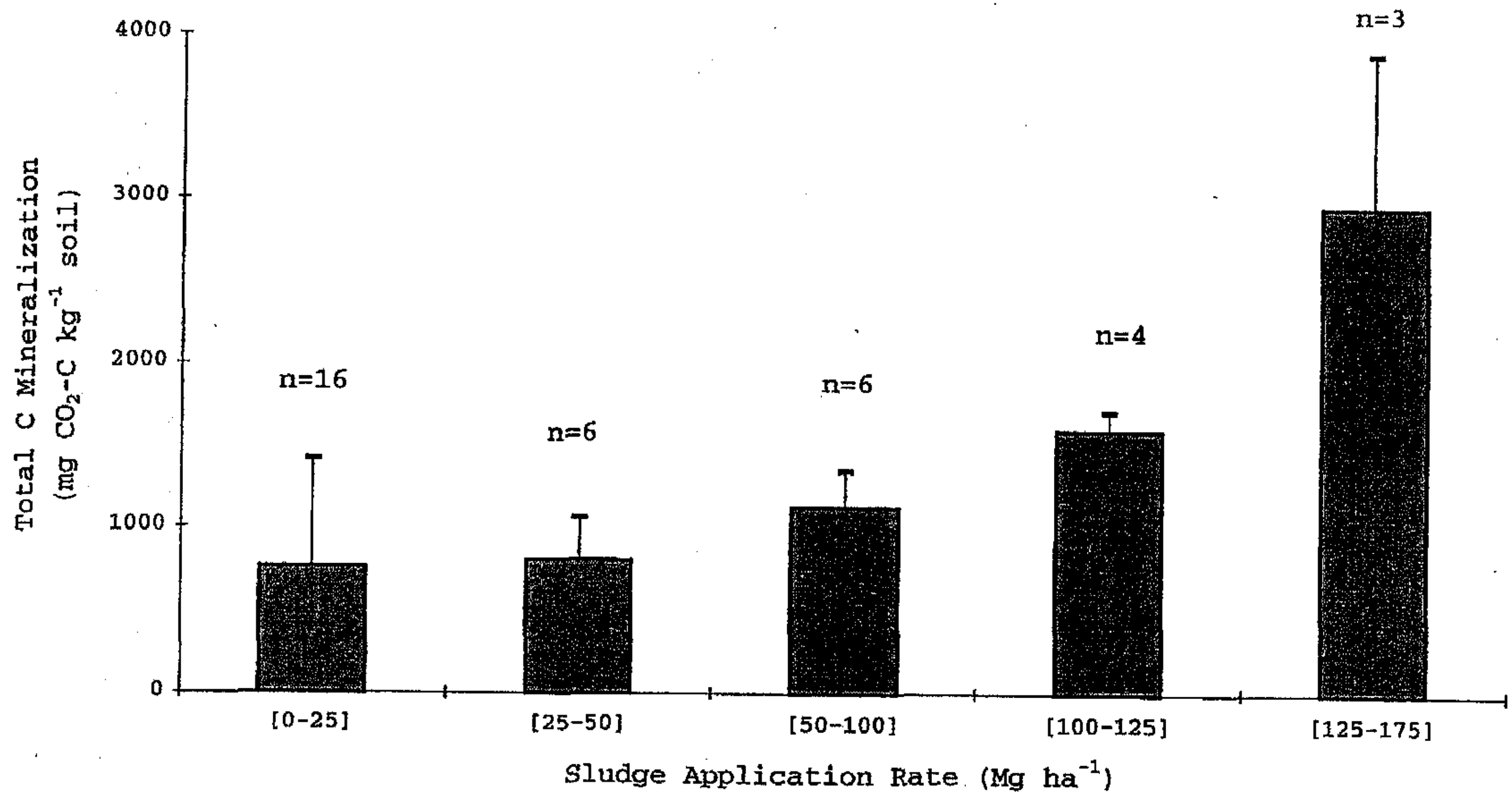
[†] Data from Agbim et al. (1977); Boyle and Paul (1989); Hattori and Mukai (1986); Hsieh et al. (1981a); Lerch et al. (1992); Neilson and Pepper (1990); Pichtel and Hayes (1990); Terry et al. (1979); Tester et al. (1977); Wiseman and Zibilske (1988). Error bar and n represent 1 standard deviation and number of observations, respectively.

1981a; Terry et al., 1979). The effects of temperature are most apparent in the first month of incubation (Miller, 1974; Terry et al., 1979). This suggests that temperature has more effect on labile sludge organic C than on resistant sludge organic C.

As sludge application rate increases, total C mineralization tends to increase due to the additional C substrate (Figure 9). The highest rate of increase in total C mineralization as more sludge is added occurs in aerobically digested sludge-amended soils (Figure 10). The rate of increase is less in anaerobically digested and composted sludges than aerobically digested sludges (Figures 11 and 12). This might be due to higher lignin content in anaerobically digested and composted sludges than aerobically digested sludges (Hattori, 1988; Hsieh et al., 1981a).

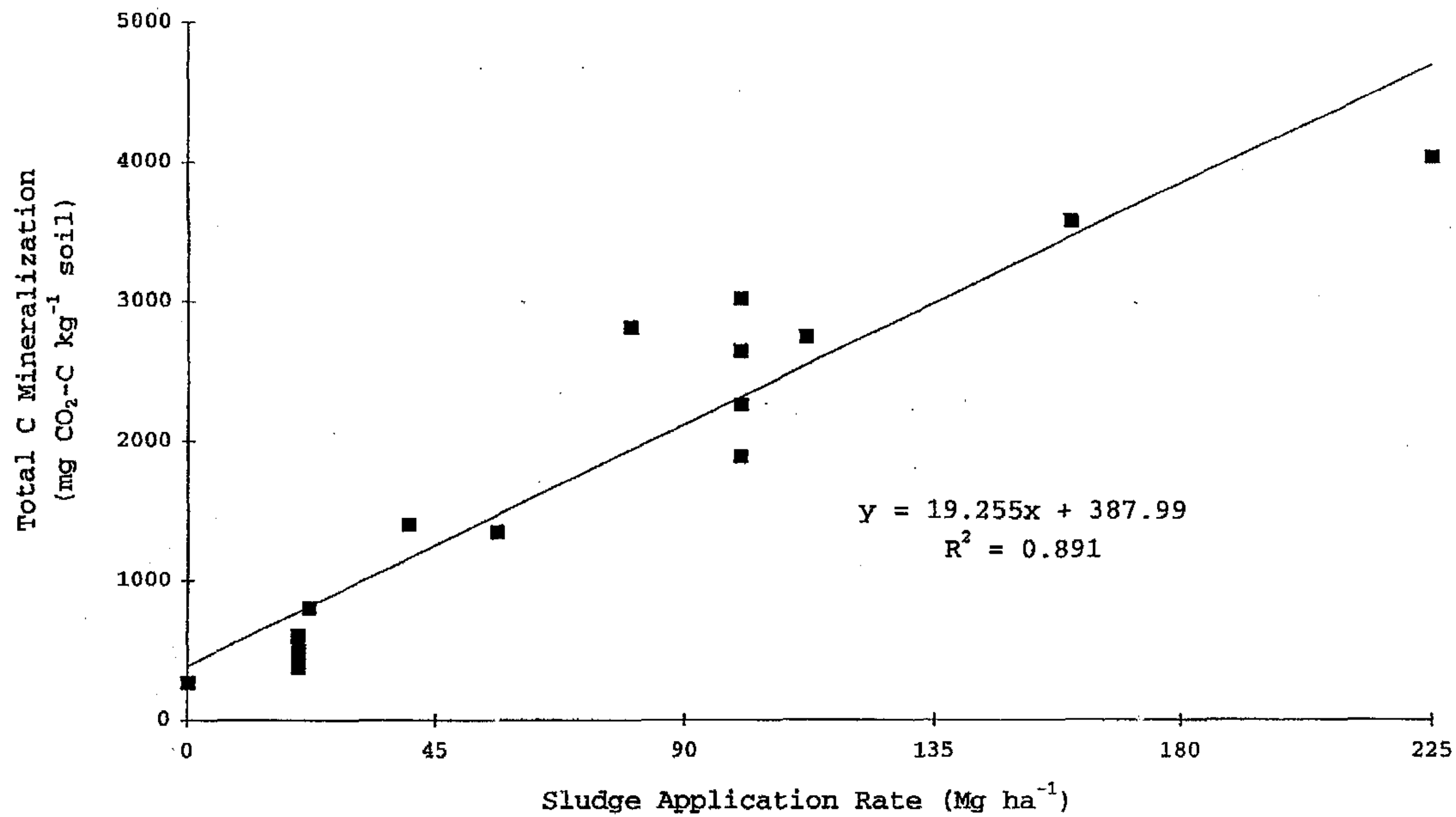
Although total C mineralization increases, as more sludge is added, the % organic C mineralized from sludge decreases regardless of sludge type (Figure 13). The % C mineralization is higher in aerobically digested sludge-amended soils than in anaerobically digested sludge-amended soils, probably because aerobically digested sludges have more labile substrate C and less lignin (Hattori, 1988; Hsieh et al., 1981a).

Figure 9. The relationship between total C mineralization and sludge application rate†.



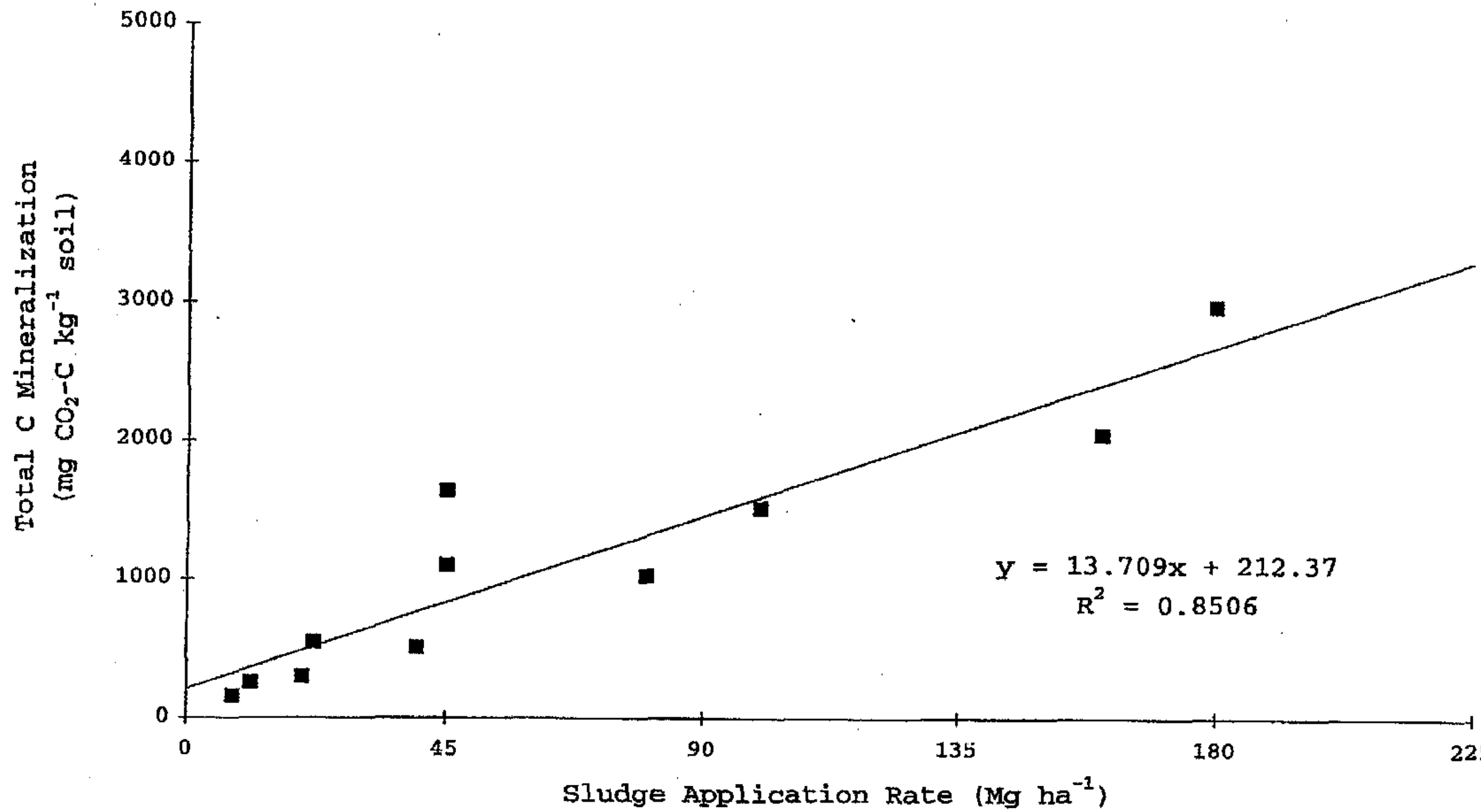
† Data from Agbim et al. (1977); Boyle and Paul (1989); Hattori and Mukai (1986); Hsieh et al. (1981a); Lerch et al. (1992); Neilson and Pepper (1990); Pichtel and Hayes (1990); Terry et al. (1979); Tester et al. (1977); Wiseman and Zibilske (1988). Error bar and n represent 1 standard deviation and number of observations, respectively.

Figure 10. Relationship between total C mineralization and sludge application rate in aerobically digested sludge[†].



[†] Data from Agbim et al. (1977); Hattori (1988); Hsieh et al. (1981a)

Figure 11. Relationship between total C mineralization and sludge application rate in anaerobically digested sludge[†].



[†] Data from Boyle and Paul (1989); Hattori (1988); Hsieh et al. (1981a); Terry et al. (1979).

Figure 12. Relationship between total C mineralization and sludge application rate in composted sludge[†].

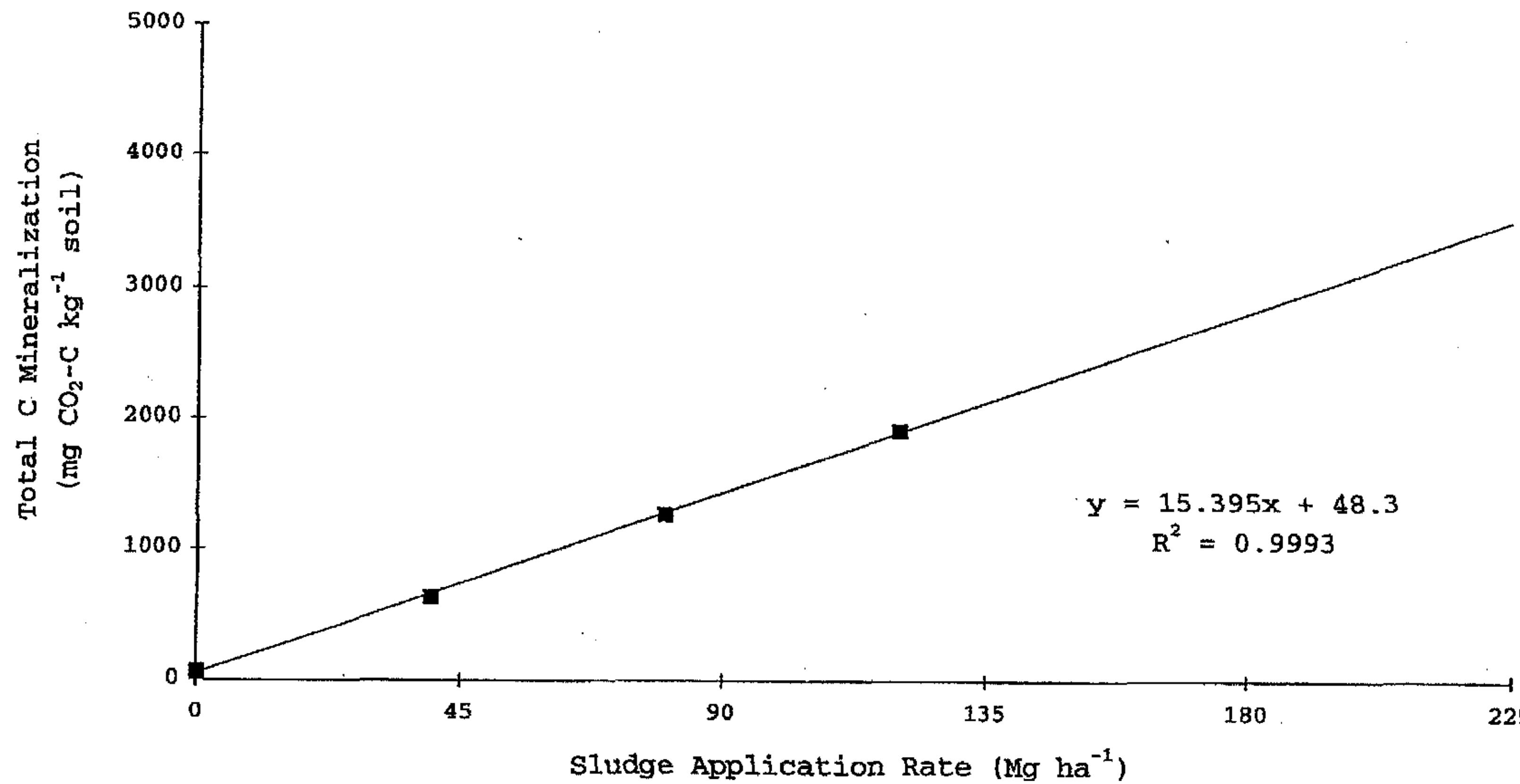
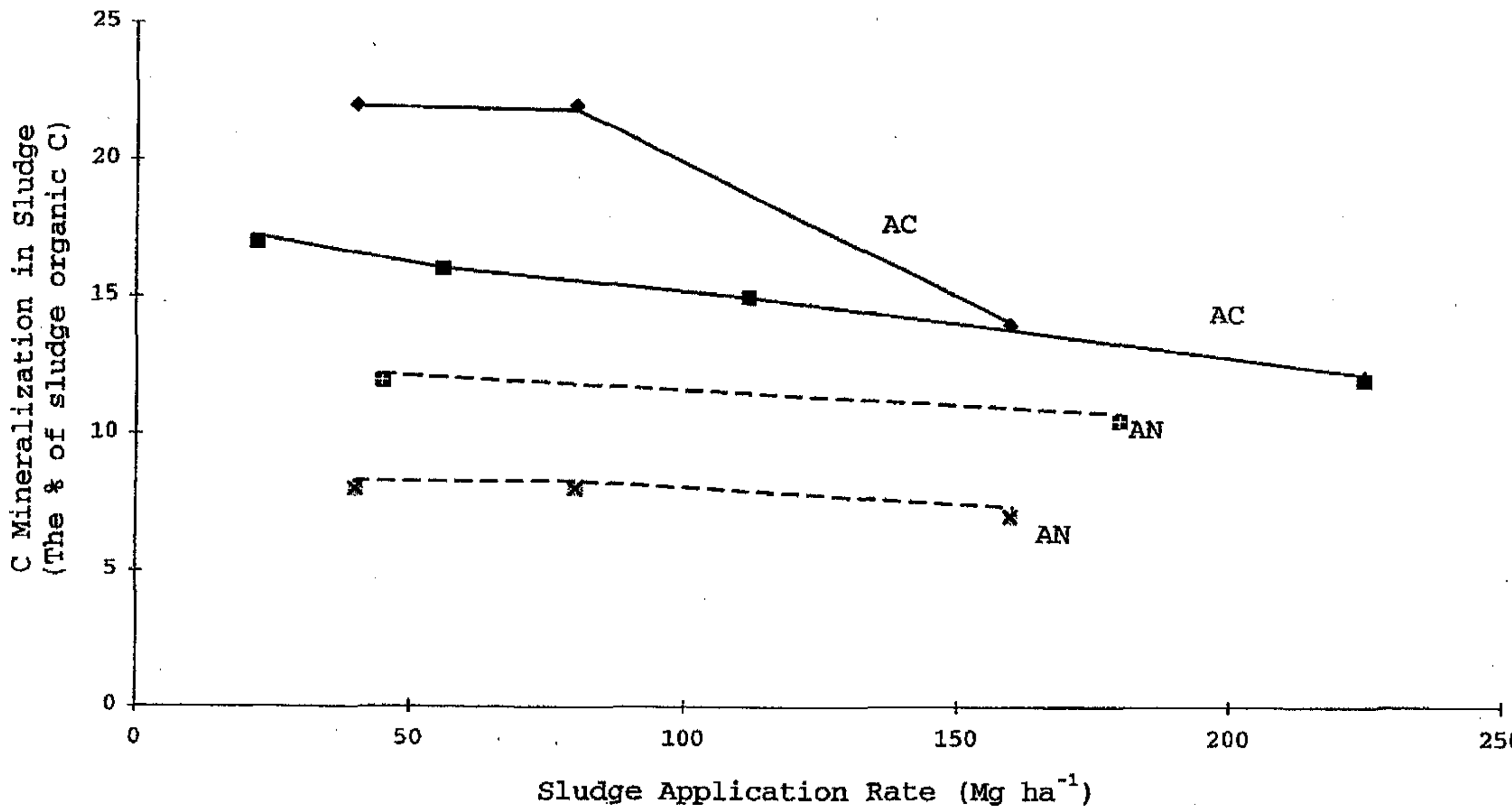


Figure 13. Relationship between the % C mineralization in sludge and sludge application rate†.



† Data from Agbim et al. (1977); Boyle and Paul (1989); Hsieh et al. (1981a); Tester et al. (1977). AC: Aerobically Digested Sludge; AN: Anaerobically Digested Sludge.

The decrease in % C mineralization as more sludge is added isn't due to changes in soil physical properties. Studies show that sludge amendment, especially at high rates, improved soil physical properties for microbial growth. The improvements are better soil aggregation, water retention (Epstein, 1975), and porosity (Gupta et al., 1977; Pagliai et al., 1981). Sludge addition increased total porosity by 13 % (Gupta et al., 1977) and the number of pores sized from 0.5 µm to 500 µm (Pagliai et al., 1981).

Sludge applied at high rates, however, may change soil chemical properties and inhibit soil microbes. Sludge salinity increases when sludge is treated with lime [$Ca(OH)_2$], alum [$Al_2(SO_4)_3$], or ferric chloride ($FeCl_3$) to reduce pathogen and odor problems. Sludges treated with these chemicals increase the soil salinity when they are applied at high rates (Agbim et al., 1977; Tester and Parr, 1983). Electrical conductivity values in sludge-amended soils are near toxic levels for C mineralization (Johnson and Guenzi, 1963; Nelson et al., 1996).

Although many studies have focused on C mineralization in sludge treated soils, only a few studies have assessed C mineralization in fly ash-LSR treated soils (Arthur et al., 1984; Pichtel, 1990). Even fewer studies have examined fly ash-LSR and sludge treated soils (Pichtel, 1990).

A 100 Mg ha^{-1} sludge amendment increased total C mineralization in soils amended with fly ash-LSR (Pichtel, 1990). The increase in total C mineralization by sludge was significant in soils treated with fly ash-LSR at rates up to 200 Mg ha^{-1}, but not at 400 Mg ha^{-1}.

Fly ash-LSR is unlikely to increase C mineralization in soil by itself since it doesn't contain organic C. Arthur et al. (1984) and Pichtel, (1990) determined that fly ash-LSR at a 100 Mg ha^{-1} application rate didn't affect respiration, but it did greatly reduce respiration at rates higher than 200 Mg ha^{-1}. These authors concluded that the decrease in respiration was due to toxicity caused by heavy metals in fly ash-LSR.

Limited information exists to clearly identify the role of fly ash-LSR amendment in decreasing total C mineralization in soils. The heavy metals in fly ash-LSR used in the Arthur et al. (1984) and Pichtel (1990) C mineralization studies were chromium (Cr), cobalt (Co), copper (Cu), lead (Pb), nickel (Ni), and zinc (Zn). Concentrations of these heavy metals ranged from 100 to 1000 ppm. Arthur et al. (1984) implied that heavy metals in fly ash-LSR were at least 10 times greater than those that could cause inhibition of soil microorganisms. However, they ignored the dilution of heavy metals in fly ash-LSR caused by mixing with soil. For example, a 200 Mg ha^{-1} fly ash-LSR amendment to soil dilutes heavy metal concentrations in fly ash-LSR approximately 10 fold.

The availability of heavy metals to soil microorganisms depends on soil pH. Many heavy metals are most available under acidic conditions. At higher pH, most heavy metals precipitate or bind to clay and organic material in soil. In Pichtel's (1990) work, however, pH levels in fly ash-LSR treated soils were either neutral or alkaline. This suggests that inhibition of C mineralization wasn't due to heavy metals in fly ash-LSR since most heavy metals are much less available in alkaline soils.

Fly ash-LSR pH can vary from 4.5 to 12, depending on the S content of the coal (Adriano, et al., 1980). Eastern coals have a higher S content and produce acidic fly ash-LSR while western coals have lower S content and produce alkaline fly ash-LSR. Weathering processes neutralize alkalinity in fly ash-LSR by carbonation, a process in which CO_2 from the atmosphere combines with Ca^{+2} and Mg^{+2} oxides, forming stable Ca and Mg carbonates.

Alkaline fly ash-LSR (usually unweathered) has been used to neutralize acid soils and provide a better pH for plant growth. In some cases, however, fly ash-LSR amendment increased soil pH well above neutral levels and caused toxicity to plants grown in the amended soil (Adriano et al., 1982; Elseewi et al., 1980; Petruzzelli et al., 1987).

In one study, an alkaline ash was added to both acidic and neutral soils (Adriano et al., 1982). The ash amendment increased soil pH to 7.0 and 9.0 in the acidic and neutral soils, respectively. The ash amendment increased yield of sudan grass (S. vulgaris) in acidic soil while it decreased yield in neutral soil.

Soil buffering capacity can be more important in determining the pH effects of ash addition than initial soil pH (Elseewi et al., 1980). Petruzzelli et al. (1987) found that when an ash was added to 2 acidic soils, the pH increased to 9.0 in poorly buffered soil, and there was little increase in well buffered soil.

Unweathered ashes are high in soluble salts. The dominant cation in fly ash-LSR is Ca^{+2} (Sakata, 1987; Spencer and Drake, 1987; Theis et al., 1978). When such ashes are applied to soil, EC can exceed 13 dSm^{-1} (Adriano et al., 1978; Mulford and Martens, 1971). These levels are above the EC considered to cause adverse effects for most plants (EC > $4dSm^{-1}$). Weathering decreases salinity in fly ash-LSR by carbonation processes which form carbonates of Ca^{+2} and Mg^{+2} cations.

Multiple factors may control toxicity to C mineralization in soil treated with fly ash-LSR. For example, the fly ash used in Arthur et al.'s (1984) C mineralization study had a high B concentration (300 ppm) and a pH of 5.0. Soil amendment with this fly ash-LSR may have caused toxicity due to both high B and low pH. Acidic conditions may mobilize the heavy metals contained in fly ash-LSR. Consequently, synergistic effects of these factors may increase the toxicity. Boron concentrations in these ash materials range from 10 to 5,000 ppm, but are mostly in the range of 250 to 600 ppm (Hatcher and Wilcox, 1950). The availability of B is much greater in acidic unweathered fly ash-LSR than from weathered fly ash-LSR (Mulford and Martens, 1971).

N mineralization in Sludge or Fly Ash-LSR Treated Soil

Nitrogen (N) mineralization in sludge treated soils has been often investigated (Epstein et al., 1978; Fine et al., 1989; Garau et al., 1986; Hsieh et al., 1981b; Magdoff and Chromec, 1977; Oba and Nguyen, 1982; Parker and Sommers, 1983; Ryan et al., 1973; Terry et al., 1981; Yoneyama and Yoshida, 1978). Sludge N mineralization is important since the final product of this process, ammonium (NH_4^+), serves as a plant nutrient and a substrate for nitrification. Nitrification produces nitrite (NO_2^-) and nitrate (NO_3^-) that are subject to leaching to groundwater and denitrification. Another fate of nitrogen is through ammonia (NH_3) volatilization when sludge has high NH_4^+ concentrations and pH is alkaline in the amended soil.

Whenever N mineralization occurs, assimilation of inorganic N into microbial cells (immobilization) runs counter to it. Therefore, most studies measure net N mineralization, that is, the difference between N mineralized and N immobilized. Since the N cycle is composed of interlinked processes the type of measurement and experimental protocol become important when assessing net N mineralization. Measurements of net changes in the inorganic N pool have been employed to assess net sludge N mineralization. However, a change in inorganic N status during incubation doesn't account for all N fates, namely ammonia (NH_3) volatilization and denitrification. For example, Hsieh et al. (1981b) determined that the net gain in inorganic N during an 8 week incubation accounted for just 25 % to 50 % of the net organic N loss. Thus, the reliability of net changes in inorganic N as a measure of net N mineralization depend on the rate and magnitude of N fates.

Few studies have assessed volatilization of NH_3 from sludge-treated soil (Fine et al., 1989; Ryan et al., 1973). These studies showed that NH_3 volatilization is a major fate of mineral N when sludge has NH_4^+-N contents of 7400 ppm or higher. Fine et al. (1989) determined that approximately 28, 35, and 48 % of the sludge organic N was ammonified and lost via NH_3 volatilization at 20, 100, and 200 Mg ha^{-1} sludge application rates, respectively.

As sludge rates increase, the NH_4^+-N content of treatments and the potential for NH_3 volatilization increase. Ryan et al. (1973) observed approximately 5 to 8 times higher NH_3 volatilization than Fine et al. (1989), but their sludge had an extremely high NH_4^+-N content (47700 ppm). Most studies use sludges with NH_4^+-N contents ranging from 1000 to 4000 ppm. Since these sludges have a lower inorganic N content, NH_3 volatilization in soils treated with such sludges should be less than those observed by Fine et al. (1989) and Ryan et al. (1973).

Denitrification in sludge-amended soils is controlled by the availability of anaerobic conditions and substrate (i.e. NO_2^-, NO_3^-). Sludge treated soil samples are generally incubated in poylethylene bags that are permeable to air. Early in the incubation, probably within a few days, the rate of oxygen (O_2) consumption by soil microorganisms may exceed the rate of O_2 diffusion into the bags, promoting anaerobic microsites in sludge-amended soil. The NO_2^--N + NO_3^--N content of treatments is usually low initially and only increases as nitrification occurs (Hsieh et al., 1981b; Oba and Nguyen, 1982; Terry et al., 1981). Consequently, denitrification at high rates is often unlikely in most sludge-amended soils since when O_2 is rapidly consumed initially, there isn't substrate (NO_3^-–N), and when the substrate is formed later, O_2 concentration is too high.

However, denitrification in sludge amended soil could occur when there is either a high initial NO_3^-–N content or rapid O_2 consumption during the whole experiment. A heavy sludge amendment may maintain high C mineralization rates for a long time because of abundant labile C substrate. This will increase the duration of high O_2 consumption and create microsites, even during late incubation. Consequently, denitrification may occur in these treatments as NO_3^- is formed. For example, Fine et al. (1989) measured a denitrification rate of 6.5 mg N_2O-N day^{-1} in a 200 Mg ha^{-1} treatment at the fourth week of incubation. However, mineral N losses via NH_3 volatilization were 10 to 15 times higher than mineral N loss via denitrification.

Measurements of net changes in the organic N pool are a more accurate predictor of net N mineralization than changes in the inorganic N pool. This is because there are few fates of organic nitrogen except through N mineralization.

This review will address studies that measured organic N loss. It will also deal with studies that measured mineral N gain only if these studies accounted for NH_3 volatilization or when sludge NH_4^+-N content and application rates were low.

Two methods for removing mineral N from incubating samples are generally used. Some studies leach inorganic N from incubating samples repeatedly throughout incubation. Other studies periodically extract inorganic N from either a replication or a subsample of a treatment. The leachate method may underestimate net N mineralization since leaching removes some of the organic N, which could be mineralized in samples (Parker and Sommers, 1983). On the other hand, it may also overestimate net N mineralization since leaching not only removes organic N but also inorganic N which could be immobilized. Garau et al. (1986) found that the net N mineralization values were approximately 2 times greater when the leachate method was used compared to the extraction method. Because the measurements made by the leachate method are biased, this review deals with studies that used the extraction method.

Independent of the extraction technique employed, estimates of net N mineralization in sludge-treated soils show great variation This variation is a function of: 1. Sludge type; 2. Sludge organic N content and composition; 3. Sludge C:N ratio; 4. Soil texture; 5. Soil pH; 6. Soil water potential; 7. Temperature; 8. Sludge application rate.

When O_2 is supplied in excess, aerobic treatment of wastewater can biodegrade a greater range and amount of substrates in a given time than anaerobic treatment.

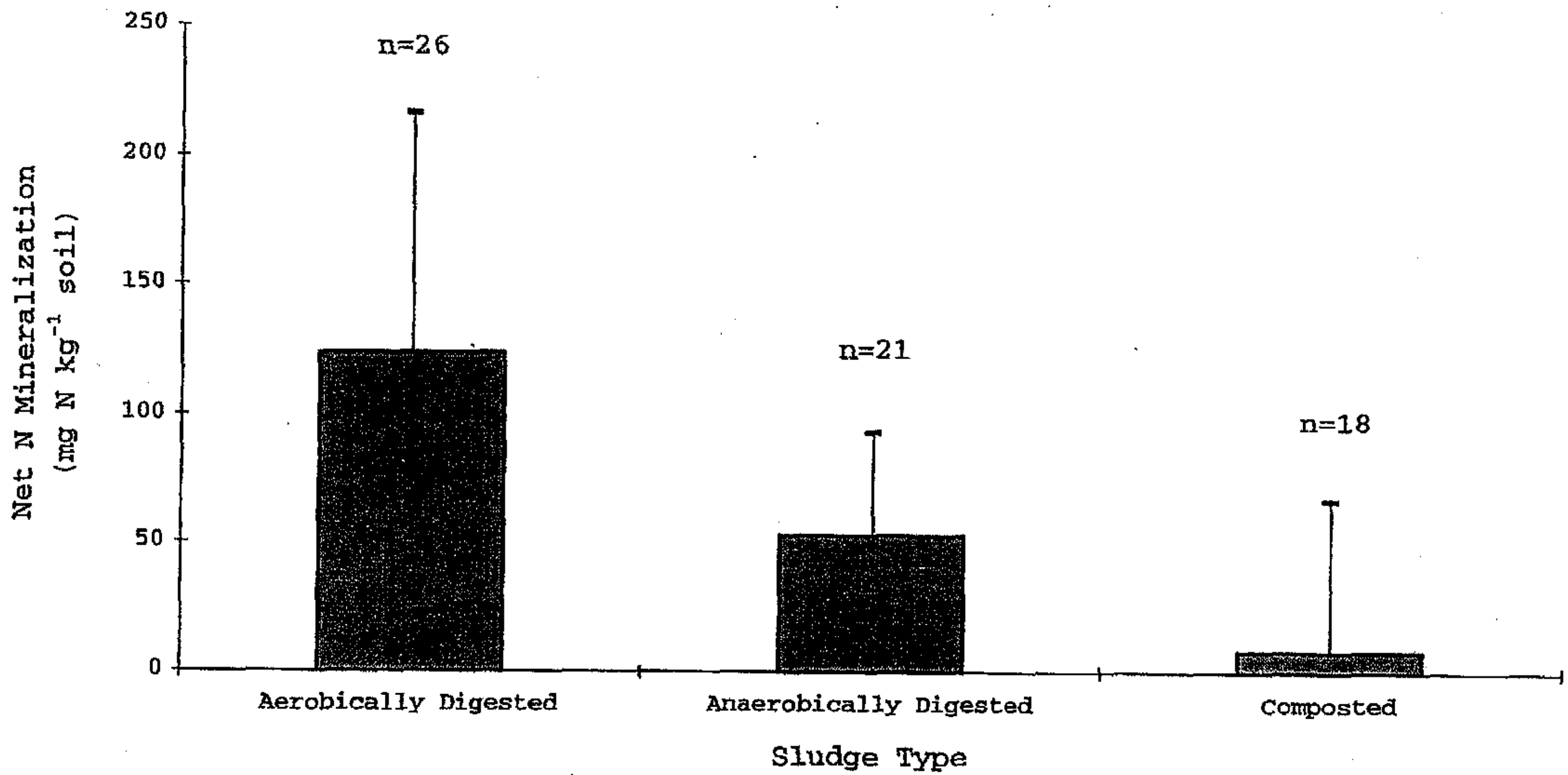
However, solids in sludge are wet and compact, which limits the O_2 supply to microbes. Due to the limited supply of O_2, aerobic sludge treatment usually is less efficient in biodegrading organic N in sludge than anaerobic sludge treatment (Hobson et al., 1974). This results in more organic N remaining in aerobically digested sludge than in anaerobically digested sludge. A survey of 123 sludges from 6 north-central states of the U.S. showed that aerobically digested sludges have approximately 25 % more organic N than anaerobically digested sludges (Sommers, 1977). Consequently, aerobically digested sludges tend to have higher net N mineralization when they are applied to soil than anaerobically digested sludges (Figure 14). When the same experimental conditions are used, aerobically digested sludge yields 2 to 5 times higher net N mineralization than anaerobically digested sludges (Hsieh et al., 1981b; Parker and Sommers, 1983).

As sludge application rate increases, the difference in net N mineralization arising from sludge type decreases. In a study by Hsieh et al. (1981b) net N mineralization was approximately 2 times higher in an aerobically digested sludge than in an anaerobically digested sludge when the application rate was 80 Mg ha^{-1}, and only 1.2 times higher when the application rate was 160 Mg ha^{-1} and above. This may be related to decreased % mineralization when sludge application rates are high. Consequently, the higher the sludge application rate, the less difference in net N mineralization among different sludge types.

The lowest net total N mineralization occurs in composted sludges (Figure 14). Composted sludges used in these studies have the smallest mean organic N content (1.4 %), compared to that of aerobically (3.1 %) and anaerobically (2.0 %) digested sludges. Furthermore, composted sludges tend to have higher C:N ratios than digested sludges. This also contributes to low net N mineralization. Net N immobilization occurs when the C:N ratio of composted sludge exceeds 30:1 (Sims, 1990).

Although the type of treatment affects organic N content, it has small influence on the organic N composition of sludge. Higgins et al. (1982) treated a raw sludge by different stabilization methods. They determined that the amino acid contents of treated sludges were similar regardless of the sludge stabilization method employed (Table 2). This only holds if the same raw sludge is used. Wastewaters entering treatment plants, therefore, raw sludge, differs greatly in chemical composition. Oba and Nguyen (1982) found that amino acids represented 47 % of the total hydrolyzable N in an aerobically digested sludge. Ryan et al. (1973) observed that it was 24 % in an anaerobically

Figure 14. The relationship between net N mineralization and sludge type†.



† Data from Epstein et al. (1978); Garau et al. (1986); Hsieh et al. (1981b); Lerch et al. (1992); Lindeman and Cardenas (1984); Oba and Nguyan (1982); Ryan et al. (1973); Sims (1990). Net N mineralization is as NH_4^+-N + NO_2^--N + NO_3^--N mineralized. Error bar and n represent 1 standard deviation and number of observations, respectively.

Table 2. Effect of stabilization method on the amino acid content of sludges[†].

Constituent	% of Total Organic N		
	Aerobically Digested	Anaerobically Digested	Composted
Alcohol-soluble	1.1	3.4	1.7
Alcohol/ether-insoluble	48.0	50.6	43.3

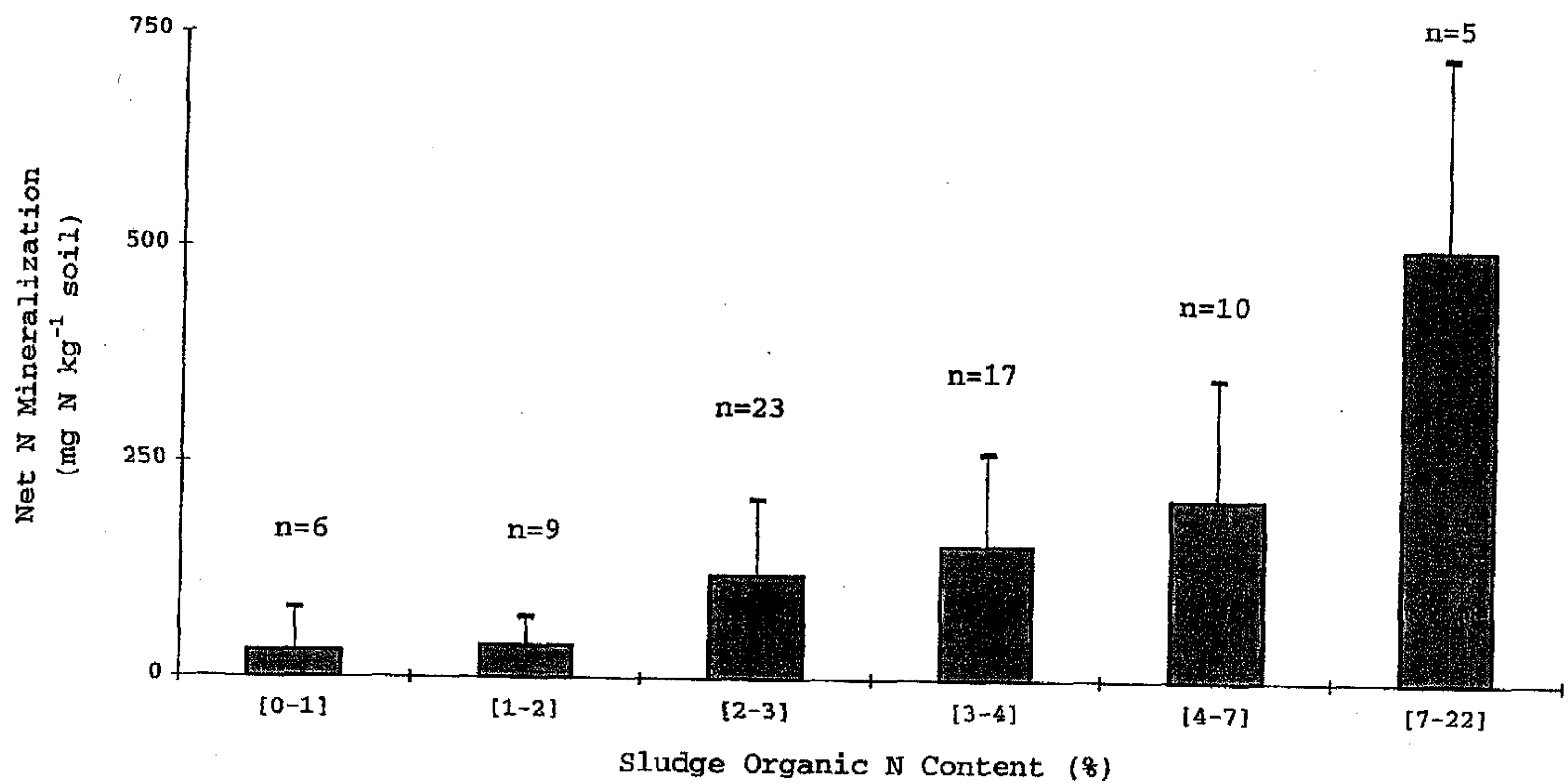
[†] Data from Higgins et al., 1982.

digested sludge. Oba and Nguyen (1982) measured 3 times higher net total N mineralization than Ryan et al. (1973).

As sludge organic N content increases net N mineralization increases but there is great variation in the individual classes (Figure 15). This variation could be explained by differences in the immobilization of N, and the composition of sludge organic N. Epstein et al. (1978) observed differences of up to 60 % in net N mineralization of soil amended with sludges, although the sludges had similar organic N contents. In their study, the C:N ratio of sludges ranged from 8:1 to 17:1, which could make N immobilization, thereby, net N mineralization quite variable.

Sludge proteins are important because of their degradability and their abundance in sludge. Most sludge contains approximately 20 % protein by dry weight (Hobson et al., 1974). However, Lerch et al. (1992) found that a regression for the relationship between extractable protein content of sludge and net N mineralization gave a low correlation. In contrast, a regression for the relationship between primary amines (molecular weight < 14,000) and net N mineralization gave high correlation. The effect of sludge C:N ratio was reflected in the latter regression whereas it wasn't reflected in the former regression. Had C:N ratio of sludge been reflected, the regression between protein

Figure 15. The relationship between net N mineralization and sludge organic N content†.



† Data from Epstein et al. (1978); Garau et al. (1986); Hsieh et al. (1981b); Lerch et al. (1992); Lindeman and Cardenas (1984); Oba and Nguyan (1982); Ryan et al. (1973); Sims (1990). Net N mineralization is as NH_4^+-N + NO_2^--N + NO_3^--N mineralized. Error bar and n represent 1 standard deviation and number of observations, respectively.

content of sludge and net N mineralization could have also yielded a high correlation.

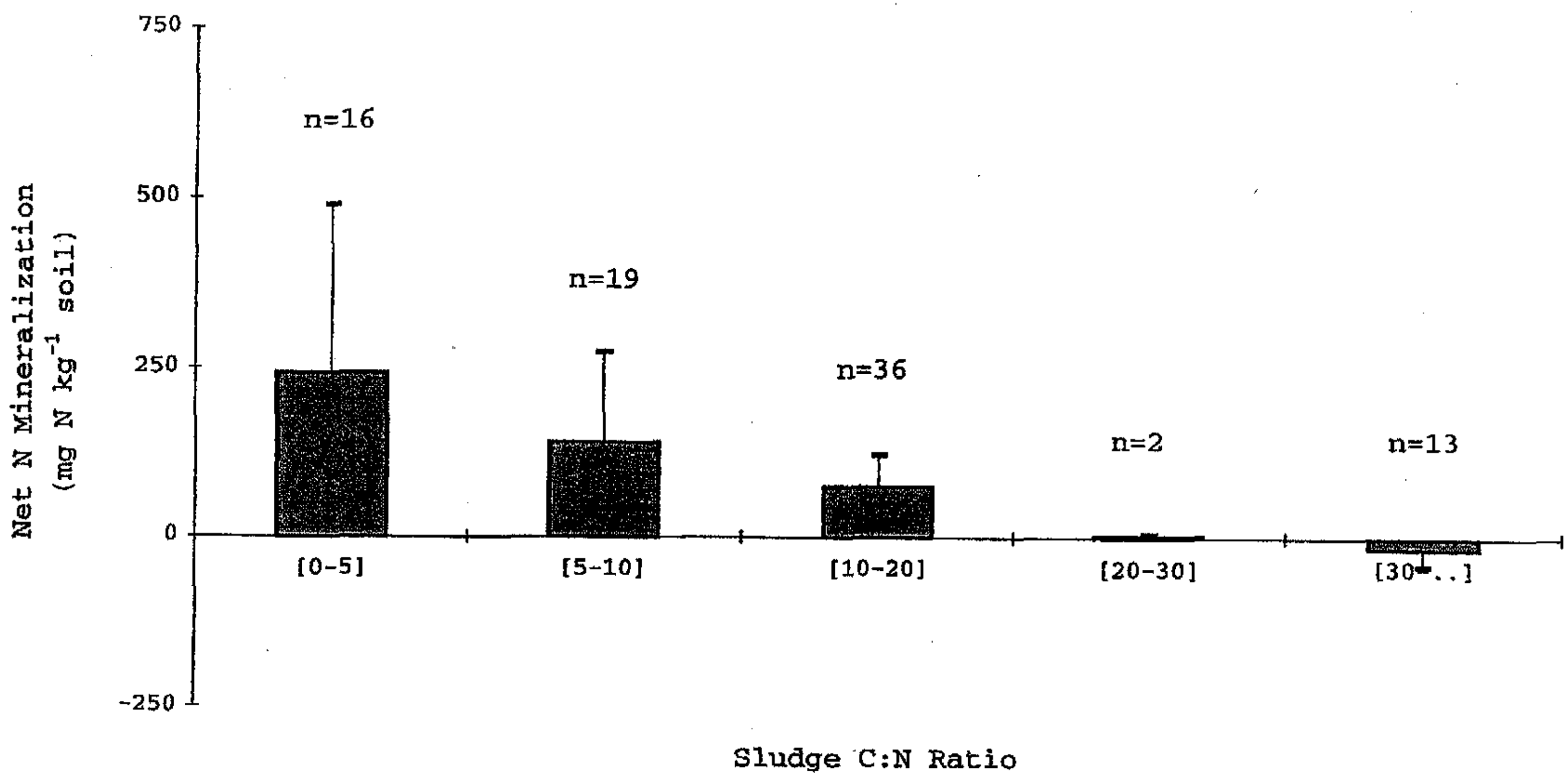
Therefore, the conclusion of Lerch et al. (1992) that sewage sludge proteins may not be a labile organic N pool doesn't hold. In fact, there is evidence that suggests otherwise. Hattori (1988) observed high correlations between proteinase activity and net N mineralization. This, and the high correlation between proteinase activity and numbers of bacteria during early incubation suggest that proteins in sludge are a labile organic N pool.

Proteins may not be the only abundant labile organic N in sludge. Most of the aerobically digested sludge organic N compounds have molecular weights less than 1,500. Approximately 5 % fall in the range of 0 to 165, 50 % in the range from 165 to 1,800, 10 % are greater than 1,800, and 35 % are unaccounted for (EPA, 1978). Molecular weights of up to 165 include organic N constituents such as amino acids, urea and small nucleic acid bases. Molecular weights ranging from 165 to 1,800 include small proteins, peptides and nucleotides. Those molecular weights that are greater than 1,800 reflect large proteins, enzymes, RNA, DNA, and humic acids. There is little information about how these constituents of sludge affect net N mineralization. But it is most likely that as labile organic N, including amino acids, peptides and small proteins, in sludge increase net N mineralization increases.

As the C:N ratio of sludge increases, net N mineralization decreases (Figure 16). The ratio of C:N for the cellular component of soil microorganisms ranges from 5:1 to 10:1. Soil microorganisms assimilate C and N from sludge organic compounds to synthesize new cells. If the organic C is too high relative to available N, soil microorganisms will utilize mineral N from the environment. The critical C:N ratio of substances entering soil ranges from 20:1 to 30:1. Wider ratios than this range favor N immobilization and narrower ratios favor N mineralization.

Several studies have mentioned N immobilization in sludge-amended soils (Epstein et al., 1978; Parker and Sommers, 1983; Tester et al., 1977). Sims (1990) measured net N immobilization in soils treated with composted sludge. As sludge C:N ratio increased from 33:1 to 53:1, at an application rate of 22 Mg ha^{-1}, net N immobilization increased from 7 ppm N to 39 ppm N. In the same study, sludge with a C:N ratio of 60:1 immobilized less N than sludge with a C:N ratio of 53:1. However, the sludge with a C:N ratio of 60:1 contained limestone which could have neutralized the acid soils used in this study, and therefore provided a pH more conducive for N mineralization. Another explanation for the lower immobilization of N in this sludge

Figure 16. The relationship between net N mineralization and sludge C:N ratio[†].



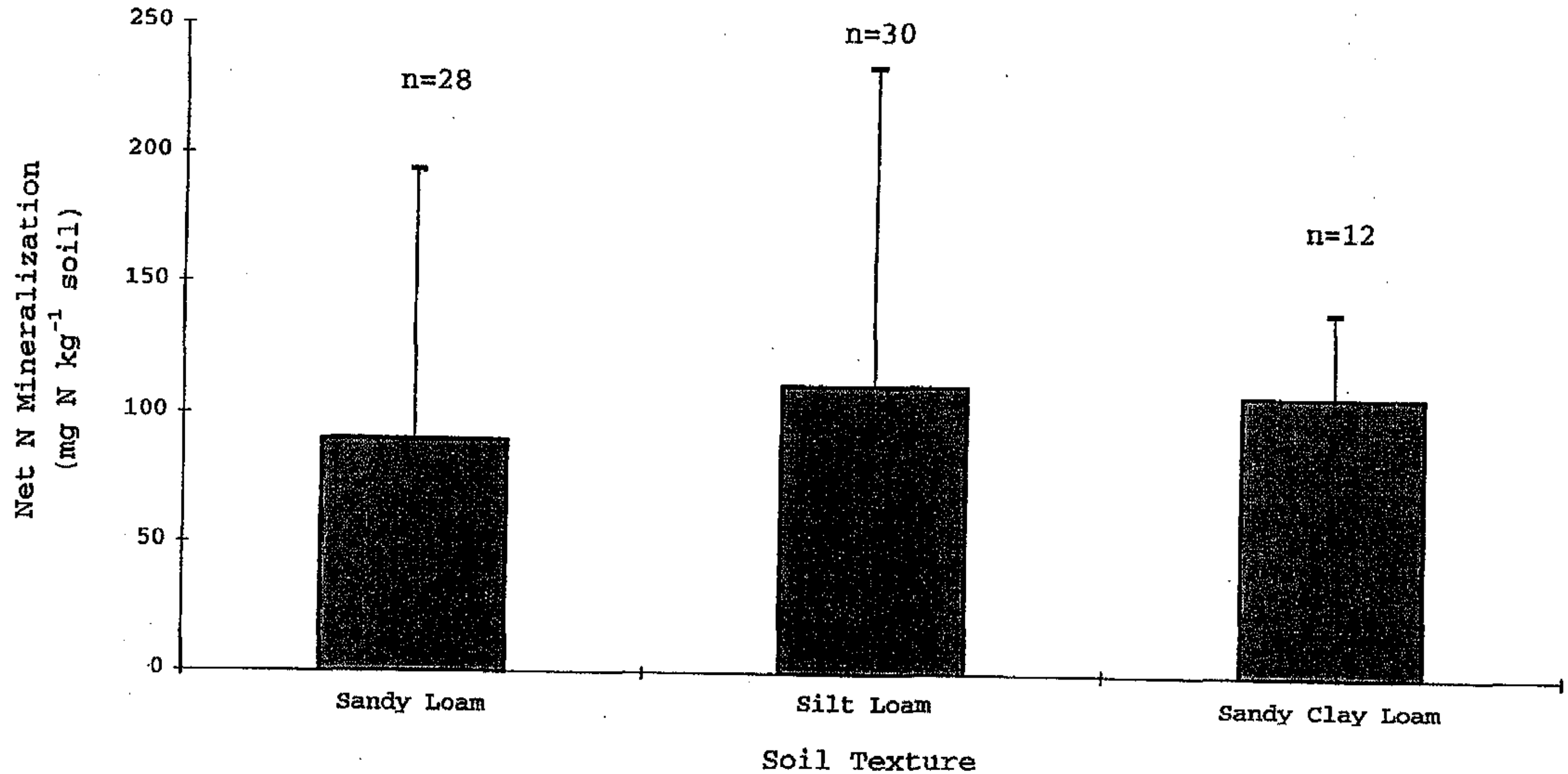
[†] Data from Epstein et al. (1978); Garau et al. (1986); Hsieh et al. (1981b); Lerch et al. (1992); Lindeman and Cardenas (1984); Oba and Nguyan (1982); Ryan et al. (1973); Sims (1990). Net N mineralization is as NH_4^+-N + NO_2^--N + NO_3^--N mineralized. Error bar and n represent 1 standard deviation and number of observations, respectively.

might relate to the availability of organic C. As we have seen, the availability of organic C in sludges, thus, total C mineralization, differs greatly. If sludge organic C is less available, its utilization by microbes in soil will be reduced. In this case, mineral N will be utilized to a lesser extent because demand for it is reduced. One implication of these observations is that aerobically digested sludges have more labile organic C, therefore, they could immobilize more mineral N than anaerobically digested sludges, even though both sludges would have similar C:N ratios.

In Sims' (1990) work, as more sludge with a high C:N ratio was added, plant dry weight decreased. As more sludge is added more mineral N will be incorporated into microbial biomass. Soil microorganisms can outcompete plant roots in utilizing the mineral N.

A summary of data from 9 studies shows that soil texture has little effect on net N mineralization in sludge treated soils (Figure 17). Lindeman and Cardenas (1984) found there wasn't a significant difference in N mineralization potential of a clayey soil and of a sandy soil. The soils used in this study varied greatly in texture, but they had similar organic N and C levels, and similar pH values.

Figure 17. The relationship between net N mineralization and soil texture[†].



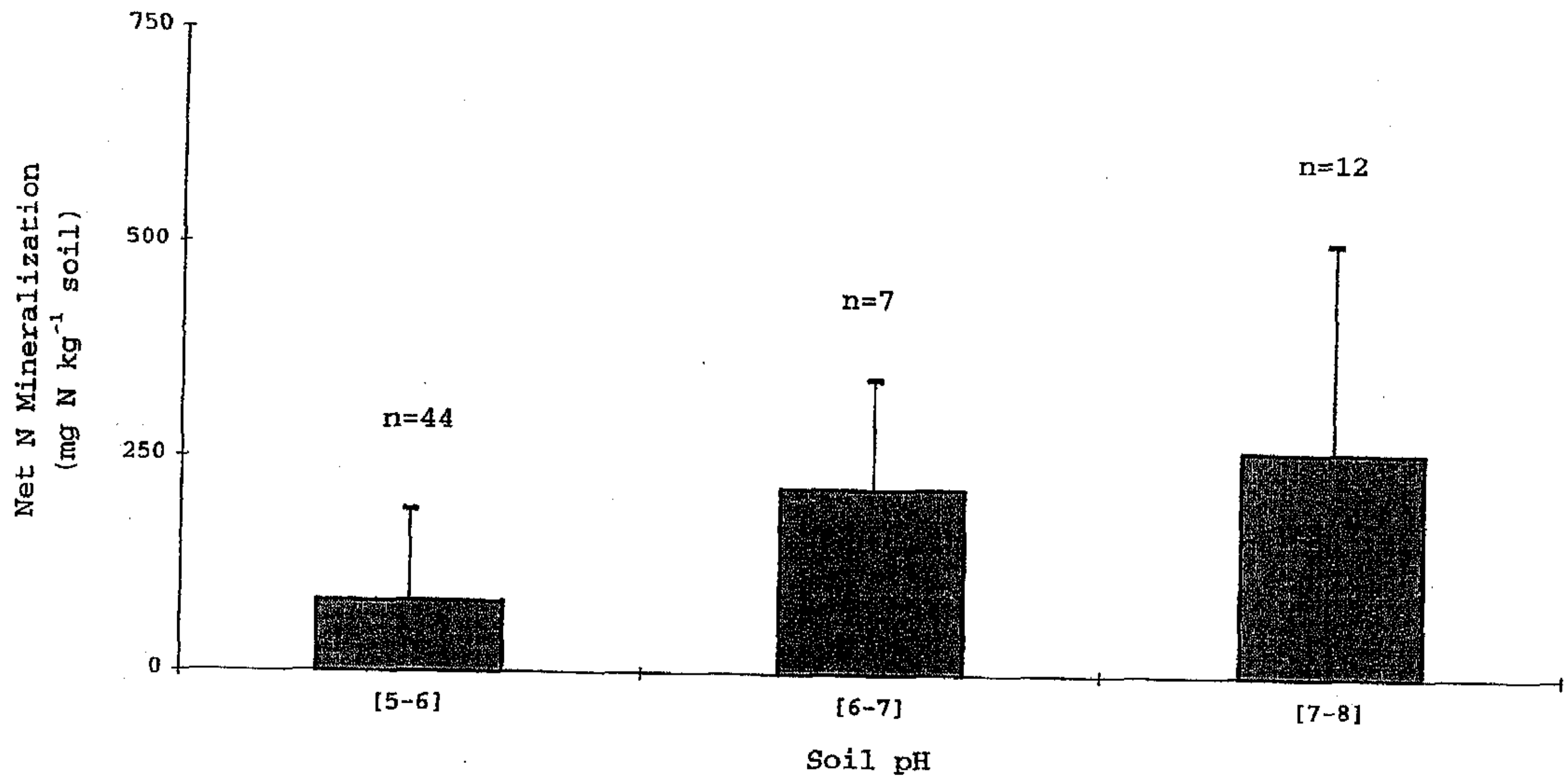
[†] Data from Epstein et al. (1978); Garau et al. (1986); Hsieh et al. (1981b); Lerch et al. (1992); Lindeman and Cardenas (1984); Oba and Nguyan (1982); Ryan et al. (1973); Sims (1990). Net N mineralization is as NH_4^+-N + NO_2^--N + NO_3^--N mineralized. Error bar and n represent 1 standard deviation and number of observations, respectively.

Net N mineralization in sludge amended soil was highest when soil pH was neutral (Figure 18), since neutral pH levels tend to maximize heterotrophic growth. Most bacteria grow best at higher pH (6.5-7.5), and as soil pH decreases, net N mineralization decreases.

Garau et al. (1986) applied a neutral sludge at rates up to 20 Mg ha^{-1} to soils having pH levels of 5.5 and 7.8. The net N mineralization values were approximately 2 times higher in the more alkaline soil. The soil pH wasn't the only factor accounting for this difference. Organic N content was 3 times higher and cation exchange capacity (CEC) was 4 times higher in the alkaline soil than in the acidic soil. As the organic N content of sludge increases, the N mineralization potential of soil increases. High organic matter content and CEC mean high buffering capacity, which could decrease the adverse effects of sludge salinity and are also well-correlated with high microbial populations.

Sludges with high organic N content can increase soil pH. Hattori (1988) treated an acidic soil with sludges ranging in pH from 5.3 to 9.0, at rates up 100 Mg ha^{-1}. Net N mineralization values were much higher in sludge with the lowest pH than in any other sludge, independent of sludge application rate. However, the organic N content of the most acidic sludge was approximately 2 to 6 times higher than

Figure 18. The relationship between net N mineralization and soil pH[†].



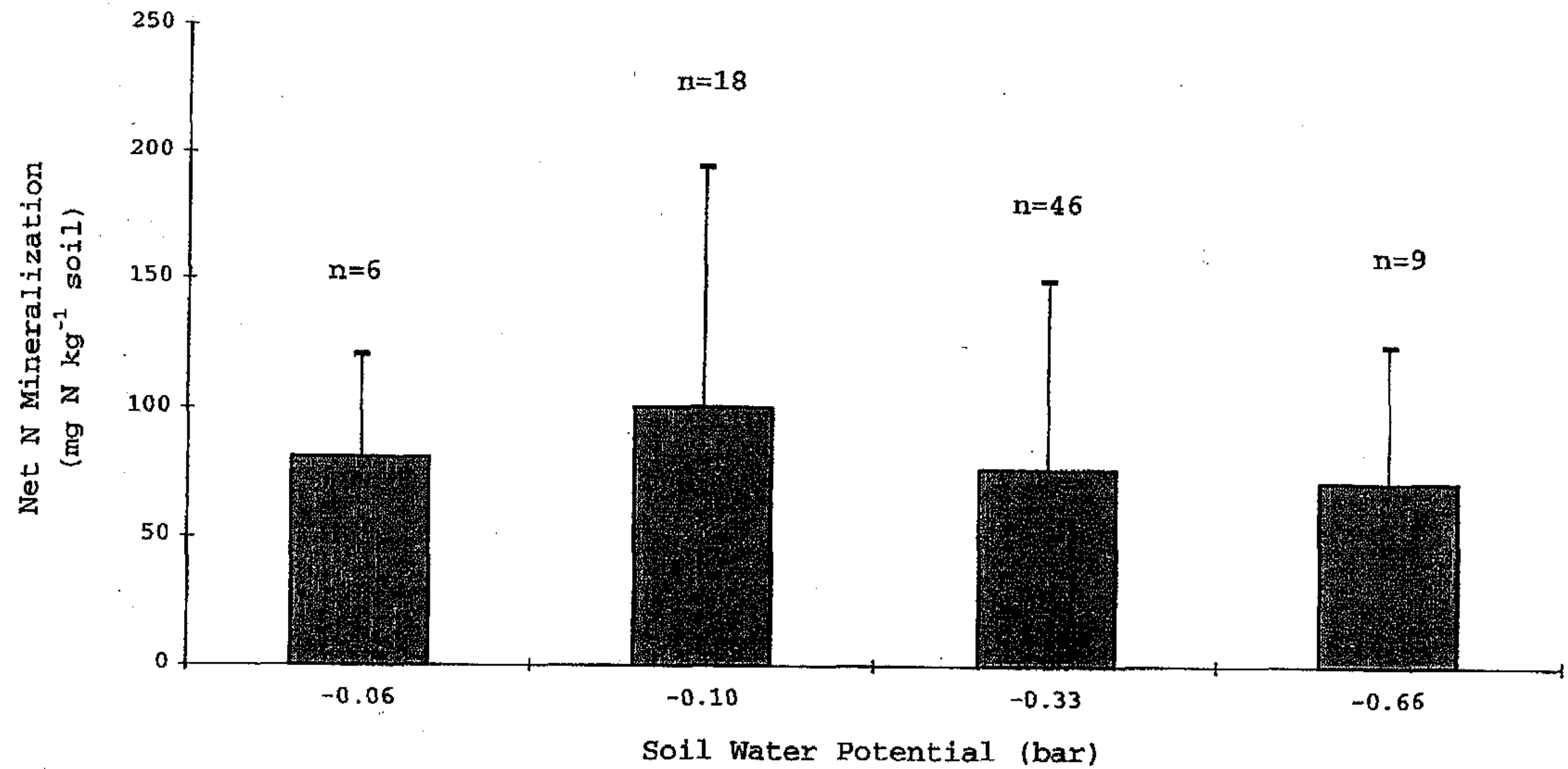
[†] Data from Epstein et al. (1978); Garau et al. (1986); Hsieh et al. (1981b); Lerch et al. (1992); Lindeman and Cardenas (1984); Oba and Nguyan (1982); Ryan et al. (1973); Sims (1990). Net N mineralization is as NH_4^+-N + NO_2^--N + NO_3^--N mineralized. Error bar and n represent 1 standard deviation and number of observations, respectively.

that of other sludges. The rapid initial NH_4^+ release in this sludge amendment caused NH_3 volatilization which increased the soil pH up to 8.0.

Soil water potential in the range of -0.06 bar to -0.66 bar has little effect on N mineralization in sludge treated soils (Figure 19). Hsieh et al. (1981b) noted that there wasn't a significant difference in N mineralization potential of sludge amended soils when soil water potentials were at -0.06 and -0.33 bar. After 2 weeks of incubation, NO_2^--N + NO_3^--N content of the samples increased at -0.33 bar but decreased at -0.06 bar. The change was proportional to the sludge application rate. This suggests that denitrification occurs at 0.06 bar, since there was considerable unaccounted N as measured by the difference between net organic N loss and net inorganic N gain. There was also unaccounted N at 0.33 bar, which could only be explained by loss of N via denitrification since NH_3 losses from treatments weren't significant.

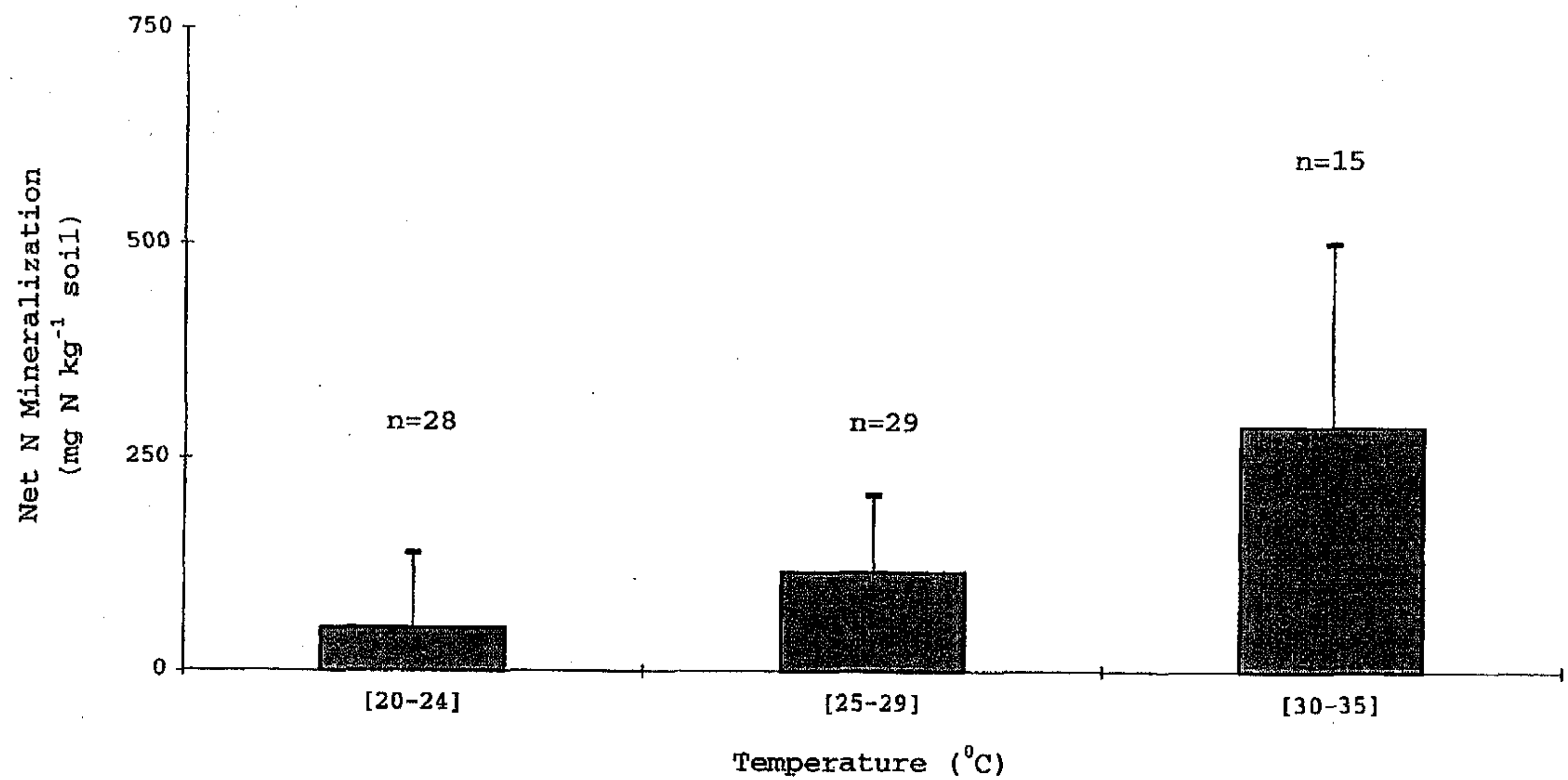
The temperature quotient of sludge organic N mineralization is approximately 2.0 at temperatures ranging from 20 ^{0}C to 35 ^{0}C (Figure 20). The temperature quotient of nitrification in sludge amended soil should also be close to 2.0. Terry et al. (1981) determined that a 10 ^{0}C increase in temperature increased nitrification approximately 2 fold. After 14 days of incubation, 22 % of the NH_4^+-N added by

Figure 19. The relationship between net N mineralization and soil water potential[†].



† Data from Epstein et al. (1978); Garau et al. (1986); Hsieh et al. (1981b); Lerch et al. (1992); Lindeman and Cardenas (1984); Oba and Nguyan (1982); Ryan et al. (1973); Sims (1990). Net N mineralization is as NH_4^+-N + NO_2^--N + NO_3^--N mineralized. Error bar and n represent 1 standard deviation and number of observations, respectively.

Figure 20. The relationship between net N mineralization and temperature[1].



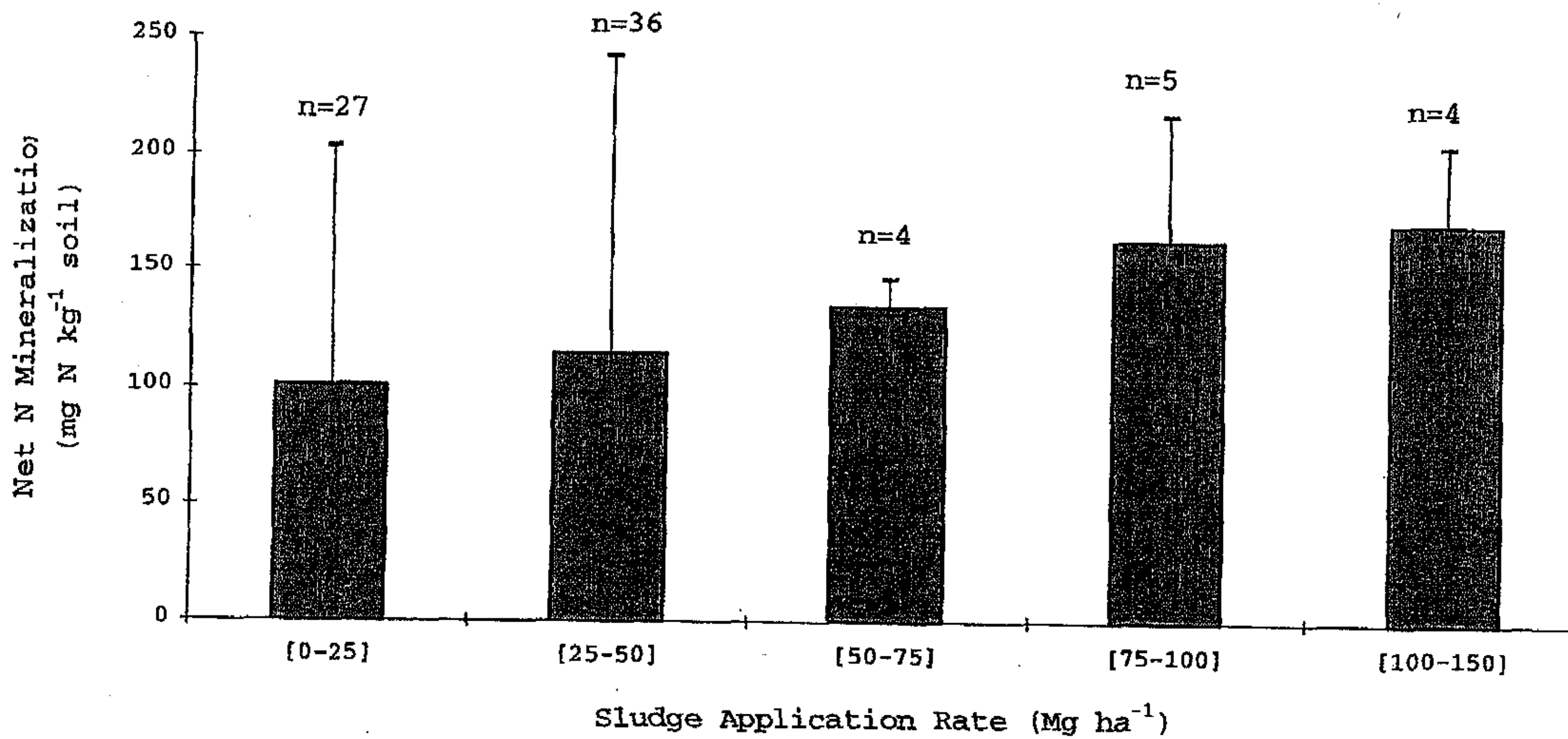
[1] Data from Epstein et al. (1978); Garau et al. (1986); Hsieh et al. (1981b); Lerch et al. (1992); Lindeman and Cardenas (1984); Oba and Nguyan (1982); Ryan et al. (1973); Sims (1990). Net N mineralization is as NH_4^+-N + NO_2^--N + NO_3^--N mineralized. Error bar and n represent 1 standard deviation and number of observations, respectively.

sludge nitrified at 15 ^{o}C, 54 % nitrified at 21 ^{o}C, and 92 % nitrified at 30 ^{o}C.

As sludge application rate increases net N mineralization increases due to greater organic N addition (Figure 21). However, as sludge application rate increases, the % of organic N in sludge mineralized decreases. The decrease in the % sludge organic N mineralization is probably related to sludge salinity and pH (Figure 22). Fine et al. (1989) observed that electrical conductivity (EC) in soil increased up to 0.9 dSm^{-1} when the sludge application rate was 20 Mg ha^{-1} and up to 8 dSm^{-1} when the sludge application rate was 200 Mg ha^{-1}. The % organic N mineralization in sludge was approximately 40 and 24 % in the 20 and 200 Mg ha^{-1} sludge treatments, respectively.

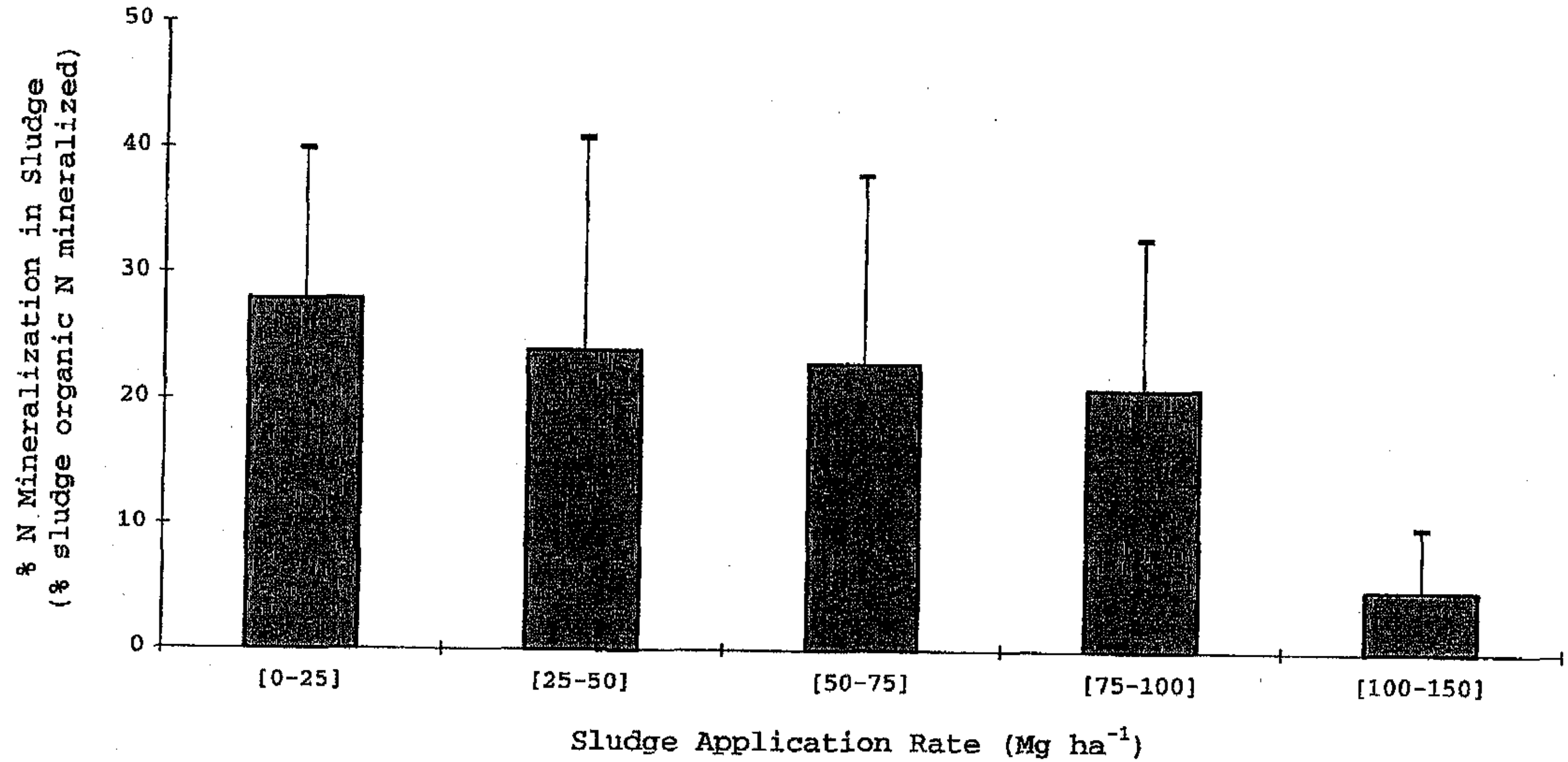
Tester and Parr (1983) observed that the % organic N mineralization in sludge at a 100 Mg ha^{-1} application rate was 15.0 % when sludge was leached prior to incubation and 3.4 % when sludge wasn't leached. A 200 Mg ha^{-1} sludge application rate resulted in 12.2 and 0.0 % organic N mineralization in leached and unleached sludge treatments, respectively. The large difference in % sludge N mineralization between leached and unleached sludge treatments when the application rate was the same and, the small difference between the leached sludge treatments at two different application rates, suggest that inhibition was

Figure 21. The relationship between net N mineralization and sludge application rate†.



† Data from Epstein et al. (1978); Garau et al. (1986); Hsieh et al. (1981b); Lerch et al. (1992); Lindeman and Cardenas (1984); Oba and Nguyan (1982); Ryan et al. (1973); Sims (1990). Net N mineralization is as NH_4^+-N + NO_2^--N + NO_3^--N mineralized. Error bar and n represent 1 standard deviation and number of observations, respectively.

Figure 22. The relationship between % sludge organic N mineralization and sludge application rate†.



† Data from Epstein et al. (1978); Garau et al. (1986); Hsieh et al. (1981b); Lerch et al. (1992); Lindeman and Cardenas (1984); Oba and Nguyan (1982); Ryan et al. (1973); Sims (1990). Error bar and n represent 1 standard deviation and number of observations respectively.

salinity related. More importantly, as sludge rate increased, the adverse effects of sludge salts on N mineralization increased. This is apparent in that as unleached sludge rate increased, the % organic sludge N mineralization decreased.

Hattori (1988) found that the % organic N mineralization was higher at 20 Mg ha^{-1} than at 100 Mg ha^{-1}. After 2 months of incubation, the pH levels decreased to approximately 6.5 and 5.0 in the 20 and 100 Mg ha^{-1} treatments, respectively. As the sludge application rate increases, N mineralization and nitrification potential increase. Nitrification is an acidifying process. Consequently, as nitrification increases, the soil will become more acidic and less suitable for microbial activity.

Although numerous studies have dealt with N mineralization in sludge-treated soil, only a few studies have assessed N mineralization in fly ash-LSR treated soil. Cervelli et al. (1987) determined that the effects of fly ash-LSR on net N mineralization in soil depended on the fly ash-LSR pH and the soil buffering capacity. For example, net N mineralization in an acid soil increased with amendment of a fly ash-LSR with a pH of 11.0, but decreased with amendment of a fly ash-LSR with a pH of 12.5. The former fly ash-LSR probably created more optimum pH levels in soil than the latter. When fly ash-LSR was added to 2 soils, initial soil pH levels were approximately 7.0. The soil pH increased to 8.7 in the poorly buffered soil and only up to 7.4 in the well buffered soil. Net N mineralization increased in the well buffered soil and didn't change in the poorly buffered soil. Net N mineralization in the well buffered soil increased, as fly ash-LSR application rate increased from 40 to 200 Mg ha^{-1}. Buffering capacity of soil is one important factor affecting the net N mineralization in fly ash-LSR amended soil.

In a follow up study, Cervelli et al. (1990) reported that addition of an alkaline fly ash-LSR at rates up to 60 Mg ha^{-1} to an acid soil previously amended with urea didn't affect net N mineralization.

To summarize, there have been many studies on C and N mineralization in sludge amended soils. In contrast, only a few studies have assessed C and N mineralization in fly ash-LSR treated soils. Even fewer studies have focused on soils amended with both sludge and fly ash-LSR.

Total C and N mineralization tend to increase as sludge application rate increases. In contrast, they usually decrease as fresh fly ash-LSR rates increase. Little information exists on whether total C and N mineralization increase when both sludge and weathered fly ash-LSR are applied at increasing rates. The two wastes combined may suppress microbial activity in soil and decrease total mineralization in soil.

Organic C and N mineralization in sludge, expressed as percentage of added organic C and N, decreases as the sludge application rate increases. In other words, the more sludge added, the less efficient the microbial utilization of sludge organic C and N. The most efficient % organic sludge C and N mineralization would probably occur at the lowest sludge application rate. The fly ash-LSR amendment may affect sludge mineralization. The nature and extent of this effect is unknown particularly when a weathered fly ash-LSR is used.

The following questions remain unclear.

1. Do sludge and fly ash-LSR increase C and N mineralization from the amended soil?
2. What are the sludge and fly ash-LSR rates that maximize total C and N mineralization?
3. Is there a synergistic effect of fly ash-LSR on sludge mineralization?

CHAPTER THREE: CARBON MINERALIZATION IN SLUDGE AND FLY ASH-LSR TREATED SOILS

INTRODUCTION

Many studies have dealt with sludge C mineralization in soil. These studies have described the effects of sludge and soil properties, and environmental conditions such as temperature. They have also shown that as sludge application rate increased, total C mineralization in sludge amended soil increased. Whether this relationship holds true for unlimited application rates is unclear. In other words, does the C mineralization rate increase regardless of how much sludge is added? Does it become constant beyond an application rate that can defined as a sludge loading rate for C mineralization?

Few studies have assessed C mineralization in fly ash-LSR treated soil. These studies found that unweathered fly ash-LSR decreased total C mineralization in fly ash-LSR-amended soil. Weathering processes remove salinity and the toxicity of B, and neutralize alkalinity in fly ash-LSR. Therefore, weathered fly ash-LSR should have little effect, if any, on total C mineralization. However, there isn't information concerning the effects of weathered fly ash-LSR on C mineralization in soil.

Carbon mineralization in sludge and fly ash-LSR treated soil has received little research attention. However, applying the two wastes together may have some advantages over applying them independently. For example, sludge provides organic C for soil microorganisms and fly ash-LSR can neutralize acid soils. In this case, each waste can have a positive effect on C mineralization in the amended soil.

Consequently, this study addresses the following questions by using fly ash-LSR and sludge application rates ranging from 0 to 135 Mg ha^{-1}.

1. What is the sludge loading rate that maximizes total C mineralization in the amended soil?
2. What is the fly ash-LSR loading rate with regard to C mineralization?
3. Does fly ash-LSR affect the sludge loading rate?

MATERIALS AND METHODS

Soil

The soil used in the incubations was a Pembroke silt (a fine silty, mixed, mesic Mollic Paleudalf) from a research site at the experimental farm of Western Kentucky University, Bowling Green, KY. Some chemical properties of the soil are given in Table 3.

Bulk soil samples were collected on 7 December 1994 to a depth of 15 cm. They were placed into 19 L plastic containers which were sealed with plastic lids before being transported to a greenhouse in Lexington, KY. The samples were stored without refrigeration in the greenhouse until 28 November 1995 when they were crushed through a 1 cm metal sieve and air dried for 1 week. Dried soil samples were returned to their original containers and stored a further 3 months in the greenhouse. They were sieved through a 2 mm metal sieve before use.

Sludge

I used aerobically digested sludge from Bowling Green Municipal Utilities, Bowling Green, KY. It was mixed with a polymer and passed through a mechanical dewatering press in the treatment plant to reduce its water content. Since the polymer content of sludge was approximately 0.05 %, the effect of this polymer on sludge decomposition in soil should be insignificant, but this wasn't tested. The sludge

Table 3. Selected properties of soil, sludge, and fly ash-LSR after air-drying.

Constituent	Soil	Sludge	Fly Ash-LSR
pH	7.5	7.1	7.8
CEC (meq 100 g^{-1})	15	ND	ND
EC (dS m^{-1})	1.25	7.35	2.45
BS (meq 100 g^{-1})	97.8	ND	ND
Total C (%)	1.8	24.8	0.58
Total N (%)	0.22	2.89	0
NH_4^+-N (mg kg^{-1})	51	4750	ND
NO_3^--N (mg kg^{-1})	154	656	ND
NO_2^--N (mg kg^{-1})	0	0	ND
Total P (%)	<0.002	2.41	<0.009
Total K (mg kg^{-1})	32	ND	ND
Total Ca (mg kg^{-1})	290	ND	ND
Total Mg (mg kg^{-1})	25	ND	ND
Total S (%)	ND	0.15	15.65
Metals (mg kg^{-1})			
Cd	0.4	0.3	0.6
Cr	0	0	2
Ni	0.7	14.5	0.4
Pb	8	8	4
Cu	4	6	7
Mo	0.1	0.1	3.1
Zn	8	22	18
B	<0.05	<0.02	50
Texture (%)			
Sand	6.86	ND	ND
Silt	67.91	ND	ND
Clay	25.22	ND	ND
WHC† (%)	33	ND	ND

† Water holding capacity is given as % gravimetric water content.
ND = Not determined.

was transported to the field where it was stored as a mass beneath a plastic cover for 4 months before sample collection. Some chemical properties of the sludge are given in Table 3.

The sludge was sampled on 27 April 1995. The samples were removed in bulk from the sludge mass and placed into 19 L plastic containers that were sealed with plastic lids before being transported to the greenhouse. The sludge was stored without refrigeration in the greenhouse for 9 months. It was air dried for 1 week, ground with a mechanical mixer, and passed through a 2 mm metal sieve before use.

Fly Ash-LSR (Limestone Scrubber Residue)

The fly ash-LSR was deposited in a pond at the Paradise Power Generation Plant, Drakesboro, KY. It was transported to the field where it was stored as a mass beneath a plastic cover for 2 months before sample collection. Some chemical properties of the fly ash-LSR are given in Table 3.

The fly ash-LSR was sampled on 7 December 1994. Bulk samples were removed from the fly ash-LSR mass and placed into 19 L plastic containers that were sealed, and transported to the greenhouse. The fly ash-LSR was stored without refrigeration in the greenhouse for 13 months. It was air dried for 1 week and sieved through a 2 mm metal sieve before use.

Experimental Design

Air dried sludge and fly ash-LSR were added to soil according to a complete factorial design with two factors: sludge addition and fly ash-LSR addition. Five rates were employed both for fly ash-LSR and sludge. These rates were 0, 2.5, 7.5, 22.5, and 67.5 g kg^{-1} soil (approximately corresponding to 0, 5, 15, 45, and 135 Mg ha^{-1} field application rates, respectively). The total number of treatments was 25 (5 sludge amendments × 5 fly ash-LSR amendments).

A 200 g batch of dry soil was combined with dried sludge, and fly ash-LSR at the chosen rates and mixed for 5 min in a mechanical mixer, and placed into soil sampling bags. Each bag contained approximately 50 g of amended soil. A 20 % gravimetric water content in the samples was obtained by adding distilled water to the bags which were well mixed afterwards. The 20 % gravimetric water content corresponded to 60 % water filled pore space in untreated soil. Addition of sludge and fly ash-LSR, particularly at high rates, might have altered the percent of water filled pore space in the mixtures, but this wasn't measured. Each treated soil was placed into a 500 mL mason jar. The jar was sealed with a gas-tight metal lid penetrated by a butyl rubber stopper and incubated at 35 °C for 28 days. The initial water content was maintained by spraying distilled water onto the incubating soils, as determined by periodically weighing the jars. Three separate incubations were performed for this study, which was necessitated by the availability of mason jars and the effort required to conduct analyses. There were 14 day intervals between incubations. The total study lasted for 56 days. The incubations were offset in this manner to optimize the time available for CO_2 measurements on a given sampling day.

Carbon Dioxide and Oxygen Measurements

A 1 mL gas-tight tuberculin syringe was used to sample the headspace of the jars. Gas samples were taken at 1 day intervals for the first 6 days and at 2 to 5 day intervals for the remainder of the incubation. The mason jars were vented for approximately 5 minutes after each sampling period. This sampling procedure ensured that O_2 concentrations in the jars remained above 10 % for the most of the incubation.

Gas samples were injected into a Varian 3700 gas chromatograph with a thermal conductivity detector (TCD) operated at 120 °C and a filament temperature of 140 °C. The carrier gas was Helium (He) with a flow rate of 20 mL min^{-1}. Measurements were made for CO_2 in all incubations, and for O_2 in the first incubation only. A 2 m Porapak Q column at a temperature of 60 °C was used for CO_2 analysis. A 2 m molecular sieve 5A column in line with the Porapak Q column (also at a temperature of 60 °C) was used for O_2 analysis. Carbon dioxide and O_2 concentrations were determined by measuring machine response to CO_2 and O_2 in samples and comparing these to the machine response to CO_2 and O_2 standards of known concentration.

Statistical Analysis

Studies have described sludge decomposition by first-order kinetics and used exponential models to describe it (Hsieh et al., 1981a; Lerch et al., 1992). In the current study, an exponential function, $y = a \ln(x) + t$, best described the relationship between carbon mineralizaton rate (y) and time (t). This function was applied to each sample. Coefficients a and intercepts c were determined for each treatment. These values were analyzed by using a randomized complete block design having 3 blocks (incubations). The treatment structure was factorial (5 sludge rate × 5 fly ash-LSR rate). The same design was used for the statistical analysis of total C mineralization values. The ANOVA tables are given in Appendix 1.

Analyses of Soil, Sludge, and Fly Ash-LSR

Air dried and sieved materials (soil, sludge and fly ash-LSR) were given to the Regulatory Services soil testing laboratory at the University of Kentucky for physical and chemical analyses. Only inorganic N (NO_2^--N, NO_3^--N, and NH_4^+-N) and soil water holding capacity were measured in laboratories of the University of Kentucky, Agronomy Department.

The following standard methods were used for chemical and physical analyses of the materials: pH by water (1:1 soil:water slurry); Cation Exchange Capacity (CEC) and Base Saturation (BS) by ammonium acetate extraction (1 N and pH=7); Electrical Conductivity (EC) by water saturation extract; Total C by wet combustion; Total N by combustion; Ammonium (NH_4^+-N) by Berthelot reaction on a microplate reader; Nitrite (NO_2^--N) by Griess reaction on a microplate reader; Nitrate (NO_3^--N) by Griess reaction after cadmium (Cd) reduction; Phosphorus (P), Potassium (K), Calcium (Ca), Magnesium (Mg), Sulfur (S) and Metals by Mehlich-III extraction; Texture by pipette method; soil water holding capacity by membrane apparatus. The specific protocols used for the analyses at the regulatory services laboratory were not obtained.

RESULTS AND DISCUSSION

Total C Mineralization

Oxygen concentrations in the mason jars were always above 10 % with the exception of day 1 in the 45 and 135 Mg ha^{-1} sludge treatments. A 10 % O_2 concentration should have been sufficient to maintain aerobic sludge decomposition in soil. Oxygen was measured only in incubation 1.

Total carbon mineralization from soils treated with sludge and fly ash-LSR for the 30 day incubation are given in Figure 23. With one exception, as sludge rate increased, total C mineralization increased, rising from 770 mg CO_2-C kg^{-1} soil in the treatments without sludge to 3810 mg CO_2-C kg^{-1} soil in the highest sludge application rate (135 Mg ha^{-1}). The increase in total C mineralization, as sludge application rate increased was significant ($P > .05$) and comparable to the total C mineralization observed in other studies (Figure 24). Fly ash-LSR didn't affect C mineralization. The ANOVA table is given in Appendix 1.

The total C mineralization values I measured are similar to results from a study that also used an aerobically digested sludge (Figure 24). However, there is not enough data to suggest that C mineralization values will always be similar when sludge is aerobically digested. Some wastewater treatment plants add chemicals to sludge,

Figure 23. Total C mineralization in sludge and fly ash-LSR treated soils during a 30 d incubation (The mean of 3 replications).

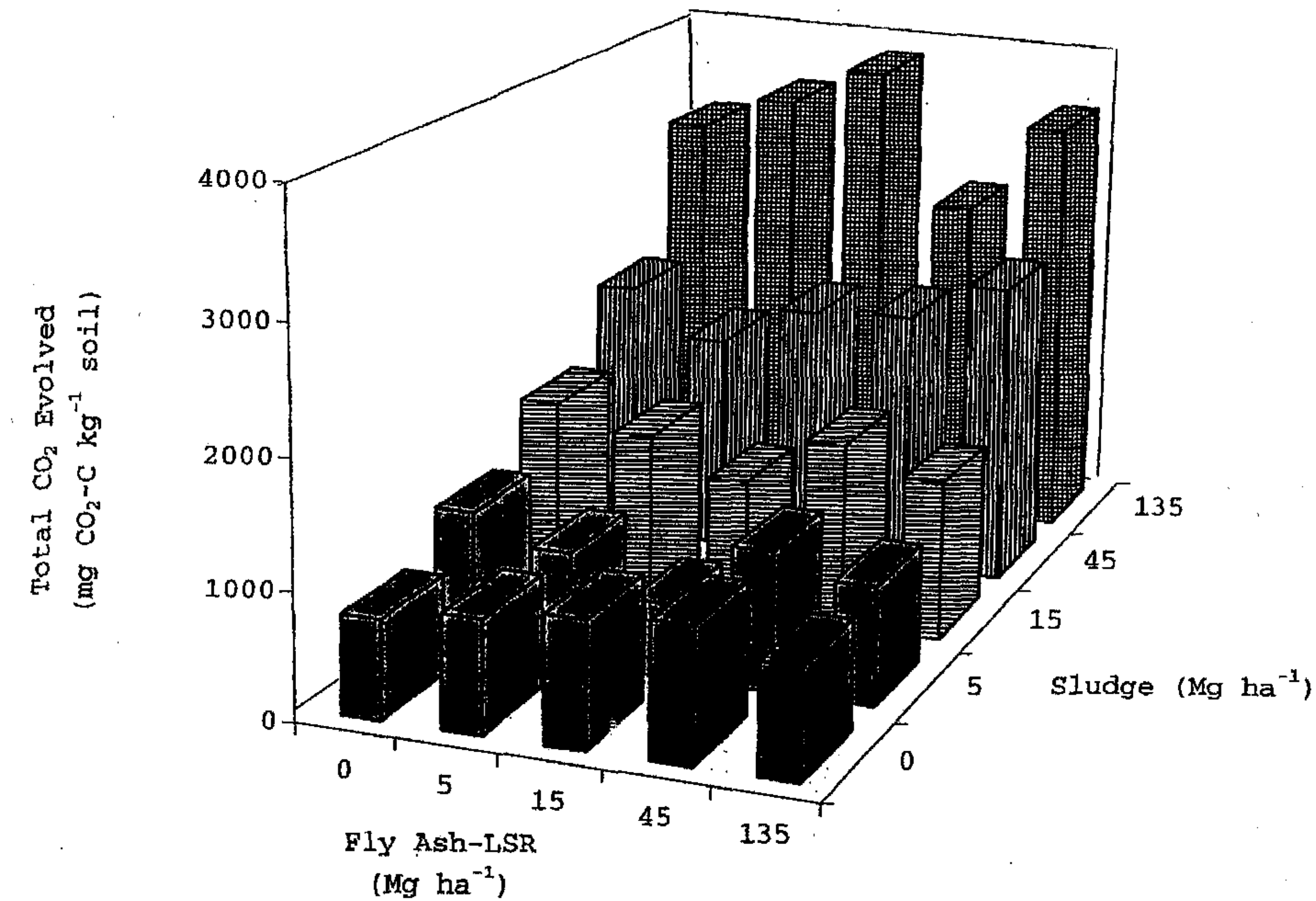
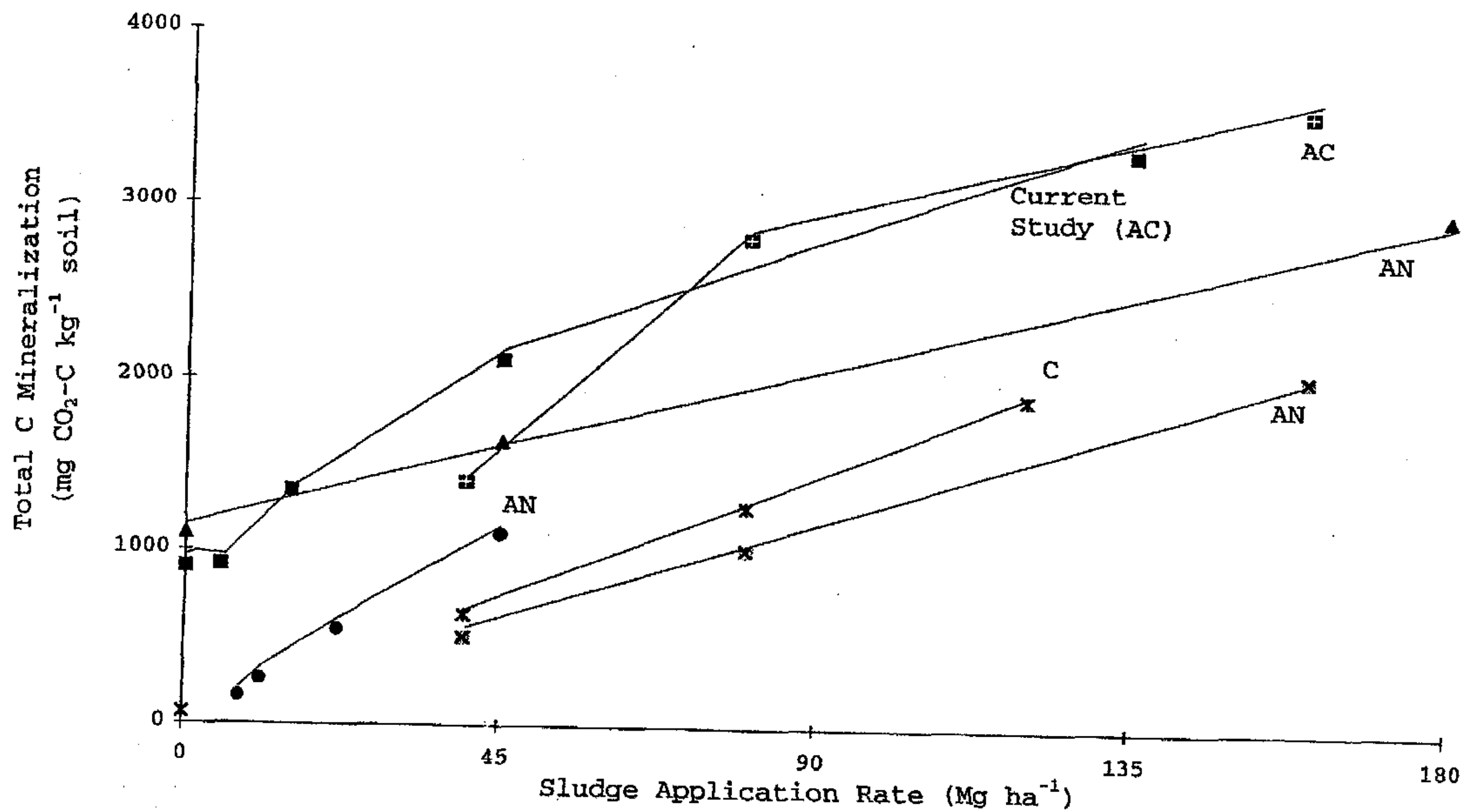


Figure 24. A comparision of total C mineralization values from the current and comparable studies[†].



[†] Data from the current study; Boyle and Paul (1989); Hsieh et al. (1981a); Terry et al. (1979); Tester et al. (1977). AC: Aerobically Digested Sludge; AN: Anaerobically Digested Sludge; C: Composted Sludge.

including lime [$Ca(OH)_2$], alum [$Al_2(SO)_4$], and ferric chloride [$FeCl_3$] to flocculate and condition sludge, or to remove P from the effluent (Tester and Parr, 1983). These chemicals increase salinity in sludge and soil amended with such sludges. Carbon mineralization in soil is higher when this salinity is removed from the sludge (Tester and Parr, 1983).

The method of sludge treatment is only one factor that affects sludge chemical composition. The chemical composition of sludges may greatly vary, even if sludges were stabilized by the same method. Other factors affecting sludge composition include the extent and nature of industrialization in the sanitary district and the seasonal variability of wastewater entering the wastewater treatment facility (Sommers et al., 1976).

Tester et al. (1979) found significant differences in total C mineralization in soil treated with a composted sludge's fractions, ranging in size <6 mm, 1 to 6 mm, and <1 mm. Although sludge application rate was the same in the treatments, C content varied. The highest C content and C mineralization were in soil treated with sludge fractions 1 to 6 mm in size, followed by those <6 mm and <1 mm in size. Bulking agent (woodchips) in the compost probably has little effect on C mineralization since: a) most of the bulking agent (woodchips) was removed when sludge was screened through 6 mm sieve; b) the % C mineralization of sludge were similar in the treatments, indicating that availability of C wasn't affected by the different sludge size fractions. Tester et al. (1979) didn't note why C content differed in different sized fractions.

My study and Hsieh et al. (1981a) sieved aerobically digested sludges through 2 and 6 mm sieves, respectively, and found similar total C mineralization values. However, Hsieh et al.'s (1981a) sludge had approximately 22.0 % more C than my study. The particle size of sludge seems to affect C mineralization, however, the nature of this effect for different types of sludge has yet to be identified.

Splitting the incubations led to significant differences among incubations ($p > .05$). The mean total C mineralization values were 1600, 1900, and 1700 mg kg^{-1} in incubations 1, 2, and 3, respectively. Incubation effects might be due to temperature fluctuations during the incubations. Prior to the experiment, the temperature inside the incubator ranged from 30 to 35 ^{0}C within a week. It wasn't recorded during the incubations.

The effect of different fly ash-LSR rates on C mineralization wasn't significant. Previous studies indicated that total C mineralization decreases in soil amended with unweathered fly ash-LSR having extreme pH levels (such as 5 and 12) (Pichtel, 1990) or high heavy metal contents (such as 1,000 ppm Zn) (Arthur et al., 1984). The inhibition of respiration was more apparent in these studies when application rates were above 200 Mg ha^{-1} and soil had low buffering capacity due to low cation exchange capacity and organic matter. Soils with high buffering capacities appear to be preferable for fly ash-LSR disposal since adverse effects of fly ash-LSR (such as increased alkalinity and more salt), if any, will be less in highly buffered soils than in poorly buffered soils.

Unlike Pichtel (1990) and Arthur et al. (1984), my study used a fly ash-LSR with a neutral pH, low heavy metal contents, and a soil having relatively high buffering capacity (Table 3). In addition, the application rates were lower in the current study than those in Pichtel (1990) and Arthur et al. (1984) works. Consequently, this fly ash-LSR up to 135 Mg ha^{-1} can be applied to soil as a means of waste disposal, without any appearent effect on C mineralization.

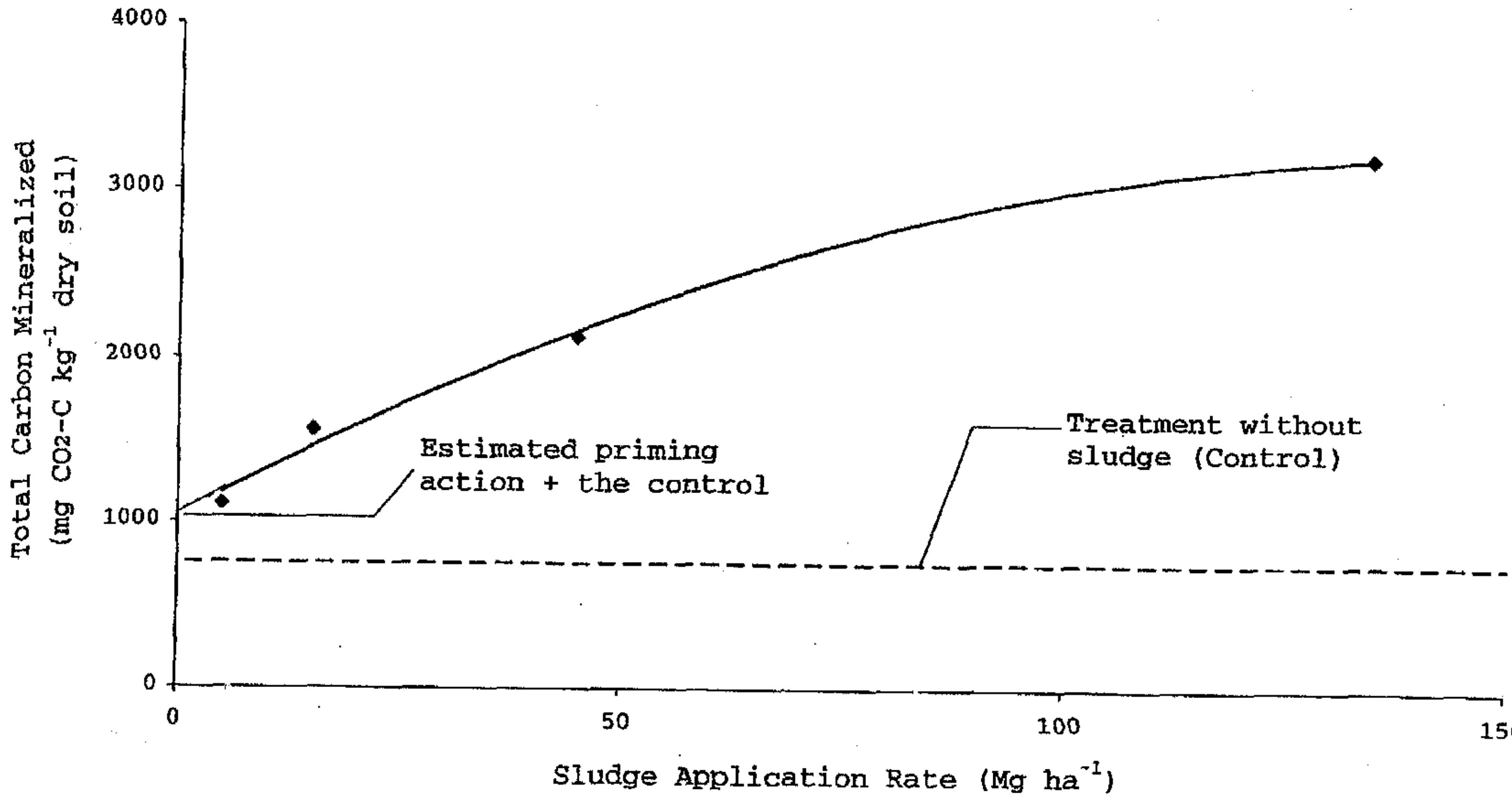
Priming Action of Sludge

An attempt was made to estimate the effect of sludge addition on the decomposition rate of native soil organic matter using Hsieh et al.'s (1981a) procedures. They suggested that a linear regression model for the relationship between sludge application rate (x) and total C mineralization in sludge amended soil (y) estimates priming action in sludge treated soil, when the regression model is extrapolated for x=0 (Figure 25).

The priming ratio is the ratio of total CO_2-C for the 0 Mg ha^{-1} sludge rate, estimated by the regression to total CO_2-C measured from the non-treated soil (Figure 25). The priming ratio in this study was 1.3, that is, sludge addition enhanced native organic C decomposition in soil by approximately 30 %. This ratio is close to that of the aerobically digested sludge, and higher than that of the anaerobically digested sludge studied by Hsieh et al. (1981a). This implies that aerobically digested sludges have similar priming effects, which exceed those of anaerobically digested sludges. Probably this is because anaerobically digested sludges have more lignin than aerobically digested sludges.

Determining the priming action in such a way has a major flaw. It assumes that the priming action at each sludge application rate is the same. However, as sludge application rate increases, labile organic matter and, therefore the priming action in the soil, should also increase. For example, Degens and Sparling (1996) found that priming ratios were approximately 1.3 and 1.7 in glucose amended soils when glucose was added at rates of 615 and 2457 ng C g^{-1} soil, respectively.

Figure 25. The mean total C mineralization values in the 0, 5, 15, 45, and 135 Mg ha^{-1} sludge treatments[†].



[†] Data are averaged across all fly ash-LSR rates and represent the mean of 3 replications.

Since the measurement of priming action relies on a regression model, the regression model only suggests that priming occurs in sludge treated soil. The priming action can be determined more accurately by the use of C tracers. Terry et al. (1979) quantified the CO_2 evolution due to priming action in a study that used anaerobically digested sludge labeled with ^{14}C . They determined that the priming ratio ranged from 1.4 to 2.3 in different soils treated with sludge at 22 Mg ha^{-1}. These values are higher than in my study. Since the organic C content of the soils and sludges used in the my study and Terry et al.'s (1979) work were similar, I may have underestimated the priming ratio by employing a regression model.

Percent C Mineralization in Sludge

Approximately 2, 26, 24, and 17 % of the organic carbon in sludge evolved as CO_2-C in the 5, 15, 45, and 135 Mg ha^{-1} sludge treatments, respectively. These percentages were found by Equation 1.

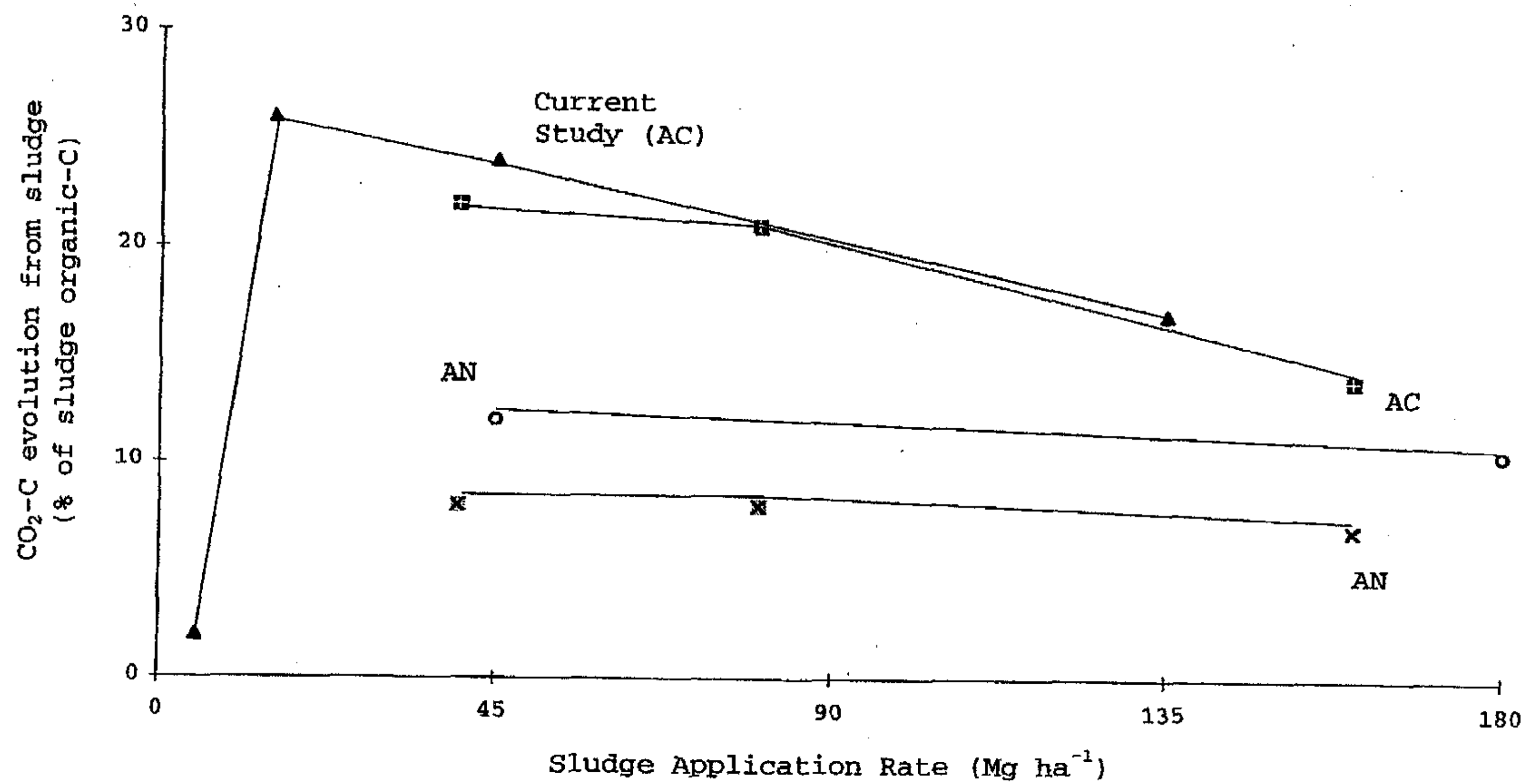
$$POCM = (T - C) / AOCS \qquad [1]$$

Where POCM= The % of organic sludge C mineralized, T= Total C mineralization in treated soil (mg CO_2-C kg^{-1} soil), C= Total C mineralization in soil without sludge (mg CO_2 kg^{-1} soil), and AOCS= Added organic C by sludge (mg C kg^{-1}). The difference between T and C is equal to carbon mineralization by sludge + priming action. Therefore, the percentages given include not only organic C mineralization from sludge, but also mineralization from soil due to priming effect of sludge.

There are 2 explanations for the low % C mineralization in the 5 Mg ha^{-1} sludge treatment. First, priming action in this treatment was expected to be low. Degens and Sparling (1996) found a positive relationship between the amount of glucose added to soil and the priming ratio. I didn't determine the soluble sugar content of my sludge, the soluble sugars, including glucose, are one important pool of substrate C in sludge (Hattori and Mukai, 1986; Rebhun and Manka, 1971; Schaumberg et al., 1980). Second, in the 5 Mg ha^{-1} sludge treatment, organic C in soil was approximately 3 times higher than that added by sludge. Consequently, the CO_2 analysis probably wasn't sensitive enough to differentiate the differences in CO_2 attributable to soil and sludge in this treatment.

The most efficient C mineralization occurred at the 15 Mg ha^{-1} sludge rate. This treatment maximized CO_2 evolution per unit of sludge (Figure 26). The percent C mineralization in sludge decreased as sludge application rate increased beyond 15 Mg ha^{-1}. These decreases may be related to salinity in the treatments, since, as sludge rate increases salinity should increase.

Figure 26. A comparision of normalized C mineralization values from the current and the comparable studies[†].



[†] Data from the current study; Boyle and Paul (1989); Hsieh et al. (1981a). AC: Aerobically Digested Sludge; AN: Anaerobically Digested Sludge; C: Composted Sludge.

Electrical conductivity values (EC) weren't measured in my study, but others did in comparable studies. Fine et al. (1989) determined that electrical conductivity (EC) values in sludge amended soil increased as sludge application rate increased during a 30 day incubation. The EC increased from 0.6 to 0.7 dSm^{-1} when sludge application rate was 20 Mg ha^{-1}. It increased from 2.0 to 8.0 dSm^{-1} when sludge rate was 200 Mg ha^{-1}. Pichtel (1990) amended soil with an EC of 0.07 dSm^{-1} with a sludge having an EC of 5.8 dSm^{-1} at a rate of 100 Mg ha^{-1}. Electrical conductivity in amended soil increased up to 503 dSm^{-1} after 42 days of incubation. Although EC wasn't measured, I calculated initial EC values by using EC of soil and sludge, and the dilution factor by sludge amendment. They range from 1.25 dSm^{-1} in soil without sludge to 1.80 dSm^{-1} in the highest sludge treatment (135 Mg ha^{-1}).

The % C mineralization in sludge, like total C mineralization, was similar to a comparable aerobically digested sludge and both were higher than anaerobically digested sludges (Figure 26). Anaerobically digested sludges tend to have resistant forms of organic C, resulting in less C mineralization when they are applied to soil than when aerobically digested sludges are soil applied.

Additional sludge had little effect on the % C mineralization in anaerobically digested sludges (Figure 28). This further indicates that anaerobically digested sludges are well stabilized and have more resistant compounds.

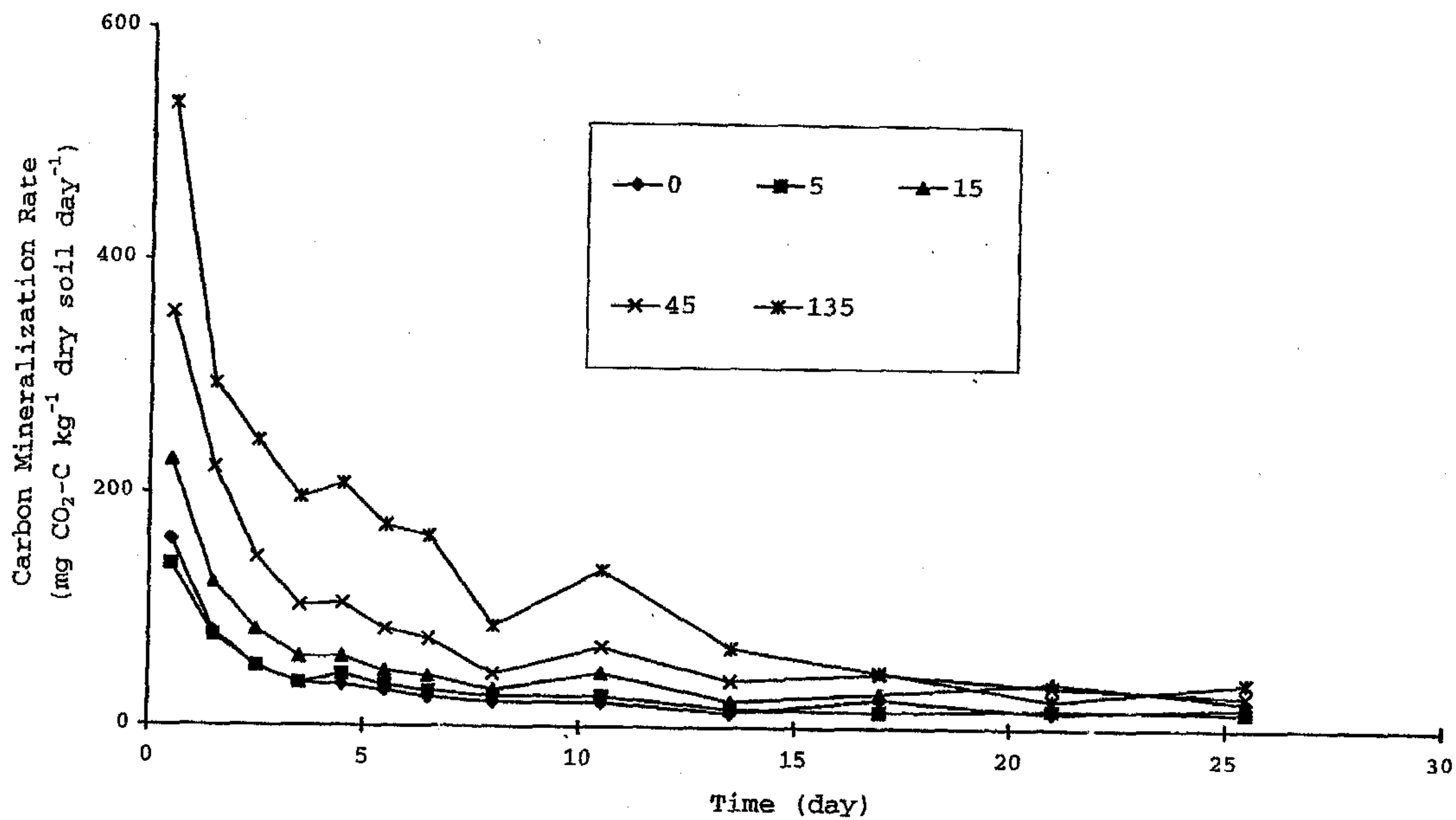
Pattern of C-Mineralization During the Incubations

The relationship between time and mean carbon mineralization rates in the sludge treatments are shown in Figures 27, 28, and 29, representing incubations 1, 2, and 3, respectively. Carbon mineralization rates were high at the beginning of the incubation due to redistribution of soil organic matter by sieving (Beare et al., 1994; Ross et al., 1985), labile organic carbon addition by sludge (Hattori, 1988; Hsieh et al., 1981a; Neilson and Pepper, 1990; Terry et al., 1979) and suitable water and temperature regimes for microbial growth (Blet-Charaudeau et al., 1990).

Respiration rates typically declined for 2 weeks and stayed relatively constant thereafter (Figure 27 to 29). Similar trends were also observed in previous sludge studies (Hattori, 1988; Hsieh et al, 1981a; Terry et al., 1979). The trend in this study is closest to that of Hsieh et al. (1981a), who used aerobically digested sludge. This pattern suggests that labile organic C in sludge degrades first and then more resistant organic C decomposes at slower rates.

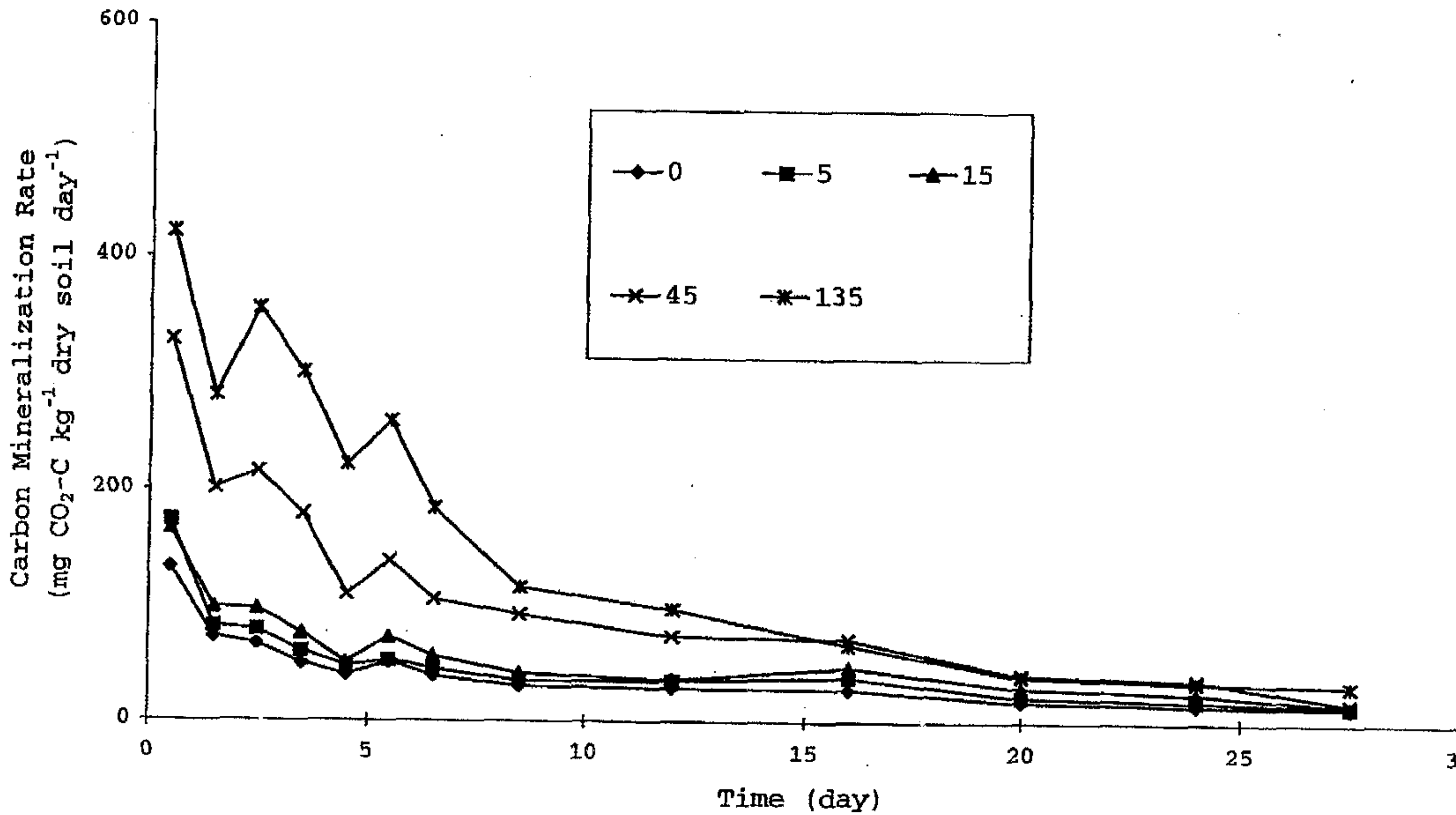
Sludge decomposition in soil has been described by first-order models (Gilmour et al., 1996a; Gilmour et al. 1996b; Gilmour and Gilmour, 1980; Lerch et al., 1992; Terry et al., 1979). These models assume that there are two

Figure 27. The mean C mineralization rates in the sludge treatments[†] (Incubation 1).



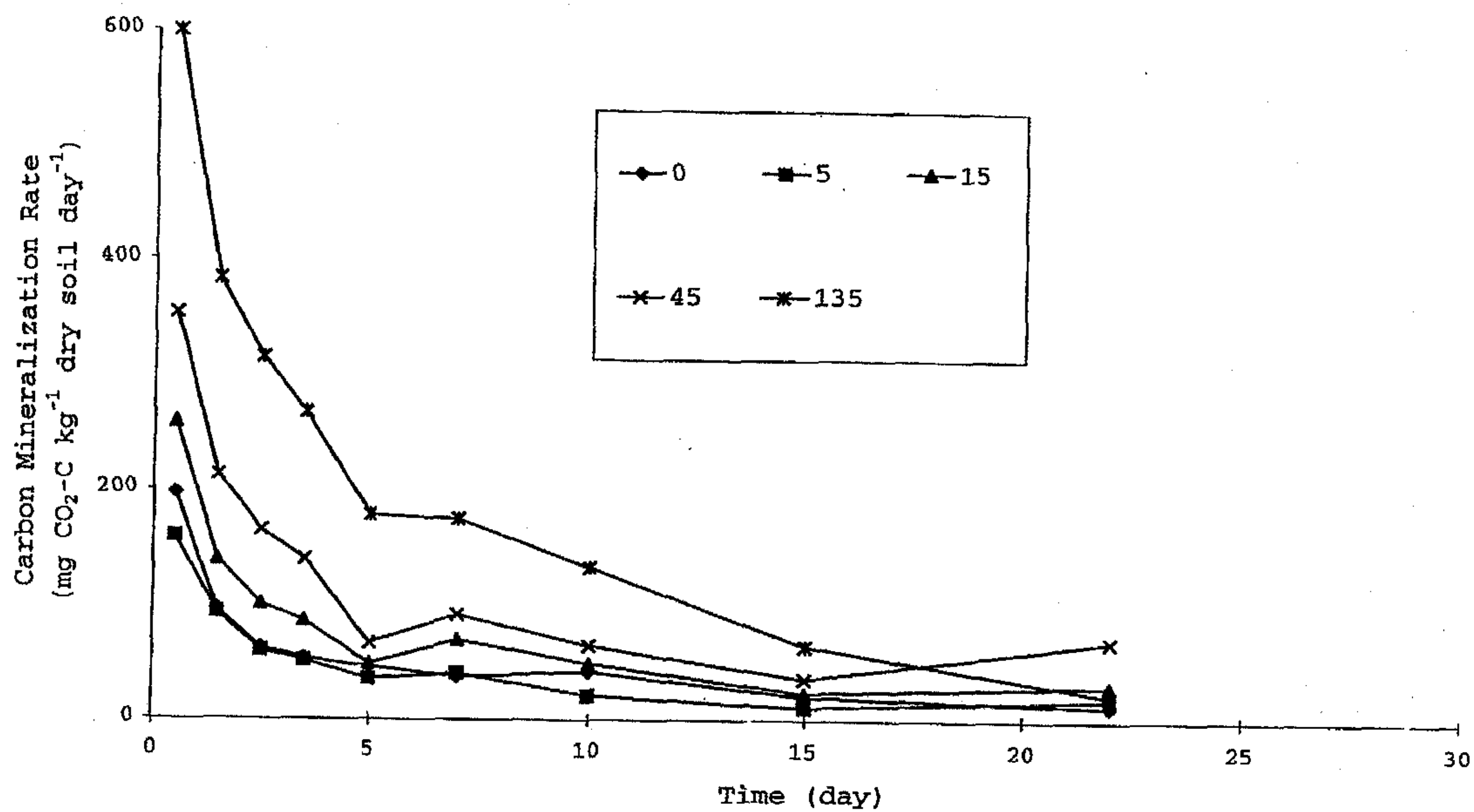
[†] Data are averaged across fly ash-LSR rates (n=5). Sludge rates are given as Mg ha^{-1}.

Figure 28. The mean C mineralization rates in the sludge treatments† (Incubation 2).



† Data are averaged across fly ash-LSR rates (n=5). Sludge rates are given as Mg ha^{-1}.

Figure 29. The mean C mineralization rates in the sludge treatments† (Incubation 3).



† Data are averaged across fly ash-LSR rates (n=5). Sludge rates are given as Mg ha^{-1}.

organic fractions in sludge: rapidly and slowly decomposable organic pools. Depending on whether these two fractions decompose independently of each other one can use two different first order decomposition models (Gilmour et al., 1996b). In the simultaneous decomposition model, rapidly and slowly decomposable organic fractions are assumed to decompose simultaneously and independently of each other (Equation 2).

$$\text{POCM} = \%\ \text{Rapid}\ [1 - \exp(-k_r t)] + (100 - \%\ \text{Rapid})\ [1 - \exp(-k_s t)] \quad [2]$$

Where POCM is the % of organic sludge C mineralized, % Rapid is the amount of sludge organic C in the rapidly decomposable fraction, t is time, and k_r and k_s are the rate constants for rapidly and slowly decomposable fractions, respectively.

In the sequential model, decomposition of the slowly decomposable fraction begins after the rapidly decomposable organic fraction mineralizes completely. Therefore, this model has 2 equations describing rapid (Equation 3) and slow (Equation 4) decomposition phases.

$$\text{POCM} = 100\ [1 - \exp(-k_r t_r)] \quad [3]$$

$$\text{POCM} = (100 - \%\ \text{Rapid})\ \{1 - \exp[-k_s(t-t_r)]\} \quad [4]$$

Where POCM, % rapid, k_r, and k_s were as described in Equation 2, and t_r is the period of decomposition of the rapid fraction.

Gilmour et al. (1996b) found higher correlations between the % decomposition at 7 days and % rapid fraction in the sequential model compared to a simultaneous model. Thus, they used a sequential model driven from Equations 3 and 4 to predict long-term decomposition (Equation 5).

$$POCM = \%\ Rapid + (100 - \%\ Rapid)$$
$$\{1 - \exp[-k_s \times (t - t_r)]\} \qquad [5]$$

Gilmour et al. (1996b) found a high correlation (R^2 =0.76) between predicted values and observed values for incubations longer than 60 days from their study and studies by Clark and Gilmour (1983), Lerch et al. (1992), and Terry et al. (1979). However, in all these studies, sludge application rates were low, within the range of 8 to 20 Mg ha^{-1}. Whether this model can estimate % decomposition correctly when the sludge rate is higher than 20 Mg ha^{-1} is unknown.

I employed a much simpler model than Gilmour et al.'s (1996b) models, using an exponential function (Equation 6), similar to that of Terry et al. (1979), to describe the relationship between time and sludge decomposition rate.

$$CMR = a\ [\ln\ (t)] + c \qquad [6]$$

Where CMR is carbon mineralization rate at time t, and a and c are constants that are specific to each treatment.

This function best described the relationship between time and carbon mineralization rate in the treatments. The

R^2 values for these regression ranged from 0.78 to 0.93. As sludge rate increased R^2 values increased. Regression lines of the sludge treatments in incubation 1 are given in Figure 30.

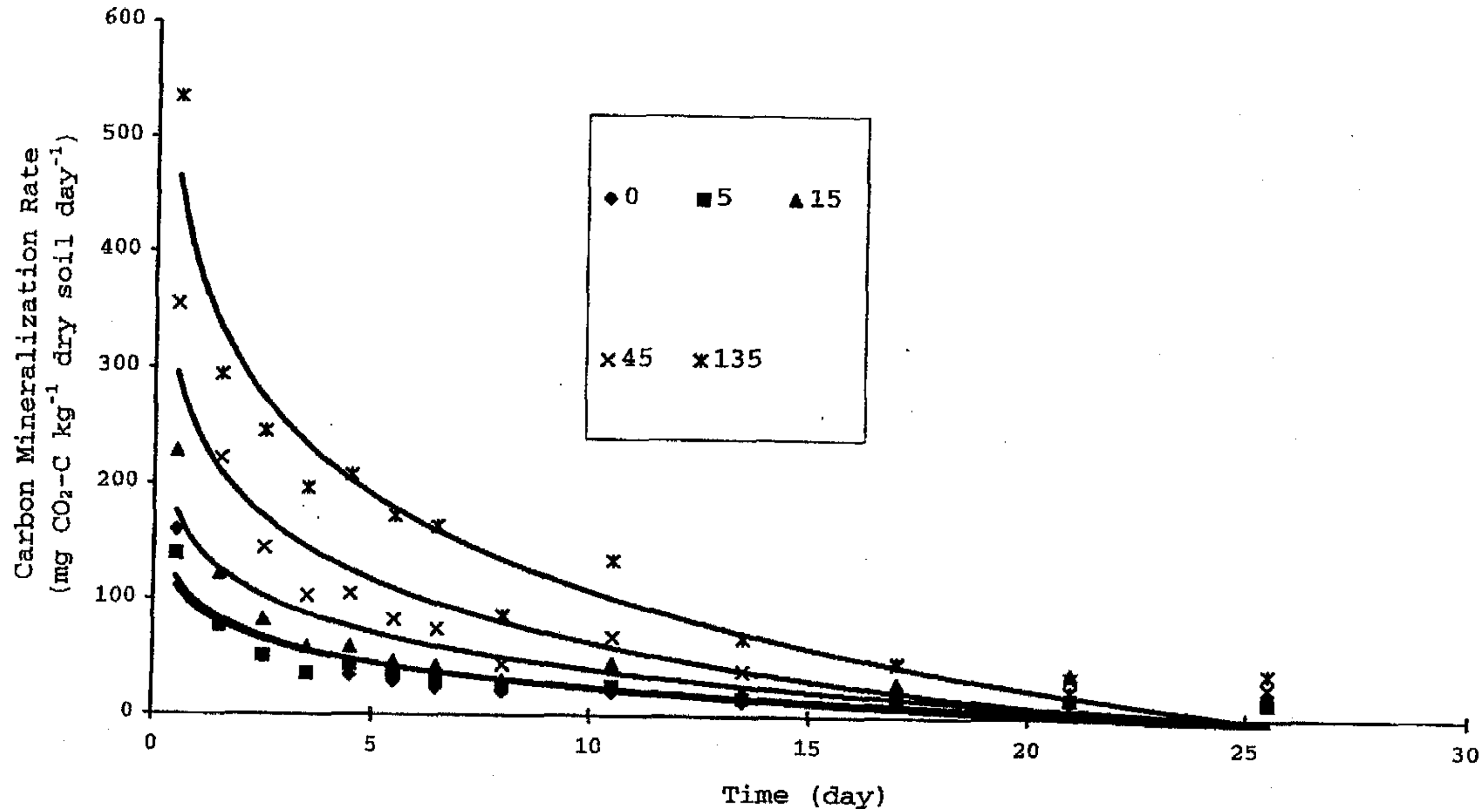
Unlike Gilmour et al. (1996b), the extrapolation of this function for time exceeding the period of incubation would have limited meaning. However, the purpose of this model wasn't to estimate total C mineralization for long term incubations. This model was developed because a) the statistical analysis couldn't be performed since the sampling days differed in the incubations, so it was done by using a and c values in a randomized block design; b) the greater the a, the greater the overall rate of C mineralization in the treatment. Therefore, the a values were used to estimate the sludge application rate maximizing the C mineralization rate.

The statistical analysis showed that sludge amendment significantly increased the absolute values of a and c ($P > .01$). Fly ash-LSR had no effect, nor was there an interaction between sludge and fly ash-LSR. The ANOVA table is given in Appendix 1.

Determination of Sludge Rate Maximizing Total C Mineralization Rate

An attempt was made to determine the sludge rate that maximized total C mineralization. The a values indicate how

Figure 30. Regression lines for the sludge treatments averaged across fly ash-LSR rates (Data from incubation 1).



quickly sludge will decompose in soil. An increase in a indicates that the overall rate of sludge decomposition increases. Based on this rationale, the mean a values for the sludge rates were pooled across fly ash-LSR rates and plotted against sludge application rates. This plot best fitted a second degree polynomial model.

$$y_a = -0.0041x^2 + 1.263x + 30.24 \quad (R^2 = 0.99)$$

where y_a = coefficient (a) and x= sludge rate (Mg ha^{-1}). The sludge application rate that maximized carbon mineralization rate was determined by taking the derivative of y_a and calculating (x) in the derivative formula where $y_a' = 0$.

$$Y_a' = -0.0082x + 1.263$$

$$0 = -0.0082x + 1.263$$

$$x = 154 \text{ Mg sludge ha}^{-1}.$$

CONCLUSIONS

Fly ash-LSR with a neutral pH and low salinity didn't affect C mineralization. Weathering probably neutralized alkalinity and lowered salinity in fly ash-LSR. Consequently, fly ash-LSR with a neutral pH and low salinity can be applied to soil at rates up to 135 Mg ha^{-1} without affecting C mineralization in soil amended with and without sludge.

This study determined that the 15 Mg ha^{-1} sludge treatment maximized the % sludge decomposition. However, more than 10 times as much sludge (154 Mg ha^{-1}) could be applied to reach a sludge application rate that maximized C mineralization.

Consequently, sludge at a rate of 154 Mg ha^{-1} could be applied to soil though there would be little reason to do so if the sludge is used to deliver additional agronomic nutrients. These high rates could create environmental problems such as NO_3^- contamination of groundwater via enhanced nitrification, if the rates of sludge N mineralization were rapid. Therefore, sludge N mineralization, a process which provides substrate for nitrification, is another concern in sludge management, even at low application rates. I addressed sludge N mineralization in the following chapter.

CHAPTER FOUR: NITROGEN MINERALIZATION IN SLUDGE AND FLY ASH-LSR TREATED SOIL

INTRODUCTION

Many studies have dealt with N mineralization in sludge treated soils. These studies showed that sludge N mineralization is a function of sludge and soil properties, and environmental conditions. They also noted that the greater the sludge rate, the higher the net N mineralization. However, this relationship may not hold true for high sludge rates.

Rapid sludge N mineralization produces NH_4^+, a substrate for nitrification. Nitrite is an intermediate product of nitrification. Usually, once NO_2^- is formed it is quickly oxidized to NO_3^-. However, Terry et al. (1981) reported a transient NO_2^- accumulation in a sludge treated soil, which is an indication of suppression in nitrification. The mechanism of this inhibition isn't well documented.

There have been only a few studies on N mineralization in soil treated with sludge and fresh fly ash-LSR combinations. Effects of weathered fly ash-LSR on N mineralization haven't been determined. To investigate this, I used fly ash-LSR rates of 0, 45, and 135 Mg ha^{-1}.

Consequently, this study addresses the following questions using sludge and fly ash-LSR application rates ranging from 0 to 270 Mg ha^{-1}.

1. Does fly ash-LSR affect sludge N mineralization in the amended soil?
2. Is there a synergistic effect of fly ash-LSR on sludge mineralization?

MATERIALS AND METHODS

Materials

The source, sampling dates, and handling of soil, sludge, and fly ash-LSR were the same as that for the C mineralization study.

Incubation Design

Air dried sludge and fly ash-LSR were added to soil according to a complete factorial design with two factors: sludge addition and fly ash-LSR addition. The sludge amendments were at rates of 0, 22.5, 45.0, 67.5, and 135.0 g kg^{-1} soil. The fly ash-LSR amendments were at rates of 0, 22.5, and 67.5 g kg^{-1} soil. The rates of 0, 22.5, 45.0, 67.5, and 135.0 g kg^{-1} soil correspond approximately to field application rates of 0, 45, 90, 135, and 270 Mg ha^{-1}, respectively. Field application rates will be used to describe the treatments.

Soil batches weighing 1050 g were mixed with sludge and fly ash-LSR at the chosen rates for 5 minutes in a mechanical mixer, placed in a plastic box, and amended with distilled water until each soil was at 20 % gravimetric water content. Then approximately 50 g of soil from each treatment was placed into each of 21 gas permeable plastic soil sampling bags. The bags were closed and incubated at 32 ± 3 ^{0}C for 45 days. The initial water content of the samples was maintained by spraying distilled water onto the soil samples in the bags.

Three bags were removed randomly from each treatment at 0, 2, 4, 6, 15, 30, and 45 days of incubation for NH_4^+-N, NO_2^--N, and NO_3^--N analysis. One molar potassium chloride (KCL) was used to extract exchangable inorganic nitrogen from the samples. Enough extractant (usually 150 mL) was poured into the bags to bring the extractant/soil ratio to 3:1 (w:w). The bags were shaken on a mechanical shaker at a slow speed for 30 minutes. Fifteen minutes after shaking, each sample was filtered through Whatman no. 1 filter paper and an aliquot was collected in an acid-washed scintillation vial. The vials were capped and stored in a freezer at 0 ^{0}C until analysis. The storage time was 1 to 3 months.

Ammonium Analysis

Ammonium concentrations were determined colorimetrically with a microplate reader after use of the Berthelot reaction (Kempers, 1974). When NH_4^+ reacts with alkaline phenol, hypochlorite, and nitroprusside, it forms a blue color characteristic of indophenol. The response of the microplate reader to the samples was measured by absorbance and compared to that of standards.

The samples were diluted 1:50 with distilled water using a digital diluter. Twenty µL from each sample and calibration standard were dispensed into the cells of a disposable microplate. One hundred µL of phenol-nitroprusside reagent (1.13 mL of 90 % liquefied phenol + 0.02 g sodium nitroprusside in 100 mL distilled water) was added to the microplate cells containing the samples, followed by 3 to 5 min shaking in a vortex mixer. One hundred µL of sodium hydroxide-hypochlorite reagent (0.5 g sodium hydroxide + 0.8 mL of 5.25 % sodium hypochlorite in 100 mL distilled water) was added to each cell. The microplate was shaken for 30 min. The plate was read for absorbance at 630 nm on the microplate auto-reader. The absorbance data were transferred to a computer that calculated NH_4^+-N concentrations.

Nitrite Analysis

Nitrite concentrations were determined colorimetrically by use of a microplate auto reader and the Griess reaction (Keeney and Nelson, 1982). A pink to red color is formed when nitrite reacts with an acidified solution of N-(1-Naphthyl)ethylenediamine dihydrochloride (NED) and sulfanilamide. The response of the microplate reader to samples of different intensity was measured and compared with those of standards.

The samples were diluted 1:50 with distilled water using a digital diluter. Two hundred µL from each sample and calibration standard were dispensed into the cells of a disposable microplate. Fifty µL of combined color reagent (100 mL 1.0 % sulfanilamide in 3.0 N HCl + 100 mL 0.1 % NED) was added to the cells containing the samples. The microplate was shaken in a vortex mixer for 5 min. The plate was read for absorbance at 540 nm on the microplate auto-reader. The absorbance data were transferred to a computer that calculated NO_2^--N concentrations.

Nitrate Analysis

Nitrate was analyzed using a Technicon auto-analyzer system 2. The samples were diluted 1:50 with distilled water by a digital diluter. Approximately 1 mL from each sample and standards were dispensed into 5 mL sample tubes. The absorbance measured by the auto-analyzer for the samples was recorded and compared to that of known concentrations.

A cadmium (Cd) column reduced NO_3^- to NO_2^- before analysis. Nitrite was analyzed colorimetrically at 540 nm using the Griess reaction as noted above (Keeney and Nelson, 1982). Nitrate content was taken to be the difference in NO_2^--N concentrations between the samples treated with and without the Cd reduction column.

Total Nitrogen Analysis

Total nitrogen analyses were performed for the samples at the beginning and end of the incubation. The samples were analyzed by the micro-Kjeldahl procedure KY (Bremner and

Mulvaney, 1982) at the College of Agriculture Regulatory Services Laboratory, at the University of Kentucky, Lexington.

Statistical Analysis

Statistical analysis of net N mineralization values was performed based on a completely randomized design. The treatment structure was factorial with 2 factors: sludge and fly ash-LSR rates.

Statistical analysis of NH_4^+-N, NO_2^--N, and NO_3^--N concentrations were analyzed according to a completely randomized design. This design had 3 treatment factors: 1. sludge application rate; 2. fly ash-LSR application rate; 3. time. The ANOVA tables are given in Appendix 2.

RESULTS AND DISCUSSION

Nitrogen Losses from the Samples

Total N contents of the treatments were lower at the end of the incubation than at the beginning of the incubation (Table 4). As sludge rate increased, on average, unrecovered N increased, rising from 40 mg N kg^{-1} soil in the treatments without sludge to 600 mg N kg^{-1} soil in the treatments with the highest sludge application rate (270 Mg ha^{-1}). The total N loss was proportional to the initial total N content and the proportion was about the same for all sludge treatments (Table 4). Most probably, N losses occurred via NH_3 volatilization and denitrification. Although N losses via these processes weren't measured, there was evidence that suggested both processes occurred during the incubation.

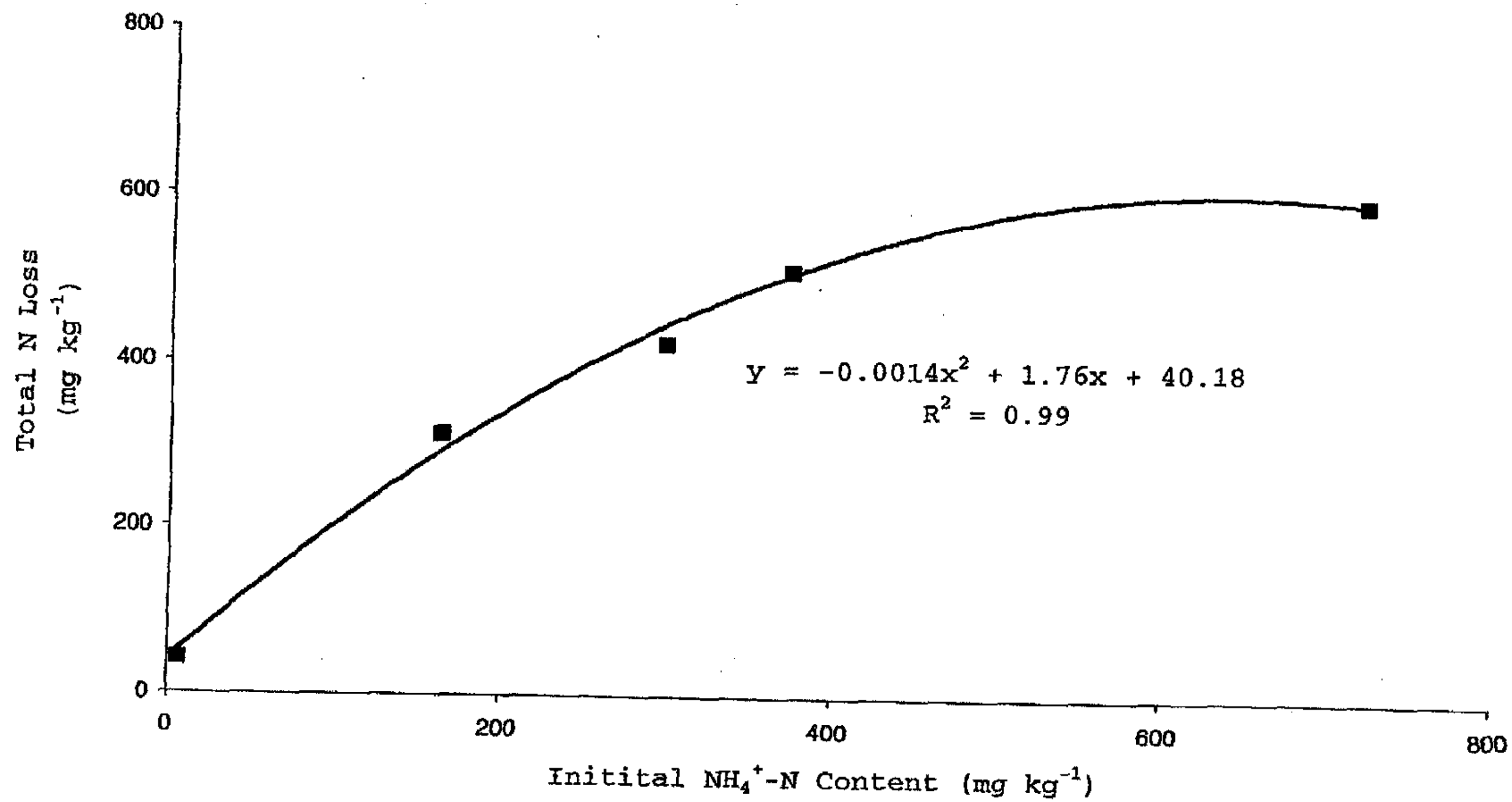
There are two reasons to suggest that part of the N loss occurred via NH_3 volatilization. First, the initial NH_4^+-N content was positively related to total N loss (Figure 31). In a neutral to alkaline environment, as occurred in this study, as NH_4^+-N content increases, NH_3 volatilization should increase. Second, NH_4^+-N contents declined in all sludge treatments that were averaged across fly ash-LSR treatments (Figures 33 to 36). The rate of NH_4^+ immobilization wasn't likely to exceed the rate of N mineralization, since there was net N mineralization in each

Table 4. The mean total nitrogen values at d =0 and d = 45[†].

Sludge Rate ($Mg\ ha^{-1}$)	Total-N, d =0	Total-N, d =45	Net Change	% Decrease
	$mg\ kg^{-1}$ soil			
0	2200 (160)[‡]	2150 (30)	-50	2
45	3200 (280)	2900 (180)	-300	10
90	3400 (160)	3000 (280)	-400	12
135	3800 (320)	3300 (180)	-500	13
270	5000 (610)	4300 (650)	-700	12

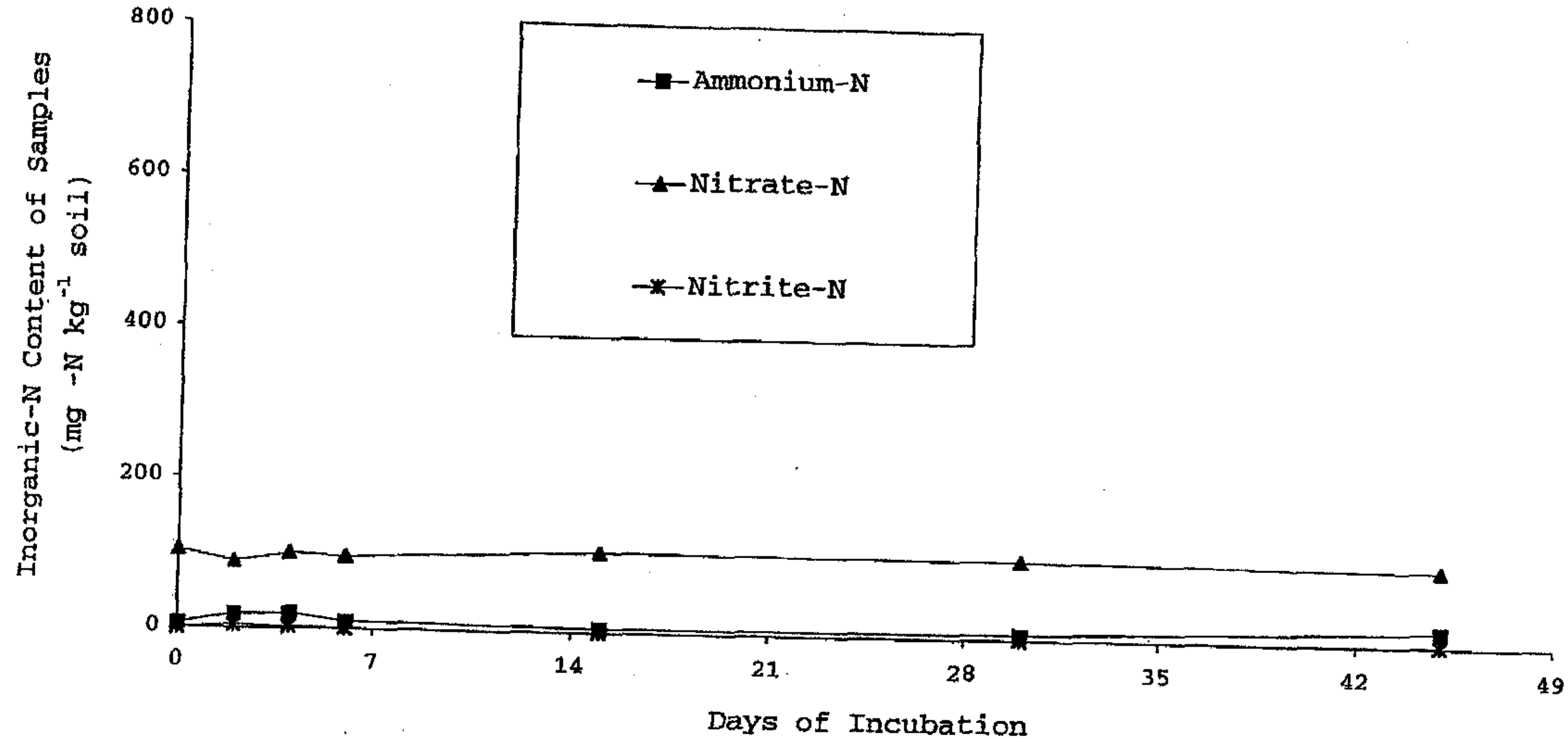
[†] The data are averaged across all fly ash-LSR rates. Each value represents the mean of 3 replications.
[‡] Standard deviations of the means.

Figure 31. The relationship between initial NH_4^+-N content and total N loss†.



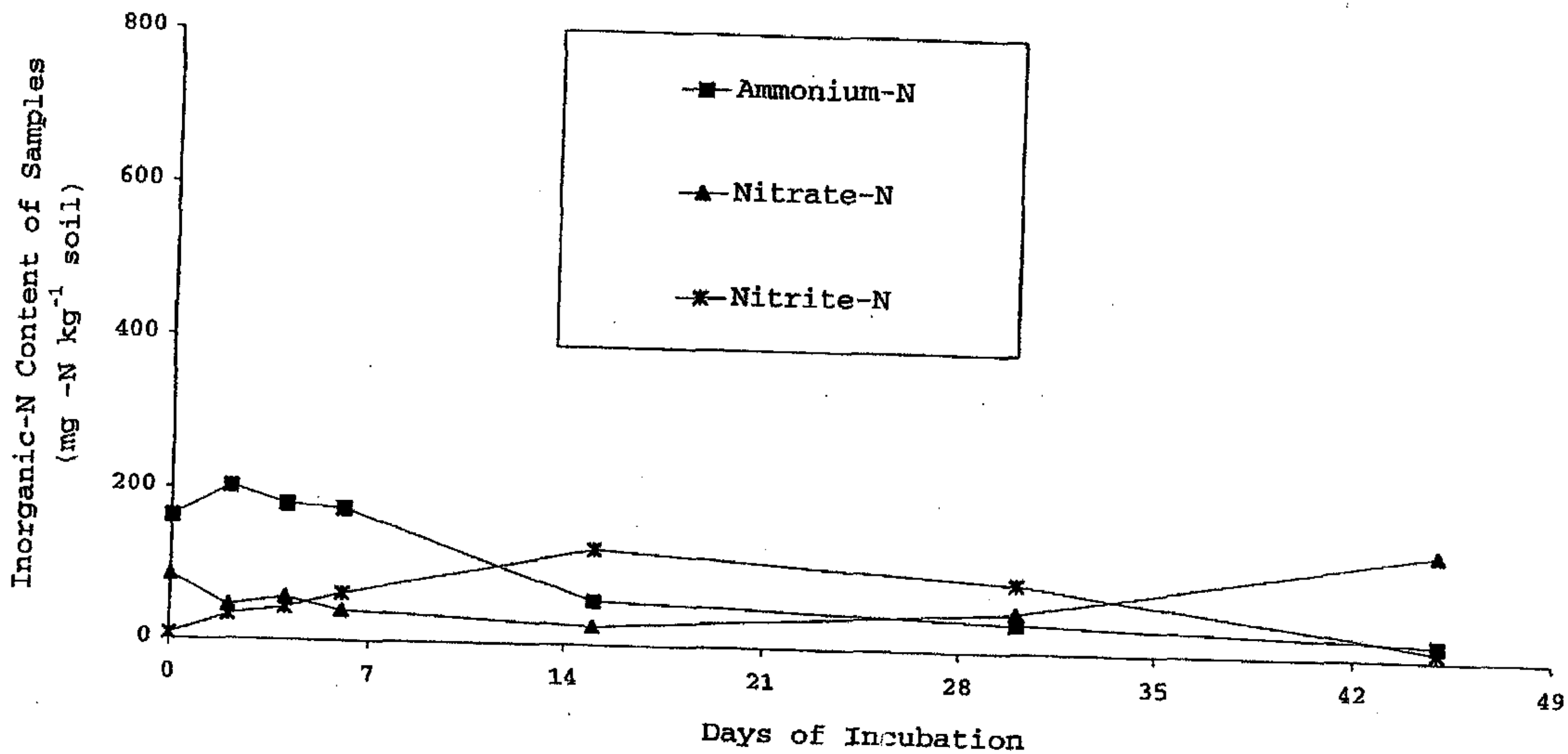
† Data are averaged across fly ash-LSR rates. Each point represents the mean of 3 replications.

Figure 32. The relationship between mean inorganic nitrogen content and time in the 0 Mg ha^{-1} sludge treatment†.



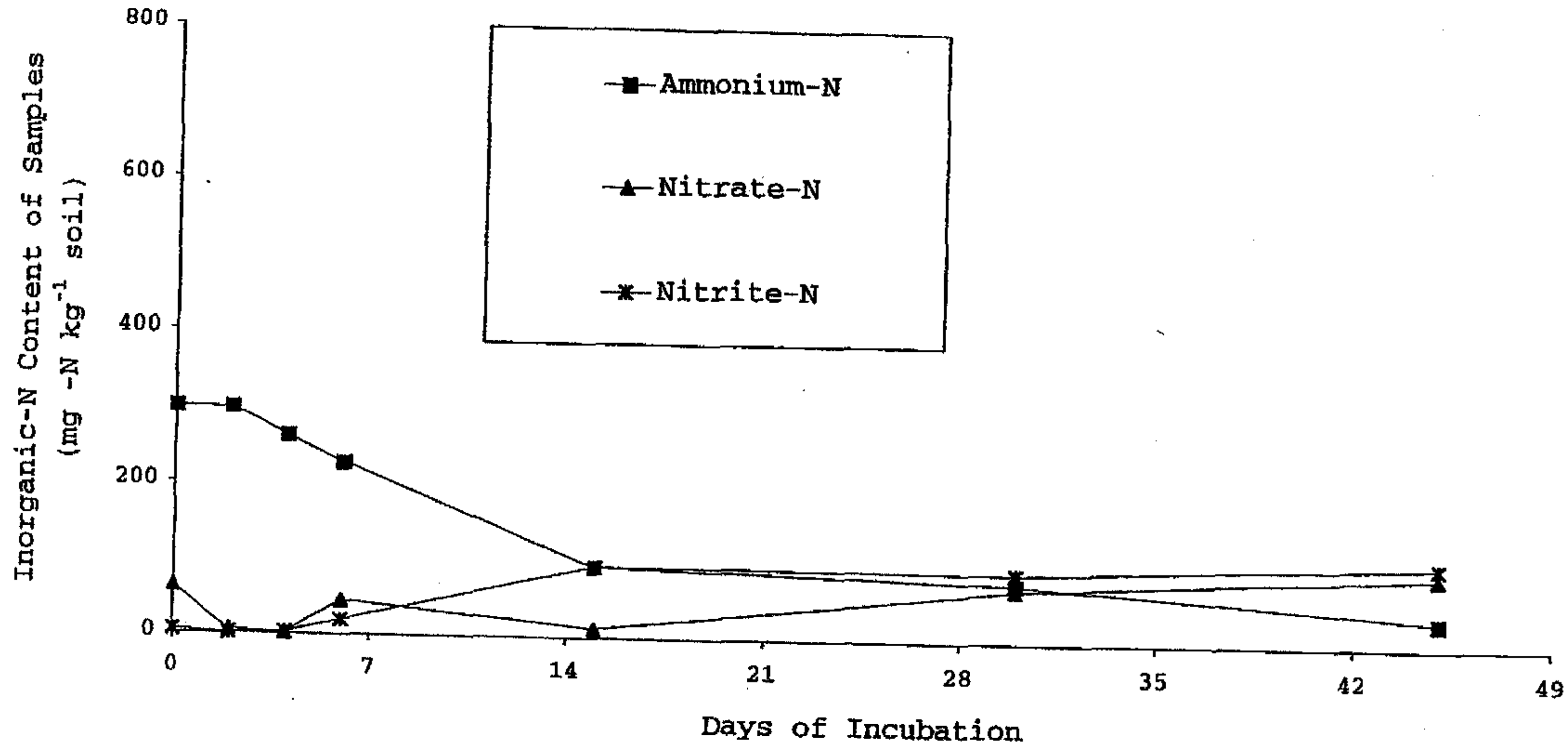
† Data are averaged across fly ash-LSR rates. Each point represents the mean of 3 replications.

Figure 33. The relationship between mean inorganic nitrogen content and time in the 45 Mg ha^{-1} sludge treatment†.



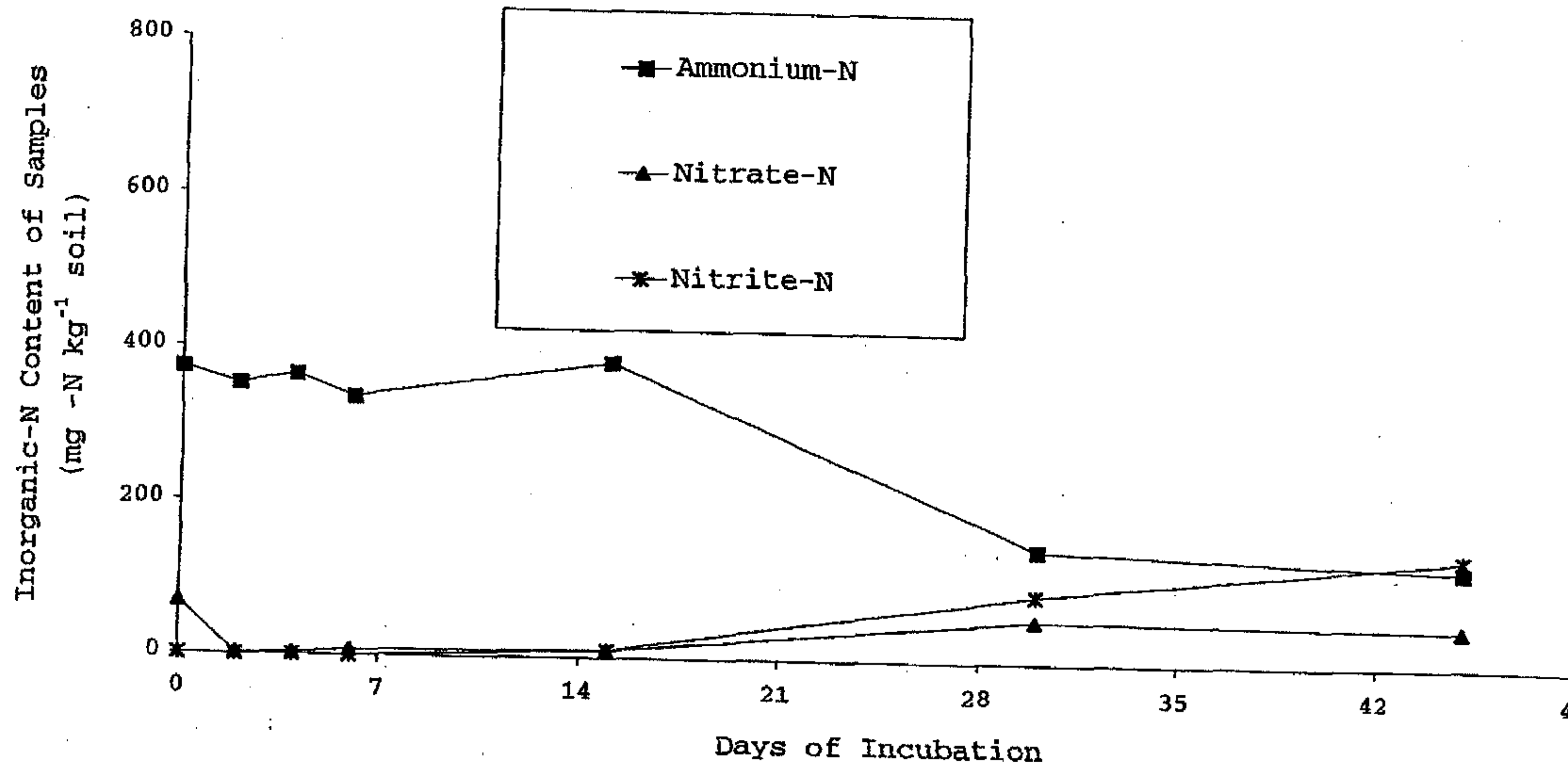
† Data are averaged across fly ash-LSR rates. Each point represents the mean of 3 replications.

Figure 34. The relationship between mean inorganic nitrogen content and time in the 90 Mg ha^{-1} sludge treatment†.



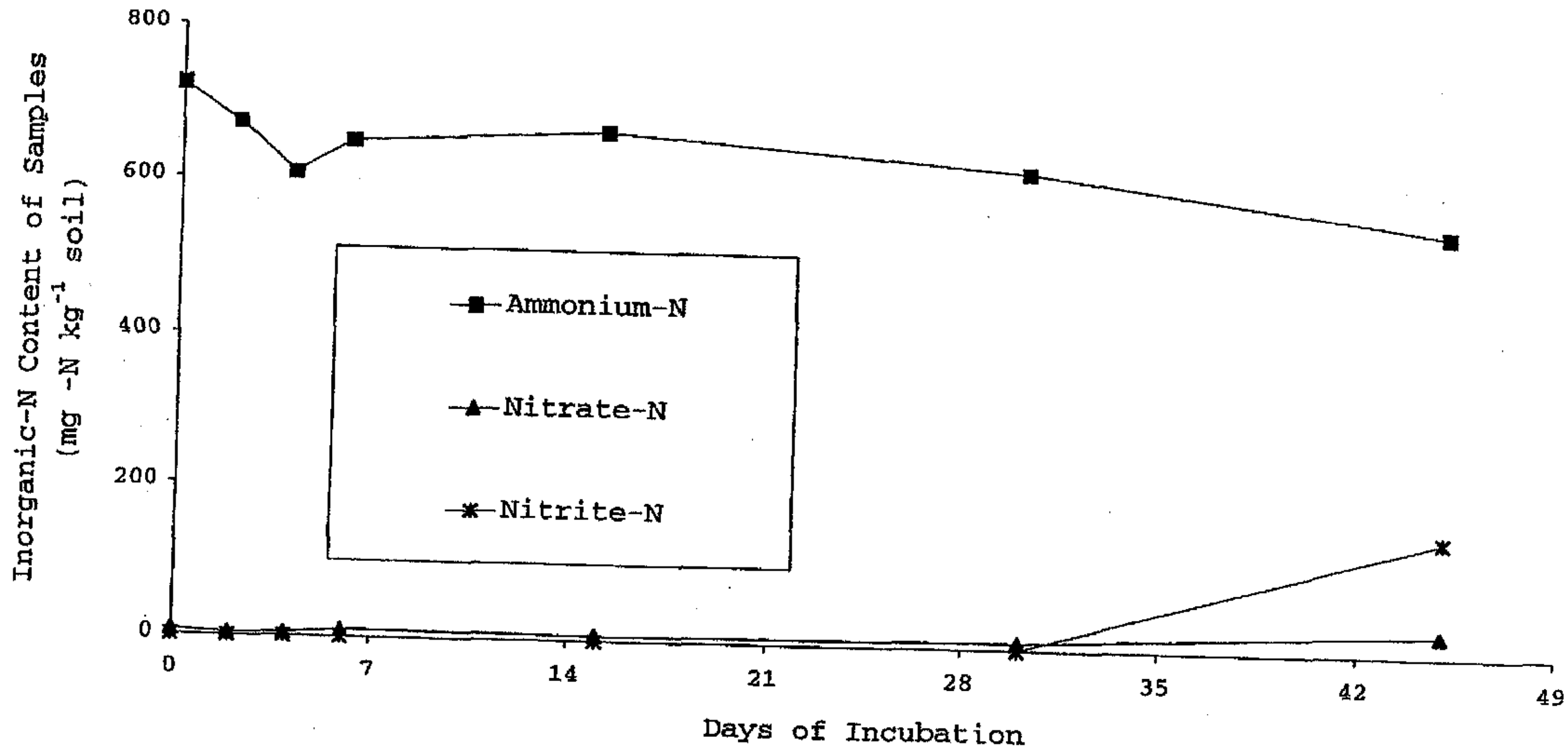
† Data are averaged across fly ash-LSR rates. Each point represents the mean of 3 replications.

Figure 35. The relationship between mean inorganic nitrogen content and time in the 135 Mg ha^{-1} sludge treatment[†].



[†] Data are averaged across fly ash-LSR rates. Each point represents the mean of 3 replications.

Figure 36. The relationship between mean inorganic nitrogen content and time in the 270 Mg ha^{-1} sludge treatment[†].



[†] Data are averaged across fly ash-LSR rates. Each point represents the mean of 3 replications.

treatment. This indicates that net N immobilization didn't occur during the incubation.

Only nitrification could account for the decrease in NH_4^+-N content, if NH_3 volatilization was assumed absent. But, the net increase in NO_2^--N + NO_3^--N accounts for only 39, 49, and 64 % of the net decrease in NH_4^+-N content in the 45, 90, and 135 Mg ha^{-1} sludge treatments, respectively. This percentage was 100 % in the 270 Mg ha^{-1} sludge treatment, suggesting that there was no NH_3 volatilization in this treatment. However, the total N measurement showed that the highest N loss occurred in the 270 Mg ha^{-1} sludge treatment (Table 4). Some of the NH_3-N lost via volatilization could have been replenished by mineralized-N in this treatment. This was obvious in that the NH_4^+-N concentration dropped initially, then increased somewhat (Figure 36).

Volatilization of NH_3 is accompanied by an increase in soil pH, but the increase in soil pH doesn't necessarily have to be great for high NH_3 volatilization. For example, Fine et al. (1989) determined that a rise of 0.6 pH unit increase in a neutral soil, caused approximately 40 % of the sludge mineralized-N to be lost via volatilization. Sludges with high NH_4^+-N content and mineralizable organic N, like the one used by Fine et al. (1989), are subject to great NH_3 volatilization in sludge-amended soil (Premi and Cornfield, 1969; Ryan and Keeney, 1975; Terry et al., 1978). As the application rate of such sludges increases, volatilization of NH_3 in amended soil increases (Fine et al., 1989).

Denmead et al. (1982) suggested that one can determine NH_3 concentration in the soil solution by the following equation.

$$[NH_3]_{solution} = [NH_3 + NH_4^+]_{solution} / 1 + 10^{(0.09 + 2730/T - pH)}$$

Where the concentrations of NH_3 and NH_4^+ are in g/m^3, and temperature (T) is in ^0K. In my study, temperature was 35 ^0C (308 ^0K), and initial NH_4^+-N concentration in the 270 Mg ha^{-1} sludge treatment was 720 mg kg^{-1} (3600 g NH_4^+-N/m^3 in solution). The only unknown in this equation is soil pH. In the 270 Mg ha^{-1} treatment, NH_3-N concentration in solution was 70 mg kg^{-1} if soil pH was 8.0, it was 260 mg kg^{-1} if soil pH is 8.5, and 750 mg kg^{-1} if soil pH was 9.0. To account for most of the total N loss in the 270 Mg ha^{-1}, soil pG should be higher than 8.0.

Another N loss from the 45, 90, and 135 Mg ha^{-1} sludge treatments was decrease in the initial NO_3^--N concentrations (Figures 33 to 36). On average, the NO_3^--N concentration decreased 70 mg kg^{-1} in these treatments. Soil was the major source of NO_3^- in each treatment. The soil had been stored at field moisture in the greenhouse for a year prior to incubation, which led to nitrification.

The 270 Mg ha^{-1} sludge treatment had the lowest initial NO_3^--N concentration, compared to other sludge treatments (Figures 33 to 36). But, the 270 Mg ha^{-1} sludge treatment should have had the highest initial NO_3^--N concentration since the sludge contained more NO_3^--N than soil (Table 1). In other words, the more the sludge is added, the higher the NO_3^--N concentration in a treatment. This could be an artifact of the NO_3^--N analysis. An attempt to analyze NO_3^--N in the treatments failed because compound(s), probably contained in the sludge, interfered with the cadmium reduction column. The samples had to be diluted to measure NO_3^--N concentrations again. But, even then, the initial NO_3^--N levels were lowest in the 270 Mg ha^{-1} sludge treatment.

The decrease in initial NO_3^--N in the 45, 90, and 135 Mg ha^{-1} sludge treatments can only be explained by denitrification. High NH_4^+-N concentrations during the incubation probably suppressed NO_3^--N immobilization (Betlach et al., 1981). Dissimilatory nitrate reduction to ammonium wasn't likely since NH_4^+-N concentrations in the treatments, with the exception of 45 Mg ha^{-1} sludge treatment at day 2, didn't increase as NO_3^--N concentrations decreased (Figures 33 to 36).

The rapid initial O_2 consumption measured in the C mineralization study suggested that anaerobic microsites were created in soil, promoting denitrification. In this study, approximately 70 mg NO_3^-–N kg^{-1} was lost, presumably via denitrification, in the 90 and 135 Mg ha^{-1} sludge treatments within a few days (Figures 35 to 36). Less than this amount was lost in the 45 Mg ha^{-1} sludge treatment within 2 weeks (Figure 33). The difference in time required to denitrify approximately the same amount of NO_3^-–N supports my microsite suggestion. As sludge rate increased, the initial O_2 consumption and, therefore, the number of microsites increased. The greater the number of microsites, the higher the rate of denitrification should be in the soil.

Most probably, denitrification didn't occur later when the substrate C consumption rate decreased (Figures 27 to 29), NO_3^-–N was either absent or was increasing (Figures 33 to 36), and O_2 concentrations were relatively high (data not shown). Therefore, denitrification should only have happened earlier, and this would account for 22, 16, and 13 % of the total N losses in the 45, 90, and 135 Mg ha^{-1} sludge treatments.

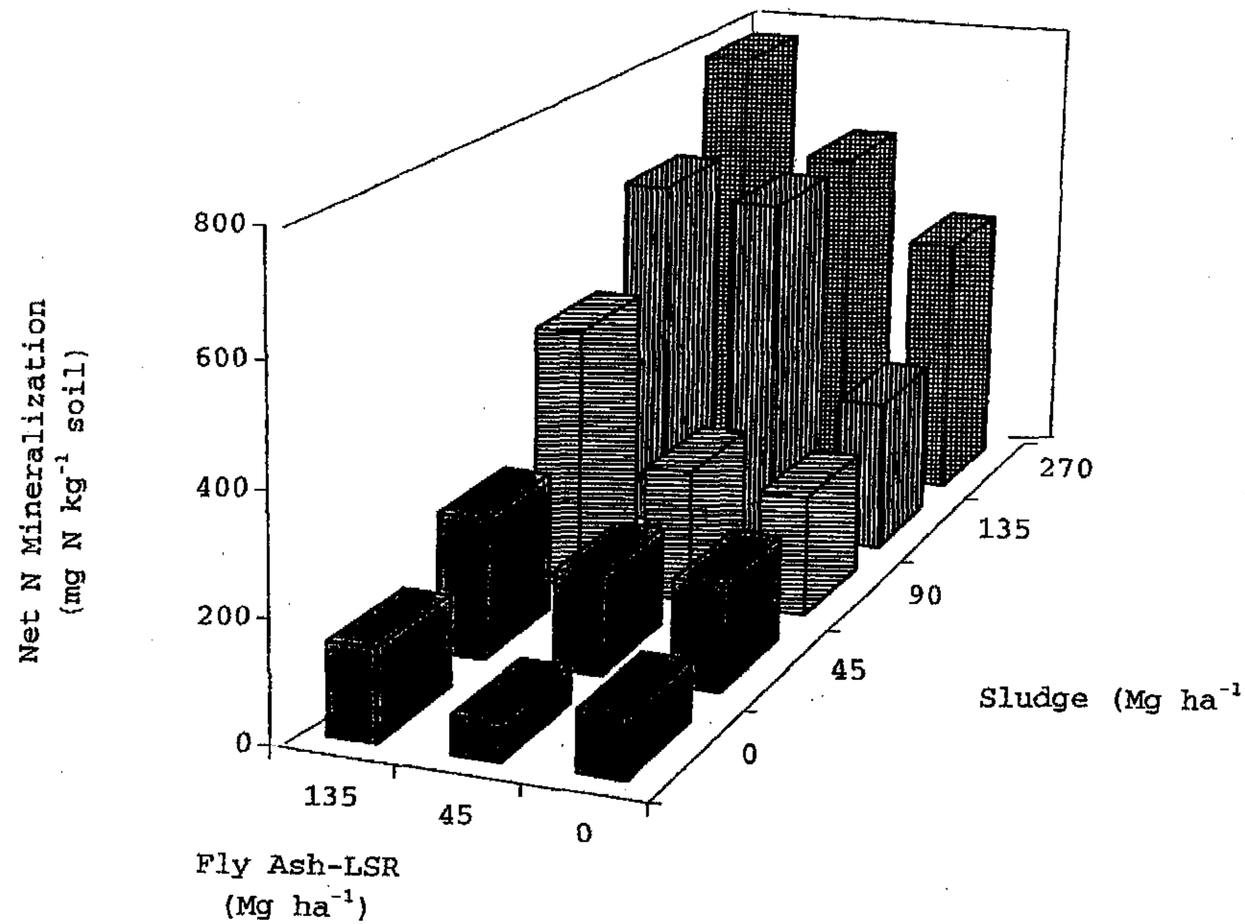
Net N Mineralization

Hart et al. (1994) noted that net change in inorganic N content during an incubation reflects net N mineralization. This approach is most valid in the absence of plant assimilation, leaching, denitrification, and NH_3 volatilization. In the presence of these processes, the net change in the inorganic N pool will underestimate net N mineralization.

In my study, nitrogen losses from soil by plant assimilation and leaching were absent. However, NH_3 volatilization and denitrification likely caused significant N losses, although these weren't directly measured. Therefore, I calculated net N mineralization in each treatment as the difference in organic N content between the first and last sample period. The organic N content of each treatment was calculated as the difference between the total N content and the total inorganic N content of that treatment.

Net N mineralization in the treatments for the 45 d incubation are shown in Figure 37. As sludge rate increased, on average, net N mineralization increased, rising from 80 mg N kg^{-1} soil in the treatments without sludge to 530 mg N kg^{-1} soil in the treatments with the highest sludge application rate (270 Mg ha^{-1}). As fly ash-LSR rate increased, on average, net N mineralization also increased, rising from 240 mg N kg^{-1} soil in the treatments without fly ash-LSR to 440 mg N kg^{-1} soil in the treatments with the highest rate of fly ash-LSR (135 Mg ha^{-1}). The increase in net N mineralization, as sludge or fly ash-LSR application rate increased, was significant ($p > .05$).

Figure 37. Net N Mineralization in sludge and fly ash-LSR treated soils during a 45 d incubation (Mean of 3 replications).



Net N mineralization results from my study are less than those of Hsieh et al. (1981b) and Fine et al. (1989) (Figure 38). Hsieh et al. (1981b) and Fine et al. (1989), like the current study, used aerobically digested sludges and measured net N mineralization as the net change in organic N content. Fine et al. (1989) used a sludge with an organic N content of 5.5 %, Hsieh et al. (1981b) used a sludge with an organic N content of 4.7 %, and my study used a sludge with an organic N content of 2.7 %. Since the soils used in these studies were similar in pH and organic N content, the differences in net N mineralization are attributable to sludge organic N content. In other words, the larger sludge organic N content, the higher the net N mineralization. Previous studies have shown that as the organic N content of sludge increases, total N mineralization increases (Figure 15). The organic N content of sludge is probably the most important factor determining the rate and extent of net N mineralization.

Unlike net N mineralization, the % organic sludge N mineralized decreased as sludge application rate increased. These percentages were 32, 28, 25, and 23 % in the 45, 90, 135, and 270 Mg ha^{-1} sludge treatments, respectively. Comparable studies had similar % organic sludge N mineralized (Figure 38). Fine et al. (1989) had the highest % sludge N mineralization, probably because their sludge had

Figure 38. A comparision of the % N mineralization from sludge in the current study; Fine et al. (1989); Hsieh et al. (1981b).

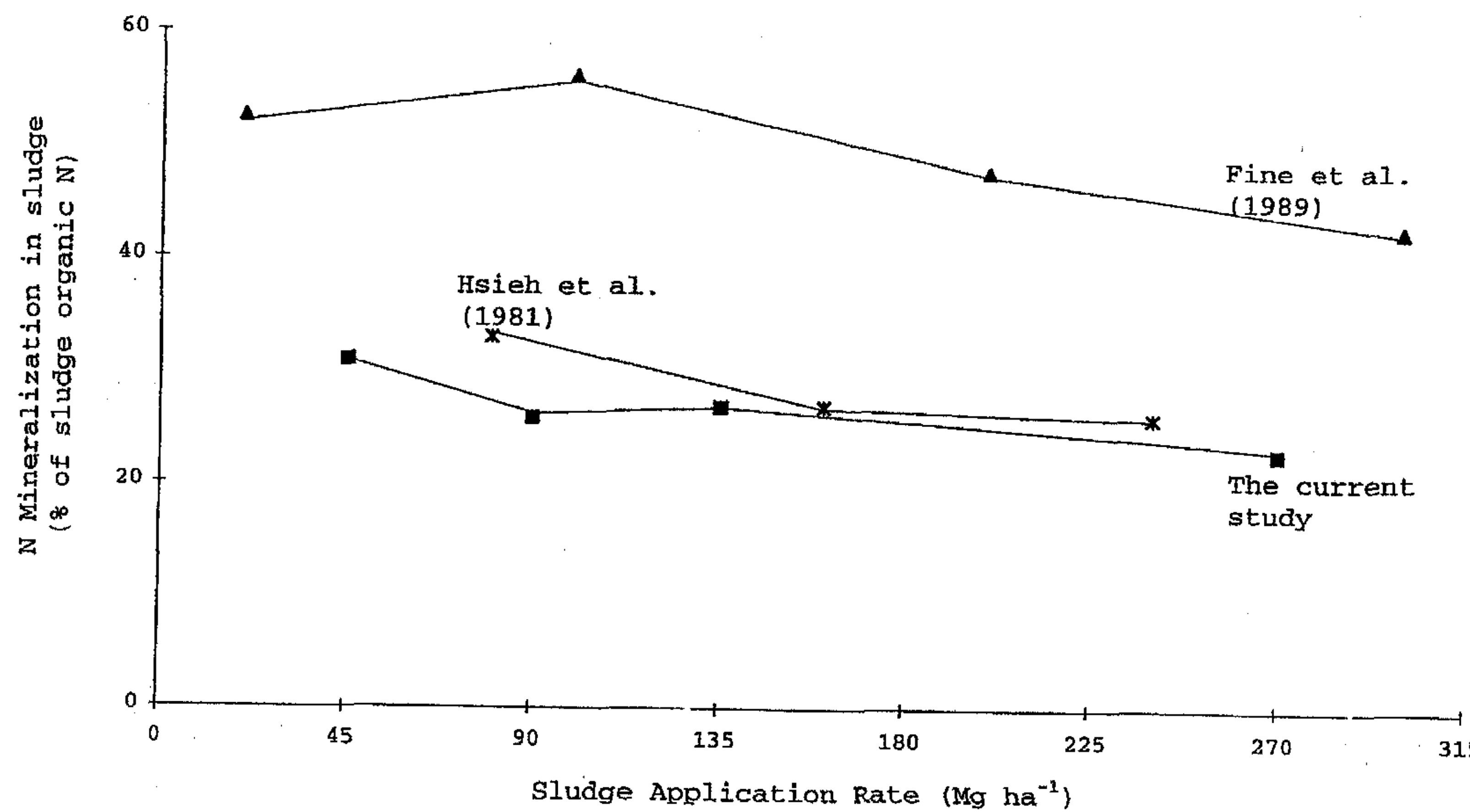
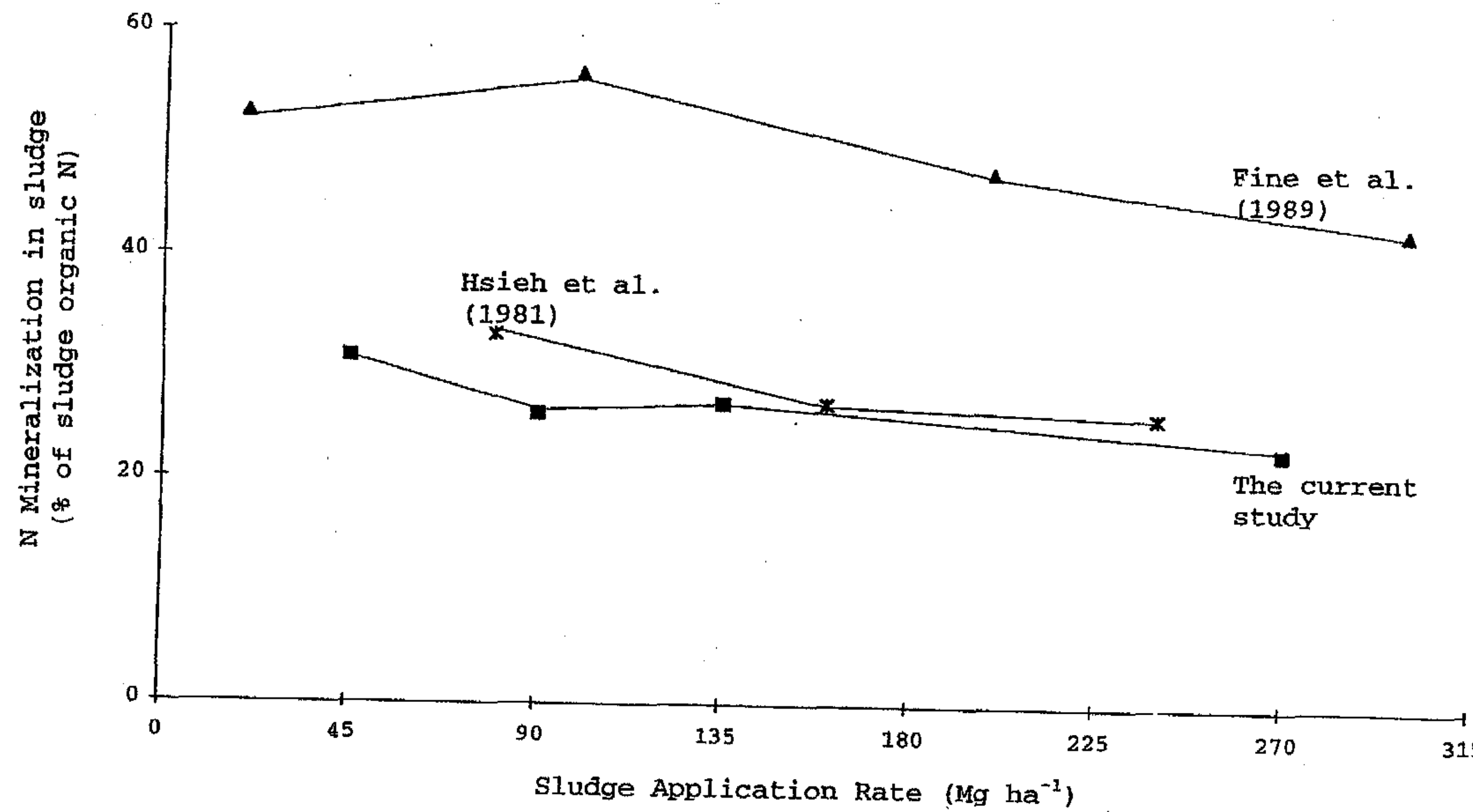


Figure 39. A comparision of the % N mineralization from sludge in the current study; Fine et al. (1989); Hsieh et al. (1981b).



the highest labile organic N content. Sludge properties vary greatly, based on the degree of industrialization in the sanitary district, and the wastewater entering the sludge treatment plant. Consequently, sludges might differ in organic composition, although they were treated with the same method, and this might yield different % sludge N mineralization.

Like sludge, fly ash-LSR increased net N mineralization in the treatments (Figure 38). Since, the fly ash-LSR doesn't contain organic N, it shouldn't increase net N mineralization but for providing a better mineralizing environment. Cervelli et al. (1987) found that an alkaline fly ash-LSR neutralized an acid soil, optimizing pH for microbes, and thereby increased net N mineralization. The current study used soil, sludge, and fly ash-LSR having near neutral pH values (Table 3). Therefore, acid neutralization by fly ash-LSR can't explain the effect I observed.

Fly ash-LSR may also improve soil physical properties such as texture (Chang et al., 1977) and aeration (Aitken and Bell, 1985: Chang et al., 1977). However, the effects of such improvements on N mineralization weren't examined by either my study or previous studies.

The reliability of total N analysis was investigated by estimating the total N contents at the beginning of the incubation and comparing them with the measured values. The total N contents on day 0 were estimated by using total N contents of soil and sludge, and the ratio of sludge:soil (w:w) in the treatments. The % difference between estimated and measured total N in the treatments are given in Table 5. As fly ash-LSR rate increased, the % difference between estimated and measured total N increased. Fly ash-LSR might have interfered with the total N analysis, resulting in biased measurements of total N content. Net N mineralization values were calculated by the use of total N contents.

My carbon mineralization study showed that fly ash-LSR has no effect on C mineralization in the amended soil. Carbon and N mineralization are usually related (Gilmour et al. 1985). The fly ash-LSR, therefore, also shouldn't affect N mineralization.

Concentrations of Inorganic-N Species During the Incubation

Statistical analysis for the relationship between time and mean concentrations of NH_4^+-N, NO_2^--N, and NO_3^--N in the sludge treatments showed that all treatment factors (sludge, fly ash-LSR, and time), and interactions among these treatment factors, were significant ($P > .05$). Although the fly ash-LSR effect was significant, there wasn't a trend between time and inorganic N content in the fly ash-LSR treatments. Consequently, only the relationship between time and inorganic N contents in the sludge treatments averaged across fly ash-LSR treatments will be discussed. Inorganic N

Table 5. The % difference between estimated and measured total N values in the treatments at day 0[†].

Fly Ash-LSR Rate (Mg ha^{-1})	Sludge Application Rate (Mg ha^{-1}) 0	45	90	135	270
0	0.9	-0.3	-0.9	2.8	8.3
	(2200-2180)[‡]	(2850-2860)	(3500-3530)	(4150-4050)	(6100-5600)
45	2.7	-1.6	3.2	0.7	28.2
	(2200-2140)	(2850-2900)	(3500-3390)	(4150-4130)	(6100-4390)
135	4.6	0.7	8.0	16.2	21.2
	(2200-2100)	(2850-2830)	(3500-3220)	(4150-3480)	(6100-4820)

[†] The % difference = (estimated total N - measured total N) * 100 / estimated total N.

[‡] (Estimated-Measured).

contents in the 0, 45, 90, 135, and 270 Mg ha^{-1} sludge treatments with respect to time are given in Figures 32, 33, 34, 35, and 36, respectively.

As sludge application rate increased, the initial NH_4^+-N content in the treatments increased, rising from 10 mg kg^{-1} soil in the treatments without sludge to 720 mg kg^{-1} soil in the treatments with the highest sludge application rate (270 Mg ha^{-1}). However, as the sludge rate increased, NH_4^+ oxidation (NO_2^--N formation) was delayed (Figures 33 to 36).

The delay in NO_2^- formation indicates that NH_4^+-oxidizer populations were suppressed possibly due to the high substrate concentrations and salt contents. A lag phase in NH_4^+-oxidation due to high substrate concentrations has been shown in soil (Broadbent et al., 1957; Harada and Kai, 1968; Justice and Smith, 1962; Malhi and McGill, 1982; Mclntosh and Frederick, 1958; Molina, 1985; Morril and Dawson, 1967; Stojanovic and Alexander, 1958). Most of these studies suggested that NH_4^+-N concentrations exceeding 300 mg kg^{-1} cause NH_3 volatilization in soil and NH_3, not NH_4^+ inhibits NH_4^+-oxidizers.

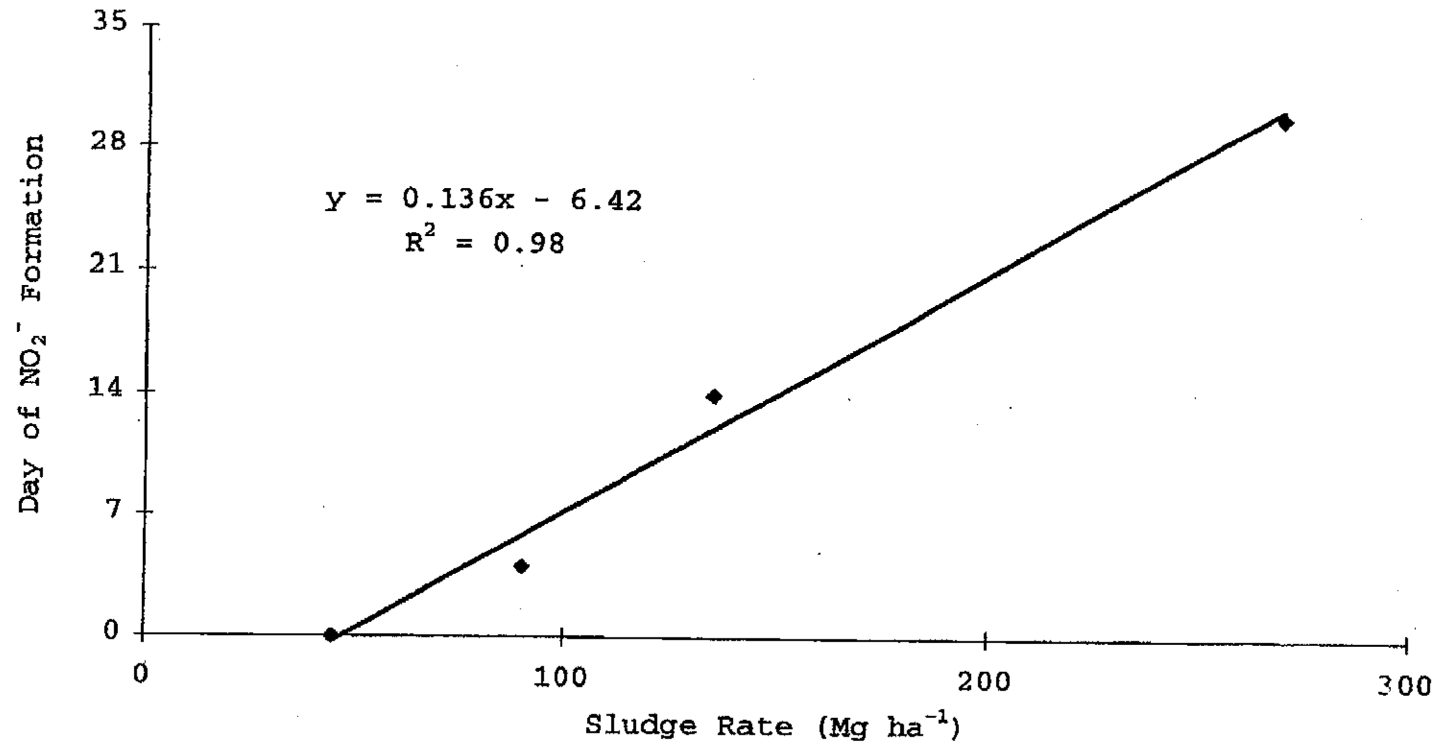
In the current experiment, with the exception of the 45 Mg ha^{-1} sludge treatment, all sludge treatments had initial NH_4^+-N concentrations above 300 mg NH_4^+-N kg^{-1} soil. There was a delay in NO_2^--N formation in all amendment rates except in

45 Mg ha^{-1} (Figures 34 to 36). The delay in NH_4^+-oxidation (NO_2^- formation) increased, as sludge application rate increased (Figure 40). This is one important piece of evidence that suggests inhibition of NH_4^+-oxidation was related to NH_3. Fine et al. (1989) determined that as sludge application rate increased, the duration and rate of NH_3 release from samples increased (Figure 41).

Like high NH_3-N concentrations, extreme salt contents also suppress NH_4^+-oxidation (Harada and Kai, 1968; Laura, 1977; Malhi and McGill, 1982). In the current study, the salinity of sludge was much higher than that of fly ash-LSR and soil (Table 1). As sludge application rate increased, salinity probably increased although this wasn't determined. There might also be a synergy between NH_3 and salt contents on the inhibition of NH_4^+-oxidation (Harada and Kai, 1968). In other words, these two factors could interact so that the net effect would be greater than that expected from the independent effects of each factor.

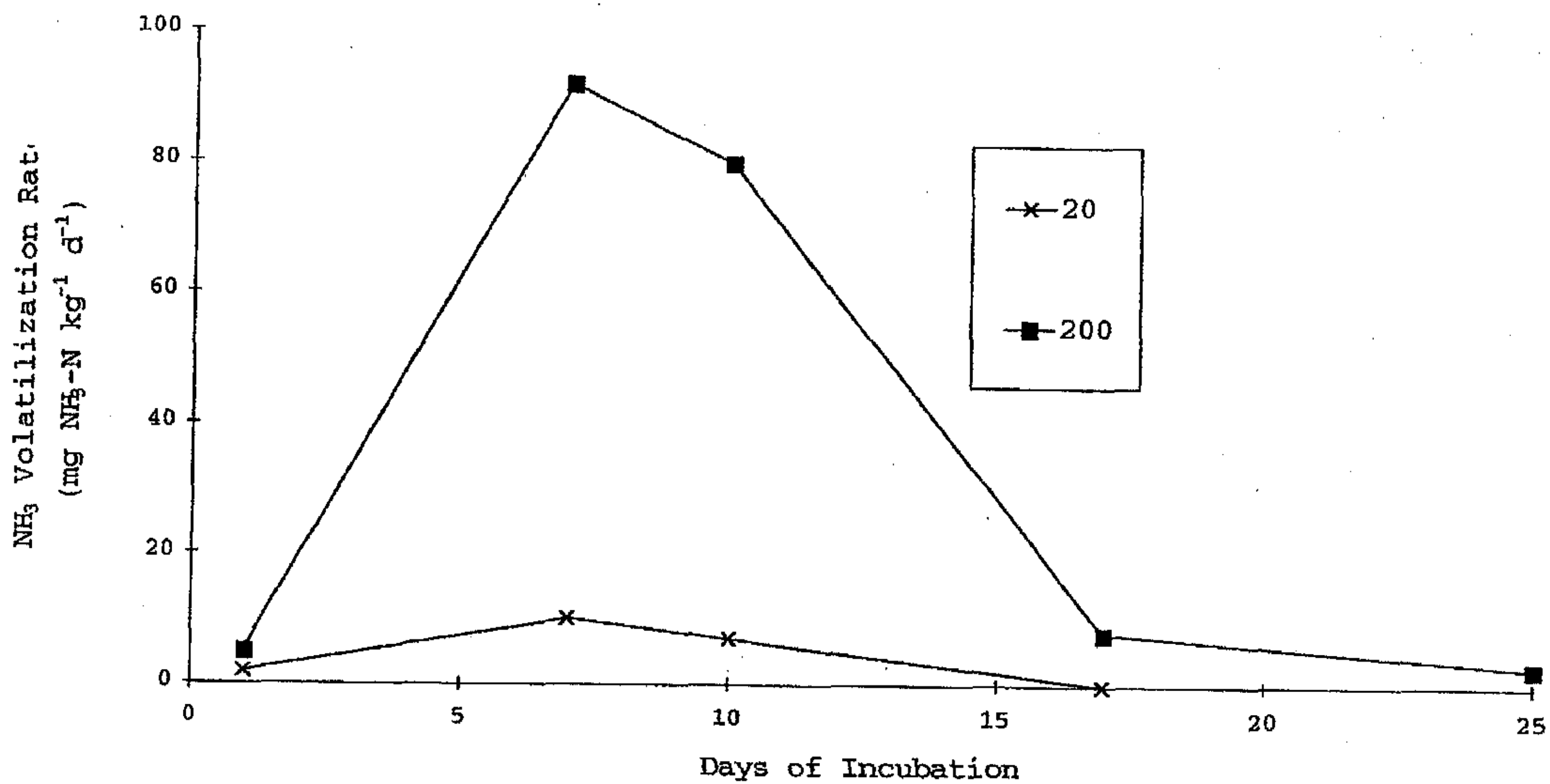
There was also an inhibition in NO_2^--oxidation (NO_3^- formation). Nitrite transiently accumulated in the treatments (Figures 33 to 36). Usually, NO_2^- doesn't accumulate in soil since the rate limiting step in nitrification is NH_4^+-oxidation. In other words, NO_2^--oxidation is more rapid than NH_4^+-oxidation and once NO_2^- is formed, it is quickly oxidized.

Figure 40. Delay in nitrification with respect to the amount of sludge added (Data are averaged across fly ash-LSR rates).



† Data are averaged across fly ash-LSR rates. Each point represents the mean of 3 replications.

Figure 41. The relationship between NH_3 volatilization in sludge treated soils and days of incubation.



† Data from Fine et al. (1989). Sludge rates are given as Mg ha^{-1}.

Nitrite is toxic to plant roots even at low concentrations such as 5 mg kg^{-1} (Hamilton and Lowe, 1981). More importantly, NO_2^- contaminates groundwater and causes blue baby syndrome (methemoglobnemia) in babies drinking NO_2^- contaminated water.

CONCLUSIONS

This study determined that fly ash-LSR increased net N mineralization but not carbon mineralization. Since C and N mineralization values are usually linked, fly ash-LSR also shouldn't affect N mineralization.

Aerobically digested sludges may be characterized by their high inorganic and/or organic N content, and a low C:N ratio. Volatile N losses from soil amended with such sludges have been great. These losses should be accounted for when determining sludge-N availability for plant growth.

The sludge amended soils also accumulated NO_2^- at sufficient concentrations to inhibit plant root growth.

GENERAL CONCLUSIONS

This study showed that sludge affects C mineralization in the amended soil. Sludge rate maximizing total C mineralization was 154 Mg ha^{-1}. Sludge rate maximizing % sludge C mineralizaiton was 15 Mg ha^{-1}. But, this was probably an artifact of high background soil C. Therefore, as sludge application decreases, % C mineralization in sludge should decrease. There wasn't a fly ash-LSR effect probably because fly ash-LSR used was neutral and had low salts compared to unweathered fly ash-LSR.

Fly ash-LSR had an effect on net N mineralization. But, this was probably an experimental error. Fly ash-LSR shouldn't affect N mineralization since C and N mineralization are linked and there wasn't a fly ash effect on C mineralization.

Appendix 1. ANOVA Tables for C mineralization studies.

Appendix 1.a. ANOVA Table for total C mineralization.

Source of Variation	dF	F Value	$P_r > F$
Incubation (Block)	2	3.3	0.047
Fly Ash-LSR	4	0.2	0.895
Sludge	4	90.3	0.001
Fly Ash-LSR × Sludge	16	1.3	0.222

Appendix 1.b. ANOVA Table for rate constants (a) of C mineralization.

Source of Variation	dF	F Value	$P_r > F$
Incubation (Block)	2	12.3	0.001
Fly Ash-LSR	4	0.2	0.916
Sludge	4	122.6	0.001
Fly Ash-LSR × Sludge	16	1.0	0.392

Appendix 2. ANOVA Tables for N mineralization study.

Appendix 2.a. ANOVA Table for net N mineralization.

Source of Variation	dF	F Value	$P_r > F$
Fly Ash-LSR	2	17.2	0.001
Sludge	4	43.8	0.001
Fly Ash-LSR × Sludge	8	2.8	0.020

Appendix 2.b. ANOVA Table for NH_4^+-N, NO_2^--N, and NO_3^--N concentrations†.

Source of Variation	dF
Sludge	4
Fly Ash-LSR	2
Sludge × Fly Ash-LSR	8
Time	6
Time × Sludge	24
Time × Fly Ash-LSR	12
Time × Sludge × Fly Ash-LSR	48

† Only degrees of freedom were given since all inorganic N species for each source of variation were significant ($P > 0.01$).

REFERENCES

Adriano, D., A. Page, A. Elseewi and A. Chang. 1982. Cadmium availability to sudangrass grown on soils amended with sewage sludge and fly ash. J. Environ. Qual. 11:197-203.

Adriano, D., A. Page, A. Elseewi, A. Chang, and I. Straughan. 1980. Utilization and disposal of fly ash and other coal residues in terrestrial ecosystems: A review. J. Environ. Qual. 9:333-334.

Adriano, D., T. Woodford, and T. Ciravolo. 1978. Growth and elemental composition of corn and bean seedlings as influenced by soil application of fly ash. J. Environ. Qual. 7:416-421.

Agbim, N., B. Sabey, and D. Markstrom. 1977. Land application of sewage sludge: V. Carbon dioxide production as influenced by sewage sludge and wood waste mixtures. J. Environ. Qual. 6:446-451.

Aitken, R. and L. Bell. 1985. Plant uptake and phytotoxicity of boron in Australian fly ashes. Plant Soil. 84:245-257.

Arthur, M., T. Zwick, D. Tolle, and P. Van Voris. 1984. Effects of fly ash on microbial CO_2 evolution from an agricultural soil. Water, Air, and Soil Pollut. 22:209-216.

Bartlett, J. and A. Norman. 1938. Changes in the lignin fraction of corn stalks during aerobic decomposition. Soil Sci. Soc. Am. Proc. 3:210-216.

Beare, M., M. Cabrera, P. Hendrix., and D. Coleman. 1994. Aggregate-protected and unprotected organic matter pools in conventional- and no-tillage soils. Soil Sci. Soc. Am. J. 58:787-795.

Betlach, M., J. Tiedje, and R. Firestone. 1981. Assimilatory nitrate uptake in <u>Pseudomonas fluorescens</u> studied using nitrogen-13. Arch. Microbiol. 129:135-140.

Blet-Charaudeau, C., J. Muller., and H. Laudelout. 1990. Kinetics of carbon dioxide evolution in relation to microbial biomass and temperature. Soil Sci. Soc. Am. J. 54:1324-1328.

Boyle, M. and E. Paul. 1989. Carbon and nitrogen mineralization kinetics in soil previously amended with sewage sludge. Soil Sci. Soc. Am. J. 53:99-103.

Bremner, J. and C. Mulvaney. 1982. Nitrogen total. p. 595-625. In A. L. Page. et al. (ed.) Methods of soil analysis. Part 2. 2nd ed. Agron. Monogr. 9. ASA and SSSA, Madison, WI.

Broadbent, F., K. Tyler, and G. Hill. 1957. Nitrification of ammoniacal fertilizers in some California soils. Hilgardia. 27:247-267.

Cervelli, S., G. Petruzzelli, and A. Perna. 1987. Fly ashes as an amendment in cultivated soils. I. Effect on mineralization and nitrification. Water, Air, and Soil Pollut. 33:331-338.

Cervelli, S., F. Giovanni, A. Perna, and G. Petruzzelli. 1990. A two-year experience on 15N-urea fertilization of a corn field amended with fly ash. Water, Air, and Soil Pollut. 36:173-181.

Chang, A., L. Lund, A. Page, and J. Warneke. 1977. Physical properties of fly ash-amended soils. J. Environ. Qual. 6:267-270.

Clark, M. and J. Gilmour. 1983. The effect of temperature on decomposition at optimum and saturated water contents. Soil Sci. Soc. Am. J. 47:927-929.

Coppola, S. 1983. Soil microbial activities as affected by application of composted sewage sludge. p. 170-195. In G. Catroux et al. (ed.) The influence of sewage sludge application on physical and biological properties of soils. Reidel Publ. Co., Boston, MA.

Degens, B. and G. Sparling. 1996. Changes in aggregation do not correspond with changes in labile organic C fractions in soil amended with 14C-Glucose. Soil Biol. Biochem. 28:453-462.

Demuyncle, M., E. Nyns, and H. Naveau. 1985. Use of digested effluents in agriculture. p.2-3. In Williams et al. (ed.) Long term effects of sewage sludge and farm slurries applications. Elsevier, New York.

Denmead, O., J. Freeney, and J. Simpson. 1982. Dynamics of ammonia volatilization during furrow irrigation of maize. Soil Sci. Soc. Am. J. 64:149-155.

Elseewi, A. and A. Page. 1984. Molybdenum enrichment of plants grown on fly ash-treated soils. J. Environ. Qual. 13:394-398.

Elseewi, A., I. Straughan, and A. Page. 1980. Sequential cropping of fly ash-amended soils: Effects on soil chemical properties and yield and elemental composition of plants. Sci. Total Environ. 15:247-259.

Environmental Protection Agency. 1978. Soluble organic nitrogen characteristics and removal. EPA-600/2-78-030. EPA, Washington, DC.

Environmental Protection Agency. 1984. Use and disposal of municipal wastewater sludge. EPA-625/10-84-003. EPA, Washington, DC.

Environmental Protection Agency. 1988. Wastes from the combustion of coal by electric utility power plants. USEPA Rep. 530-SW-88-002. USEPA, Washington, DC.

Epstein, E. 1975. Effect of sewage sludge on some soil physical properties. J. Environ. Qual. 4:139-142.

Epstein, E., D. Keane, J. Mesinger, and J. Legg. 1978. Mineralization of nitrogen from sewage sludge and sludge compost. J. Environ. Qual. 7:217-222.

Epstein, E., J. Taylor, and R. Chaney. 1976. Effects of sewage sludge and sludge compost applied to soil on some physical and chemical properties. J. Environ. Qual. 5:422-426.

Fine, P., U. Mingelgrin, and A. Feigin. 1989. Incubation studies of the fate of organic nitrogen in soils amended with activated sludge. Soil Sci. Soc. Am. J. 53:444-450.

Fuller, W. and A., Norman. 1943. Decomposition by Pseudomonas ephemerocyanea of jute preparations containing different quantities of cellulose and lignin. J. Bacteriol. 46:291-297.

Garau, M., M. Felipo, and M. Villa. 1986. Nitrogen mineralization of sewage sludges in soils. J. Environ. Qual. 3:225-228.

Gilmour, J., M. Clark, and G. Sigua. 1985. Estimating net nitrogen mineralization from carbon dioxide evolution. Soil Sci. Soc. Am. J. 49:1398-1402.

Gilmour, J., F. Roman, and M. Clark. 1996a. Decomposition of biosolids in a disposal site soil. J. Environ. Qual. 25:1083-1086.

Gilmour, J., M. Clark, and S. Daniel. 1996b. Predicting long-term decomposition of biosolids with a seven-day test. J. Environ. Qual. 25:766-770.

Gilmour, J. and C. Gilmour. 1980. A simulation model for sludge decomposition in soil. J. Environ. Qual. 9:194-199.

Goldstein, N. 1991. Sludge management practices in the US. Biocycle 32:46-48.

Gupta, S., R. Dowdy, and W. Larson. 1977. Hydraulic and thermal properties of a sandy soil as influenced by incorporation of sewage sludge. Soil Sci. Soc. Am. J. 41:601-605.

Hamilton,J. and R. Lowe. 1981. Organic matter and N effects on soil nitrite accumulation and resultant nitrite toxicity to tobacco transplants. Agron. J. 73:787-790.

Harada, T. and H. Kai. 1968. Studies on the environment conditions controlling nitrification in soils. I. Effects of ammonium and total salts in media on the rate of nitrification. Soil Sci. Plant. Nutr. 14:20-26.

Harding, S., C. Clapp, and W. Larson. 1984. Nitrogen availability and uptake from field soils five years after addition of sewage sludge. J. Environ. Qual. 14:95-100.

Hart S., J. Stark, E. Davidson, and M. Firestone. 1994. Nitrogen mineralization, immobilization and nitrification. p. 985-1018. In R. W. Weawer (ed.) Methods of soil analysis, part 2. Microbiological and bichemical properties. SSSA, Madison, WI.

Hasbach, A. 1991. Putting sludge to work. Pollut. Eng. 23:62-65.

Hatcher, J. and L. Wilcox. 1950. Colorimetric determination of boron using carmine. Anal. Chem. 22:567-573.

Hattori, H. 1988. Microbial activities in soil amended with sewage sludge. Soil. Sci. Plant Nutr. 34:221-232.

Hattori, H. and S. Mukai. 1986. Decomposition of sewage sludges in soil as affected by their organic matter composition. Soil Sci. Plant Nutr. 32:421-432.

Higgins, A., A. Kaplovsky, and J. Hunter. 1982. Organic composition of aerobic, anaerobic, and compost-stabilized sludges. J. Water. Pollut. Control Fed. 54:466-473.

Hinedi, Z., A. Chang, and R. Lee. 1988. Mineralization of phosphorus in sludge amended soils monitored by phosphorus-31-nuclear magnetic resonance spectroscopy. Soil Sci. Soc. Am. J. 52:1593-1596.

Hobson, P., S. Bousfield, and R. Summers. 1974. Anaerobic digestion of organic matter. CRC Crit. Rev. Environ. Control 4:131-191.

Hohla, G., R. Jones, and T. Hinesly. 1978. The effect of anaerobically digested sewage sludge on organic fractions of Blount silt loam. J. Environ. Qual. 7:559-563.

Hsieh Y., L. Douglas, and H. Motto. 1981a. Modeling sewage sludge decomposition in soil: I. Organic carbon transformation. J. Environ. Qual. 10:54-59.

Hsieh, Y., A. Lowell, A. Douglas, and H. Motto. 1981b. Modeling sewage sludge decomposition in soil: II. Nitrogen transformations. J. Environ. Qual. 10:59-63.

Johnson, D., and W. Guenzi. 1963. Influence of salts on ammonium oxidation and carbon dioxide evolution from soil. Soil Sci. Soc. Am. Proc. 27:663-666.

Justice, J. and R. Smith. 1962. Nitrification of ammonium sulfate in a calcareous soil as influenced by a combination of moisture, temperature and levels of added N. Soil Sci. Soc. Am. Proc. 26:246-250.

Keeney, D. and D. Nelson. 1982. Nitrogen inorganic forms. p. 643-698. In A. L. Page et al. (ed.) Methods of soil analysis. Part 2. 2nd ed. Agron. Monogr. 9. ASA and SSSA, Madison, WI.

Kelling, K., D. Keeney, L. Walsh, and J. Ryan. 1977. A field study of the agricultural use of sewage sludge. III. Effect on uptake and extractability of sludge-born metals. J. Environ. Qual. 6:352-358.

Kempers, A. 1974. Determination of sub-microquantities of ammonium and nitrates in soils with phenol, sodiumnitroprusside and hypochlorite. Geoderma 2:201-206.

Kladivko, E. and D. Nelson. 1979. Changes in soil properties from application of anaerobic sludge. J. Water Pollut. Control Fed. 51:325-331.

Laura, R. 1977. Salinity and nitrogen mineralization in soil. Soil Biol. Biochem. 9:33-336.

Lawson, R. and C. Still. 1957. The biological decomposition of lignin. Tappi. 40:56-61.

Lerch, R., K. Barbarick, L. Sommers, and D. Westfall. 1992. Sewage sludge proteins as labile carbon and nitrogen sources. Soil Sci. Soc. Am. J. 56:1470-1476.

Lindeman, W. and M. Cardenas. 1984. Nitrogen mineralization potential and nitrogen transformations of sludge-amended soil. Soil Sci. Soc. Am. J. 48:1072-1077.

Lunt, H. 1959. Digested sewage sludge for soil improvement.Conn. Agric. Exp. Stn. (New Haven) Bull. 622.

Magdoff, F. and F. Chromec. 1977. Nitrogen mineralization from sewage sludge. J. Environ. Sci. Health. 4:191-201.

Malhi, S. and W. Mcgill. 1982. Nitrification in three Alberta soils: Effect of temperature, moisture, and substrate concentration. Soil Biol. Biochem. 14:393-399.

McLaughlin, M. and L. Champion. 1987. Sewage sludge as a phosphorus amendment for sesquioxic soils. Soil Sci. 143:113-119.

McLintosh, T. and L. Frederick. 1958. Distribution and nitrification of anhydrous ammonia in a Nicollet sandy clay loam. Soil Sci. Soc. Am. Proc. 22:402-405.

Miller, R. 1974. Factors affecting the decomposition of an anaerobically digested sewage sludge in soil. J. Environ. Qual. 3:376-380.

Molina, J. 1985. Components of rates of ammonium oxidation in soil. Soil Sci. Soc. Am. J. 603-609.

Morril, L. and J. Dawson. 1967. Patterns observed for the oxidation of ammonium to nitrate by soil microorganisms. Soil Sci. Soc. Am. Proc. 31:75-760.

Mulford, F. and D. Martens. 1971. Response of alfalfa to boron in fly ash. Soil Sci. Soc. Am. Proc. 35:296-300.

Neilson, J. and I. Pepper. 1990. Soil respiration as an index of soil aeration. Soil Sci. Soc. Am. 54:428-432.

Nelson, P., J. Ladd, and J. Oades. 1996. Decomposition of 14C-labelled plant material in a salt affected soil. Soil Biol. Biochem. 28:433-441.

Oba, Y. and Q. Nguyen. 1982. Decomposition of sewage sludge in soil: III. Nitrogen mineralization of 15N-labeled sewage sludge in soil. Soil Sci. Plant Nutr. 1:119-130.

Pagliai, M., G. Guidi, M. La Marca, M. Giachetti, and G. Lucamante. 1981. Effects of sewage sludges and compost on soil porosity and aggregation. J. Environ. Qual. 10:556-561.

Parker, C. and L. Sommers. 1983. Mineralization of nitrogen in sewage sludges. J. Environ. Qual. 12:150-156.

Petruzzelli, G., L. Lubrano, and S. Cervelli. 1987. Heavy metal uptake by wheat seedlings grown in fly ash-amended soils. Water Air Soil Pollut. 32:389-395.

Pichtel, J. 1990. Microbial respiration in fly ash/sewage sludge-amended soils. Environ. Pollut. 63: 225-237.

Pichtel, J. and J. Hayes. 1990. Influence of fly ash on soil microbial activity and populations. J. Environ. Qual. 19:593-597.

Plank, C. and D. Martens. 1974. Boron availability as influenced by application of fly ash to soil. Soil

Sci. Soc. Am. Proc. 38:974-977.

Plank, C., D. Martens, and D. Hallock. 1975. Effect on soil application of fly ash chemical composition and yield of corn (Zea mays L.) and on chemical composition of displaced soil solutions. Plant Soil 42:465-476.

Premi, P. and A. Cornfield. 1969. Incubation study on nitrification of digested sewage sludge added to soil. Soil Biol. Biochem. 1:1-4.

Rebhun, M. and J. Manka. 1971. Classification of organics in secondary effluents. Environ. Sci. Technol. 5:606-609.

Ross, D., T. Speir, K. Tate, and V. Orchard. 1985. Effects of sieving on estimations of microbial biomass and carbon and nitrogen mineralization in soil under pasture. Australian J. Soil Research 23:319-324.

Ryan, J. and D. Keeney. 1975. Ammonia volatilization from surface applied wastewater sludge. J. Water Pollut. Control Fed. 47:386-393.

Ryan, J., D. Keeney, and L. Walsh. 1973. Nitrogen transformations and availability of an anaerobically digested sewage sludge in soil. J. Environ. Qual. 2:493.

Sakata, M. 1987. Movement and neutralization of alkaline leachate at coal disposal sites. Environ. Sci. Technol. 21:771:777.

Schaumberg, G., D. LeVesque-Madore, G. Sposito, and L. Lund. 1980. Infrared spectroscopic study of the water-soluble fraction of sewage sludge-soil mixtures during incubation. J. Environ. Qual. 9:297-303.

Schmidt E. and L. Belser. 1994. Autotrophic nitrifying bacteria. p. 159-176. In R. W. Weaver et al. (ed). Methods of soil analysis, Part 2. Microbiological and biochemical properties. SSSA. Madison, WI.

Sims, J. 1990. Nitrogen mineralization and elemental availability in soils amended with cocomposted sewage sludge. J. Environ. Qual. 19:669-675.

Sommers, L. 1977. Chemical composition of sewage sludges and analysis of their potential use as fertilizers. J. Environ. Qual. 6:225-231.

Sommers, L.E., D.W. Nelson, and K.J. Yost. 1976. Variable nature of chemical composition of sewage sludges. J. Environ. Qual. 5:303-306.

Spencer, L. and L. Drake. 1987. Hydrogeology of an alkaline fly ash landfill in eastern Iowa. Ground Water. 25:519-526.

Stojanovic, B. and M. Alexander. 1958. Effect of inorganic nitrogen on nitrification. Soil Sci. 86:208-215.

Terry, R., D. Nelson, L. Sommers, and G. Meyer. 1978. Ammonia volatilization from wastewater sludge applied to soils. J. Water Pollut. Control. Fed. 50:2657-2665.

Terry, R., D. Nelson, and L. Sommers. 1979. Carbon cycling during sewage sludge decomposition in soils. Soil Sci. Soc. Am. J. 43:494-499.

Terry, R., D. Nelson, and L. Sommers. 1981. Nitrogen transformations in sewage sludge-amended soils as affected by soil environmental factors. Soil Sci. Soc. Am. J. 45:506-512.

Tester, C., L. Sikora, J. Taylor, and J. Parr. 1977. Decomposition of sewage sludge compost in soil: I. Carbon and nitrogen transformation. J. Environ. Qual. 6:459-463.

Tester, C., and J. Parr. 1983. Decomposition of sewage sludge compost in soil: IV. Effect of indigenous salinity. J. Environ. Qual. 12:123-126.

Tester, C., L. Sikora, J. Taylor, and J. Parr. 1979. Decomposition of sewage sludge compost in soil: III. Carbon, nitrogen, and phosphorus transformations in different sized fractions. J. Environ. Qual. 8:79-82.

Theis, T., J. Westrick, C. Hsu, and J. Marley. 1978. Field investigation of trace metals in groundwater from fly ash disposal. J. Water Pollut. Control Fed. 50:2457-2469.

Varanka, M., Z. Zablocki, and T. Hinesly. 1976. The effect of digestion sludge on soil biological activity. J. Water Pollut. Control Fed. 48:1728-1740.

Wallace, A., G. Alexander, S. Soufi, and R. Mueller. 1980. Micronutrient supplying power of pyrite and fly ash. J. Plant Nutr. 2:147-153.

Wallace, A. and G. Wallace. 1986. Enhancement of the effect of coal fly ash by a polyacrylamide soil conditioner on growth of wheat. Soil Sci. 141:387-389.

Wiseman, T. and L. Zibilske. 1988. Effects of sludge application sequence on carbon and nitrogen mineralization in soil. J. Environ. Qual. 17:334-339.

Yoneyama, T. and T. Yoshida. 1978. Nitrogen mineralization of sewage sludges in soil. Soil Sci. Plant Nutr. 24:139-144.

VITA

The author of this work was born in 1971, Konya, Turkey. He received his B.S. diploma in soil science at Selcuk University, Konya, Turkey. He was appointed as a research assistant at Gaziosmanpasa University, Tokat, Turkey. Same year, he came to the United States for an M.S. program in soil science. Since then, he has been working under Dr. Mark Coyne at University of Kentucky. He doesn't have a published work yet.