PHOSPHORUS DISTRIBUTION AND MOBILITY THROUGH SOILS USING SWINE LAGOON WASTE

By

DAMON GENE WRIGHT

Bachelor of Science

Oklahoma State University

Stillwater, OK

1998

Submitted to the Faculty of the
Graduate College of the
Oklahoma State University
in partial fulfillment of
the requirements for
the Degree of
MASTER OF SCIENCE
May, 2002

PHOSPHORUS DISTRIBUTION AND MOBILITY THROUGH SOILS USING SWINE LAGOON WASTE

Thesis Approved:

Jeffory Hattey
Thesis Adviser

Nicholas T. Basta

Doug Hamilton

Hailin Zhang

Timothy J. Pettibone
Dean of Graduate College

ACKNOWLEDGMENTS

I truly appreciate all the faculty and students who have assisted me throughout my research and studies. I offer my utmost gratitude to my committee members, Drs. Jeff Hattey, Nick Basta, Hailin Zhang, and Doug Hamilton. Specifically, I would like to thank Dr. Hattey for his flexibility, friendship, and wisdom over the past seven years. Dr. Hattey inspired me to get involved in the field of soil science and helped me discover my passion for teaching. I would also like to offer my thanks to the Department of Plant and Soil Sciences for funding my research in part through the teaching assistantship.

I sincerely thank my stepfather who originally spurred my interest in science as a child with a simple chemistry set, a telescope, and countless hours of viewing Mr. Wizard. Also, I would like to thank my mother who unconditionally supported and encouraged me throughout my educational career. Thanks Mom, Ritchie, and Bradley.

Additionally, I appreciate the hard work my fellow students and friends gave freely which made this project possible: Clemn Turner, Jason Warren, Jeremy and Amy Seiger, Libby Dayton, Felecia Armstrong, Shawn Zupancic, and Mike Pumphrey.

I would also like to thank my wife, my best friend, and my love, Krista. She not only gave me emotional support during my graduate career, but she also assisted in hours upon hours of lab work (usually late nights). I could not have done this without you.

Finally, I praise God for his faithfulness, guidance, and sacrificial love. "I am the way and the truth and the life. No one comes to the Father except through me." John 14:6

TABLE OF CONTENTS

Chapter Page

III. PHOSPHORUS MOBILITY IN SOILS FROM LEACHING OF SWINE LAGOON WASTE

LIST OF TABLES

Table Page

CHAPTER 2

CHAPTER 3

LIST OF FIGURES

Figure Page

CHAPTER 2

CHAPTER 3

FOREWORD

This document is presented as three chapters. The first chapter consists of background information to aid in the understanding of the following two chapters. The second and third chapters are formatted as stand-alone articles under the specifications of two journals, Soil Science Society of America Journal and Journal of Environmental Quality. This approach facilitates a more streamlined method of preparing manuscripts for publication without the necessity of rewriting the thesis.

CHAPTER I

MANAGEMENT OF SWINE WASTE PHOSPHORUS AND IMPACT OF EXCESS PHOSPHORUS ON THE ENVIRONMENT

INTRODUCTION

The continued expansion of the swine industry in Oklahoma and throughout the United States raises numerous environmental concerns. One of these concerns is the negative effect of phosphorus (P) on water quality due to agricultural nonpoint source pollution. In areas of intensive crop and livestock production, continual P applications of mineral fertilizer and manure have been made at levels exceeding crop uptake (Sharpley et al., 1994). As a result, surface soil accumulations of P have occurred to such an extent that the loss of P in surface runoff has become a priority management concern. In several cases, the capacity of the soil to adsorb further P additions has become limited, with increased accumulation of P in ground water observed in the U.S. (Federico et al., 1981) and internationally (Breeuwsma and Silva, 1994). Swine manure management is becoming ever more important as water quality deteriorates for the use of drinking, industry, agriculture, recreation, and numerous other purposes.

SWINE PRODUCTION IN OKLAHOMA

Swine production in Oklahoma, particularly the Panhandle region, has increased tremendously over the past decade. As hog production rises, swine manure increases accordingly. In most large swine production systems the waste is stored in earthen lagoons where the feces, urine, flush water, and spilled feed are all biologically treated by ubiquitous microorganisms. The main function of the lagoon treatment system is to reduce the pollution potential of the waste into waterbodies (Hamilton, 1996). When a lagoon reaches maximum capacity, the effluent, liquid portion of the lagoon waste, and the sludge, semi-solid portion that settles at the bottom of the lagoon, is applied to crops and forages due to the waste's nutrient benefit. The land application of animal wastes for plant nutrition has been practiced around the world for centuries (Sims, 1995).

Between 1992 and 1997 hog inventory within the state of Oklahoma increased exponentially, growing from 260,682 to 1,689,700 total head (National Agricultural Statistics Service, 1997). In 1996, Oklahoma ranked 10th nationally in total hog numbers, with the Oklahoma Panhandle being the leading area of swine growth in the United States. Over the past decades, swine production has switched from numerous small family operations to fewer and significantly larger facilities where the animals are more confined. More than 6,600 concentrated animal operations existed in the United States in 1992 and a large majority of these facilities used lagoons as their waste management practice. (Kosco and Hall, 1999). Worldwide, swine production creates 1.7 billion tons of liquid manure annually (Choudhary et al., 1996).

IMPORTANCE OF P IN AGRICULTURE

Phosphorus is an essential nutrient for plant growth. Low concentrations in native soils (100-3000 mg P kg^{-1}) and low solubility (<0.01 mg P L^{-1}) make it a critical nutrient limiting plant growth (Sharpley, 1999). The primary roles of P in plants are for energy storage and transfer (ATP and ADP), a critical component of cellular membranes (phospholipids), and a constituent of DNA and RNA (Brady and Weil, 1999). Also, P is associated with increased root growth in most plants (Tisdale et al., 1993). Plant roots absorb phosphorus dissolved in the soil solution mainly as the orthophosphate ions HPO_4^{-2} and $H_2PO_4^-$. Once P is added to soils through animal waste or fertilizers, numerous reaction sinks exist in the soil in which the P is "tenaciously held by soils through both electrostatic and nonelectrostatic mechanisms" (Sparks, 1995). Before we can develop agronomically and environmentally sound management systems for P, we need to understand what forms of P occur in soils and how these sinks change with P additions from animal waste. A description of the processes that occur between added fertilizer or animal waste P and soils is described below.

REACTIONS OF P IN SOIL

An understanding of how swine waste P reacts in soil is important for environmental and agronomic implications. The bioavailability of P for plants and organisms is highly dependant upon the mineral pool in which the P exists. Most P in the soil and P added from animal wastes or fertilizers are unavailable for immediate plant uptake. Less than 30% of applied P is actually recovered in plants due to the sorption of P into mineral fractions (Johnson et al., 1997). From an environmental perspective, the soluble organic and inorganic phosphates are the forms that are of primary concern. Very low concentrations of soluble P exist in most soils due to the sorption of P into stable mineral fractions. However, continuous applications of swine waste can buildup the loosely bound/soluble P soil fraction to levels that can potentially impact environmental quality.

Sorption by soil mineral pools is the mode by which the applied P is rendered nonlabile and immobile. Once applied to soil, P has numerous mineral pools where sorption processes, a collective term for adsorption and/or precipitation reactions from the soil solution (Sposito, 1989), can occur. The main property controlling the inorganic P form is soil pH. Ligand exchange reactions between aluminum (Al), manganese (Mn), and iron (Fe) oxides with P causes the formation of specifically adsorbed complexes. Strong covalent bonds are created when phosphate exchanges with hydroxyl groups associated with hydrous oxide clays, which render the P virtually insoluble (Sparks, 1995; Sharpley, 1999). In low pH soils the kinetics and equilibrium thermodynamics of these ligand reactions is increased dramatically. Also, in highly acid soils (pH <5), free Fe^{3+} and Al^{3+} in solution can precipitate strengite, variscite, or other related insoluble minerals. Alkaline soils promote the precipitation of calcium phosphate minerals such as brushite ($CaHPO_4\cdot 2H_3O$), monetite ($CaHPO_4$), octacalcium phosphate ($Ca_4H(PO_4)\cdot 2.5H_2O$), and hydroxyapatite ($Ca_5(PO_4)_3OH$) (Lindsay, 1979). These P minerals form readily when the soil pH >7 and the precipitated calcium phosphates are relatively insoluble.

The kinetics of P sorption occurs in a two-step process. Adsorption is believed to be the primary process during the initial rapid reaction that takes place within the first four hours of contact between the P and soil particulates (Singh and Jones, 1976). Nonspecific adsorption, and ligand exchange reactions on clay edges are the processes that dominate during the adsorption of P. Over the period of several weeks and longer, the slower process of precipitation takes place between the added P and Fe, Al, or Ca (Sposito, 1989; Sawhney and Hill, 1975; Singh and Jones, 1976).

IMPACT OF EXCESS P ON THE ENVIRONMENT

Surface runoff is the primary mode of P transport to surface water. The loosely bound/soluble P sorbed to the eroded soil particles must detach to present a problem to water quality. Soils enriched with P from continuous manure applications have significantly contributed to the deterioration of surface water quality in the U.S. (Eghball et al., 1996). Phosphate in drinking water is not considered to have toxic effects to humans or animals (Amdur et al., 1991); therefore, no drinking water standards have been established for P by the U.S. Environmental Protection Agency (U.S. EPA, 1990). The problem exists in that the excess P in surface waters promotes a process known as eutrophication. Eutrophication is defined as the nutrient enrichment of lakes, ponds, and other such waters that stimulates the growth of aquatic organisms, which leads to a deficiency of oxygen in water (Brady and Weil, 1999). Eutrophication of most fresh water around the world is accelerated by P inputs (Kauppi et al., 1993; Sharpley et al., 1994). Excess P in freshwater stimulates blooms of cyanobacteria, or blue-green algae, which in turn depletes the oxygen content in the water. Other problems associated with algal blooms are foul odors in drinking water, summer fish kills, loss of biodiversity, the formation of trihalomethane during water chlorination in treatment plants, and the formation of a potentially lethal water soluble neuro- and hepatotoxins created during the death of algal blooms (Kotak et al., 1994; Martin and Cooke, 1994). P concentrations found to stimulate eutrophication in fresh water are between 0.05-0.1 mg P L^{-1} (U.S. EPA, 1976).

Phosphate has historically been considered an immobile nutrient that does not readily leach to groundwater. However, recently leaching has been demonstrated is a legitimate pathway for the movement of P into surface and groundwater (Chardon et al., 1997; Eghball et al., 1996; Lucero, 1995). The downward movement of P can occur slowly by accumulation in the soil profile or relatively quickly from preferential flow through soil cracks, root channels, or borrows of worms or other animals. Also, swine lagoon failures are becoming a significant source of P pollution. As swine production in Oklahoma increases, an increase in the utilization of lagoons to store and treat the swine waste grows as well. With increasing lagoon volumes and total number of lagoons, there is an increasing risk of lagoon failure as well (Culley and Phillips, 1989; Westerman et al., 1995).

MANAGING SWINE MANURE

Managing manure can improve overall farming operations while protecting the environment and reducing fertilizer cost to crops and forages. A waste management system should be part of a total soil and water conservation plan for farms producing livestock. A nutrient management plan is an important part of the total waste management plan. A nutrient management plan includes soil testing and testing the waste in order to match the crop needs to the nutrients available. This also means a large enough storage system will be needed to allow waste to be applied when the most benefit will be realized by the crops from the available nutrients. Having ample storage is essential to allow land application when needed by plants rather than having to apply to saturated fields or pastures where runoff is more likely. The storage and handling methods along with the method of field application can make a considerable difference in plant nutrients available.

As swine production rises in Oklahoma and throughout the United States, the careful management of manure and waste will become increasingly important. The adoption of some best management practices by swine production operations will increase the nutrient value of the waste and help protect water quality. Soil testing and periodic nutrient analysis of the manure are essential practices for operators that land apply waste. To prevent the accumulation of excess P in soil, manure should be applied according to the P needs of the crop, and then supplemented with nitrogen fertilizer to accommodate the nitrogen need of the crop (Daniel et al., 1994). Also, incorporating or injecting the swine manure into the soil will improve root contact with the added P and significantly decrease the potential of transport off the field. Avoid spreading waste near wells, ponds, streams, or sloping areas where runoff potential is high. The addition of vegetated filter strips along streams will trap sediment, waste, and/or dissolved P before it can impact water quality. The use of conservation practices that minimize runoff and erosion on land where waste is applied is recommended. Crop residue management, cover crops, contour tillage, runoff water impoundments and terracing are conservation strategies that can limit transport of P to surface waters. The regular calibration of application equipment will ensure that waste is not being over- or under applied (Johnson et al., 1997).

Proper lagoon construction and management is critical for the storage and treatment of swine waste in an environmentally safe manner. Before lagoon installation, an investigation involving the area's susceptibility to flooding and the hydrology of the site should be examined. Avoid placing lagoon structures in areas nearby sensitive watersheds, high water tables, floodplains, or sloping topography. Impermeable clay or plastic liners prevent the leaching of waste and the use of settling basins make liquid waste easier to apply or recycle. Always be aware of lagoon levels and never allow a lagoon to overflow. Finally, maintaining vegetation, rodent and varmint control near lagoon edges can reduce the potential of lagoon failures.

Dietary manipulation has been proven to significantly decrease the amount of P excreted by swine. Most swine feed contains P levels 20-30% greater than the animal's requirement to guarantee a margin of safety (Carter, 1999). Therefore, the use of additional P supplements in swine operations wastes money, consumes valuable P resources, and produces manure rich in P. Another effective method to reduce P in manure is the addition of the enzyme phytase to swine diets. The addition of phytase to feed has been shown to reduce P excretion by 30-50% (Carter et al., 1996). Phytase is necessary to degrade phytate P in feed grain and meal however it is not enzyme produced by swine metabolism. If swine facilities in the U.S. supplement feed with phytase, the estimated decrease in P excretion would be 200 million tons per year (Cromwell, 1996). Again, P resources would be conserved and the environmental impact potential declines because less land would be needed to dispose of the manure.

CONCLUSIONS

With the rapid increase of swine production in Oklahoma, the management of manure P will become increasingly important for the protection of water quality. The majority of the soils in Oklahoma possess numerous mineral sinks for P that reduces the pollution potential. However, on land where surface runoff occurs, the P can still impact surface water and accelerate eutrophication. Generally, the use of swine manure in agronomic systems is a cost efficient and environmentally safe means to recycle waste if appropriate management practices are established and followed. Soil testing and nutrient analysis of the manure are critical steps required for all waste management plans to effectively reduce the P loading into surface waters. Swine production and agriculture in general, must start taking significant steps toward reducing P nonpoint source pollution before the impact to water quality and overall ecological health becomes irreversible. The implementation of every feasible best management practice to swine facilities would drastically reduce the polluting of America's most precious nature resource…water.

REFERENCES

Amdur, M.O., J. Dull, and E.D. Klassen, ed. 1991. Casarett and Doull's Toxicology. Fourth edition. Pergamon Press, New York, N.Y., USA.

Breeuwsma, A., and S. Silva. 1994. Phosphorus fertilization and environmental effects in The Netherlands and the Po region (Italy). Rep. 57. Wageningen, The Netherlands: Agric. Res. Dep., The Winand Staring Centre for Integrated Land, Soil and Water Research.

Brady, N.C., R.R. Weil. 1999. The Nature and Properties of Soils. Twelfth ed. Prentice Hall, Upper Saddle River, N.J.

Carter, S.D. 1999. Animal nutrition and waste management- swine. p.21-31 *In* T. Springer, J. Hattey, and H. Zhang (ed.) High Plains Animal Waste Management Conference. Goodwell, OK Mar. 2, 1999. Oklahoma State Univ., Stillwater, OK.

Carter, S.D., G.L. Cromwell, M.D. Lindemann, L.W. Turner, and T.C. Bridges. 1996. Reducing N and P excretion by dietary manipulation in growing and finishing pigs. J. Anim. Sci. 74(Supp. 1):59.

Chardon, W.J., O. Oenema, P. del Vriesema, J. Japenga, and D. Blaauw. 1997. Organic phosphorus in solutions and leachates from soils treated with animal slurries. J. Environ. Qual. 26:372-378.

Choudhary, M., L.D. Bailey, and C.A. Grant. 1996. Review of the use of swine manure in crop production: effects on yield and composition and on soil and water-quality. Waste Management & Research. 14:581-595.

Cromwell, G.L. 1996. Using nutritional practices to reduce nutrient load and control odor. National Pork Producers Council – 1996 Pork Profitability Summit. p. 107-126. Dec. 10-11. Myrtle Beach, S.C.

Culley, J.L.B., and P.A. Phillips. 1989. Retention and loss of nitrogen and solids from unlined earthen manure storages. Trans ASAE 32:677-683.

Daniel, T.C., A.N. Sharpley, D.R. Edwards, R. Wedepohl, and J.L. Lemunyon. 1994. Minimizing surface water eutrophication from agriculture by phosphorus management. J. Soil Water Conservation Supplement 49:30-38.

Eghball, B., G.D. Binford, and D.D. Baltensperger. 1996. Phosphorus movement and adsorption in a soil receiving long-term manure and fertilizer application. J. Environ. Qual. 25:1339-1343.

Federico, A.C., K.G. Dickson, C.R. Kratzer, and F.E Davis. 1981. Lake Okeechobee water quality studies and eutrophication assessment. Tech. Pub. 81-2. 270 pp. West Palm Beach, Florida: South Florida Water Management District.

Hamilton, D.W. 1996. What is a waste management system? Div. Ag. Sci. & Natural Resources. F-1734. Oklahoma State Univ., Stillwater, OK.

Johnson, G.V., W.R. Raun, H.A. Zhang, and J.A. Hattey. 1997. Oklahoma Soil Fertility Handbook. 4th ed. Dept. Agronomy, Oklahoma State Univ., Stillwater, OK.

Kauppi, L., O.P. Pietilainen, and S. Knuuttila. 1993. Impacts of agricultural nutrient loading in Finnish watercourses. Water Science Technology. 28:461-471.

Kosco, J.A., and W. Hall. 1999. Joint strategy targets farm waste. Resource 6, ASAE. p. 11-12. St. Joseph, MI.

Kotak, B.G., E.E. Prepas, and S.E. Hrudey. 1994. Blue-green algal toxins in drinking water supplies: Research in Alberta. Lake Line 14:37-40.

Lindsay, W.L. 1979. Chemical Equilibria in Soils. John Wiley & Sons, New York, NY.

Lucero, D.W., D.C. Martens, J.R. McKenna, and D.E. Starner. 1995. Accumulation and Movement of phosphorus from poultry litter application on a Starr clay loam. Commun. Soil Sci. Plant Anal. 26:1709-1718.

Martin, A., and G.D. Cooke. 1994. Health risks in eutrophic water supplies. Lake Line 14:24-26.

National Agricultural Statistics Service. 1997. 1997 Census of agriculture highlights [Online]. Available at http://www.nass.usda.gov/census/census97/highlights /ok/okst.txt (verified on Feb. 1999.)

Sawhney, B.L., and D.E. Hill. 1975. Phosphate sorption characteristics of soils treated with municipal wastewater. J. Environ. Qual. 4:342-346.

Sharpley, A.N. 1999. Phosphorus availability. p. D18-D38. *In* M.E. Sumner (ed.) Handbook of Soil Science. CRC Press, Boca Raton, FL.

Sharpley, A.N., S.C. Chapra, R. Wedepohl, J.T. Sims, T.C. Daniel, and K.R. Reddy. 1994. Managing agricultural phosphorus for protection of surface waters: Issues and options. J. Environ. Qual. 23:437-451.

Singh, B.B., and J.P. Jones. 1976. Phosphorus sorption and desorption characteristics of soils as affected by organic residues. Soil Sci. Soc. Am. J. 40:389-394.

Sims, J.T. 1995. Characteristics of animal wastes and waste-amended soils: An overview of the agricultural and environmental issues. Characteristics of Wastes and Waste Amended Soils. p 1-13. *In* Animal Waste and Land-Water Interface. Lewis Publishers, Boca Raton, FL.

Sparks, D.L. 1995. Environmental Soil Chemistry. Academic Press, San Diego, CA.

Sposito, G. 1989. The chemistry of soils. Oxford University Press, New York, NY.

Tisdale, S.L., W.L. Nelson, J.D. Beaton, and J.L. Havlin. 1993. Soil Fertility and Ferilizers. Fifth ed. Macmillan Pub. Co., New York.

U.S. Environmental Protection Agency. 1976. Quality criteria for water. U.S. Govt. Printing Office, Washington, DC. USA.

U.S. EPA. 1990. National water quality inventory. 1988 Report to Congress. Office of Water, U.S. Govt. Printing Office, Washington, D.C., USA.

Westerman, P.W., R.L. Huffman, and J.S. Feng. 1995. Swine-lagoon seepage in sandy soil. Trans. ASAE 38:1749-1760.

CHAPTER II

PHOSPHORUS FRACTIONATION ON THREE SOILS FOLLOWING SWINE EFFLUENT/SLUDGE INCUBATION

ABSTRACT

Phosphorus (P) fractionation procedures can characterize the distribution of soil P mineral forms following the application of P in swine waste. Fractionation of P is beneficial agronomically by identifying labile and nonlablile P pools for crop utilization. From an environmental perspective, fractionation methods can detect potentially mobile P fractions which could impact water quality. Three soils were incubated for 35 days with swine effluent or sludge at two moisture contents, saturated (1:1 soil/liquid) and unsaturated (1 bar). The discrete mineral fractions measured were loosely bound/soluble P (LB/S-P), aluminum phosphates (Al-P), iron phosphates (Fe-P), reductant soluble P (RS-P), and calcium phosphates (Ca-P). Mehlich III extractable P was also measured to determine P availability before and following the incubation period. The LB/S-P concentrations decreased significantly after the incubation for all unsaturated and saturated soils treated with the swine wastes. An acidic loam soil treated with effluents from two lagoon depths exhibited a decrease of the LB/S-P to undetectable levels. With beginning concentrations of 58.5 and 57.5 mg P kg^{-1} soil, no P was measured in the LB/S-P fraction after 35 days. The majority of the soils distributed the waste P primarily into the Al-P and Fe-P fractions. The soils displayed a difference in P distribution when comparing the saturated and unsaturated waste treated soils. For example, 38.1% of the P

extracted for the unsaturated acid loam soil was in the Fe-P pool; however, the Fe-P percentage rose to 71.4% for the saturated soil treatment. The Ca-P for the saturated Richfield clay loam decreased by 14.6-17.5% for the swine waste treated soils in comparison to the unsaturated soils. The Al and Fe fractions for all soils demonstrated the ability to sorb more P than the Ca mineral pool after the 35 day incubation. From an environmental perspective, the results of this project can be applicable in a lagoon failure situation where leaching waste rich in P could impact surface or groundwater quality. All soils exhibited an ability to sorb large concentrations of soluble P into stable and immobile mineral sinks. In a crop production circumstance, most added P was distributed into the nonlabile Al-P and Fe-P pools. Repeated applications of swine effluent in P deficient soils would likely be required to increase LB/S-P and Mehlich III P concentrations, but the application of swine sludge would provide a substantial amount of immediate available P for plant uptake.

INTRODUCTION

Phosphorus (P) fractionation procedures can aid in identification of mineral pools that are affected by the addition of P in fertilizers or animal wastes. Sequential chemical extraction methods have been used widely to characterize the distribution of soil P mineral forms. Chang and Jackson (1957) developed a widely used P fractionation procedure that classified inorganic phosphates into four main groups: calcium phosphate, aluminum phosphate, iron phosphate, and reductant soluble phosphate for the removal of occluded iron phosphate. Several modifications have been made to the scheme since the inception of the procedure. Peterson and Corey (1966) altered the pH of the NH_4F extracting solution; Williams et al. (1971) included a sodium citrate-sodium bicarbonate extraction to minimize the resorption of P by $CaCO_3$; and Hieltjes and Lijklema (1980) implemented the use of 1.0 *M* NH_4Cl for extraction of loosely bound/soluble P, 0.1 *N* NaOH for iron phosphates (Fe-P) and aluminum phosphates (Al-P), and 0.5 *N* HCl for calcium phosphates (Ca-P).

The soil P fractions are extracted and identified based on each fraction's solubility equilibria; therefore, the extractants used in the fractionation scheme are non-selective. However, the procedure is valuable for interpreting the relative distribution of P in soils. The discrete chemical P fractions measured in this project are the loosely bound/soluble P (LB/S-P), Al-P, Fe-P, reductant soluble P (RS-P), and Ca-P. A calcareous P fractionation method and a noncalcareous method are utilized in the experimentation. The procedures differ in that the noncalcareous scheme separates Al-P from Fe-P using NH_4F, F^- complexes Al, followed by the extraction of Fe-P with NaOH and sodium citrate dithionite bicarbonate. However, the NH_4F reacts with the $CaCO_3$ in a calcareous soil to form CaF_2 and other secondary precipitates (Kuo, 1996). A NaOH and NaCl solution is used in the calcareous method to extract the Al and Fe-P fraction simultaneously.

The objective of this project is to evaluate how P in swine effluent and sludge affects soil mineral pools under saturated and unsaturated conditions. Fractionation procedures will be used to identify the mineral pools altered by the addition of P in swine waste. Swine effluent from two lagoon depths and swine sludge will be incubated with soils differing in physical and chemical properties to compare the differences in P distribution. Lagoon effluent from the 0-2 ft. depth and the 8-10 ft. depth were used to assess if differences in chemical composition based on lagoon layering altered P sorption. The intent of the project is not to quantify the P fraction changes, but to evaluate how the swine waste P affects the mineral pools of soils differing in chemical/physical properties and moisture contents.

The results of this project can be beneficial for environmental and agricultural implications. Environmentally, this study is important from the perspective of a lagoon failure incident. Numerous documented cases of subsurface lagoon failures have been reported within the past thirty years. For example, Mallin et al. (1997) studied a poultry waste lagoon rupture and a swine waste lagoon leak in North Carolina during the summer of 1995. High concentrations of orthophosphate and total P (>11.0 mg P L^{-1} measured for both) was discovered in stream waters several kilometers away from the lagoon sites. Excessive algal bloom, numerous fish and mollusk kills, and high coliform concentrations were observed in these accidents. Eight other accidents were recorded that summer in North Carolina in which an approximated 4 million liters of waste was discharged from each facility. Ciravolo et al. (1979) found low levels of groundwater contamination due to the rupture of lagoon seals from the drying of exposed subsoil during a low containment period and from gas release due to microbial activity beneath the seal. Westerman et al. (1995) evaluated P enrichment of groundwater from two newly-formed clay lined lagoons underlied by a sandy soil and Withers et al. (1998) found significant increases in groundwater P, K, N, Na, Cl, S, and *Escherichia coli* from a dairy lagoon fissure. Numerous other documented cases exist that demonstrate the impact of lagoon failures on groundwater quality.

Phosphate leaching is normally not considered a threat to groundwater quality, however past studies have shown that subsurface flow can be a significant transport pathway of P into surface waters sensitive to eutrophication (Calvert, 1975; Izuno et al., 1991). The sorption of P into mineral fractions that immobilize the phosphate (Al-P, Fe-P, RS-P, and Ca-P) is important to recognize for the protection of groundwater. The identification of P in the LB/S-P fraction is important as well. The LB/S-P is potentially mobile and could impact surface and groundwater quality.

Agronomically, this project will evaluate how swine effluent and sludge P affects labile mineral pools for crop production. The LB/S-P is the fraction immediately available for plant uptake. As for the Al-P, Fe-P, RS-P, and Ca-P, these mineral pools represent P which are not as readily available for plant uptake. Mehlich III extractable P will be analyzed for all soils to determine P availability following a 35 day incubation with the swine wastes. This project is also applicable for agronomic systems utilizing subsurface drip irrigation of swine lagoon effluent. In the Oklahoma Panhandle region, this form of irrigation is being proposed to supply crops with moisture due to the lack of precipitation in the area (approximately 30-38 cm precipitation per year). Also, subsurface drip irrigation of lagoon effluent provides nutrients for plant growth and is an effective method of waste management for swine production facilities. The experiment will provide data on the distribution of P in soils under fluctuating moisture conditions (anaerobic conditions while irrigating and aerobic after plant uptake, evaporation, or leaching from the soil profile).

The primary objective of this experiment is to determine what inorganic soil mineral pools are affected by the addition of P in swine lagoon effluent/sludge under different soil moisture contents. These factors will be evaluated at time zero and after an incubation period to assess the potential differences in P distribution over time. Additionally, plant available P (Mehlich III extractable P) will be measured for the swine waste treated soils before and after the incubation to determine how P availability changes over time.

MATERIALS AND METHODS

Soils

Three soils representing a wide range of chemical and physical properties were chosen from different locations in Oklahoma where swine production is prevalent (Figure I-1) . The Richfield clay loam (fine, smectitic, mesic Aridic Argiustoll), a soil series widely distributed in the Oklahoma Panhandle, was obtained from a cultivated field at the Oklahoma Panhandle Research Station, Goodwell, OK. The Richfield was selected because it is found in the region where the sharpest increase in swine production has occurred in Oklahoma over the last decade. The Richfield series consists of very deep, well-drained, moderately slow permeable soils formed in calcareous loess plains. The surface soil samples used were calcareous with an average pH value of 8.2 (1:2 soil/water solution). The Dennis silt loam (fine, mixed, thermic Aquic Argiudoll), a series located throughout central Oklahoma, was taken from a cultivated field in Holdenville, OK. The Dennis series is a deep, somewhat poorly drained, slowly permeable soil formed from weathered shale. The surface soil used was moderately acidic with an average pH value of 5.7. The Bethany silt loam (fine, mixed, superactive, thermic Pachic Paleustoll), a soil series found in north central Oklahoma, was obtained from a cultivated research plot in Hennessey, OK. The Bethany series consists of a deep, well drained, slowly permeable soils formed in loess or alluvium over shale (USDA-NRCS, 1996). The mean pH of the Bethany surface samples was near neutral with a value of 7.3.

Incubation Flask Preparation

Effluent and sludge for the incubations were obtained from a model facultative swine lagoon created by the Oklahoma State University, Biosystems and Agricultural Engineering Department. The model lagoon was filled with material from an anaerobic swine lagoon located at Oklahoma State University, Stillwater, OK. For incubations, three lagoon layers were sampled from the model lagoon: two-foot depth effluent, eight-foot depth effluent, and sludge from the base of the model. A 2x4 factorial arrangement of treatments was established using two moisture contents and four liquid treatments. The soils were mixed with DI water (control), 2-ft depth effluent, 8-ft depth effluent, or sludge. Moisture contents of 1-bar and a saturated were prepared using liquid treatments (DI water, the 2 or 8-ft effluents, or the sludge). The saturated treatment was created by adding 1:1 soil/liquid (1 part air-dried soil: 1 part liquid treatment). A moisture content of 1 bar was determined by the pressure plate procedure and the appropriate amount of liquid treatment was added. Since the treatments were established on a moisture content basis, unequal P additions were added due to the differences in P concentration of the wastes. One-bar moisture contents correspond to 0.12, 0.16, and 0.17 g water g soil^{-1} for the Dennis, Bethany, and Richfield soils, respectively.

One hundred grams of soil and the appropriate volume of water/effluent/sludge (based on moisture content) were incubated in 250 mL Erlenmeyer flasks for a period of 35 days. The flasks were sealed with a rubber stopper and mixed twice a week to ensure proper contact of the soil with the liquid treatment. The samples were stored in a dark container to simulate subsurface conditions under a lagoon. The incubations were all held at constant room temperature. After the 35 day incubation period, the samples were stored in a refrigerator until the P extraction commenced. All samples were air-dried prior to the fractionation process; therefore, the anaerobic soil incubations were oxidized before sample analysis. Photographs of the incubation bottles following the swine effluent or sludge additions are provided in Figure I-2.

Inorganic P Fractionation for Noncalcareous Soils

The procedure for noncalcareous P fractionation as outlined by Kuo (1996) was utilized. The extraction method has been modified several times from the original fractionation scheme presented by Chang and Jackson (1957). A graphical representation of the scheme is demonstrated on Figure 1a.

The procedure began by weighing a 1.0 g soil sample, dry weight, into a 250 mL centrifuge tube followed by the addition of 50 mL 1 *M* NH_4Cl that was shaken for 30 minutes to extract the soluble and loosely bound phosphorus (LB/S-P). The soil and NH_4Cl was centrifuged and the supernatant was transferred to a 50 mL volumetric flask, which was brought to volume. A 0.5 *M* NH_4F (pH 8.2) solution was added to the residue and followed by a 1 hour shaking period to extract the aluminum phosphate fraction (Al-P). At 1 hour, the sample was centrifuged with the supernatant decanted into a 100 mL volumetric flask, the residue was washed twice with 25 mL portions of saturated NaCl. The washings and the extract were combined in the same volumetric flask and brought to volume. The next extractant of 50 mL of 0.1 *M* NaOH was then added to the residue and shaken for 17 hours to extract the iron phosphate (Fe-P) fraction. The residue was centrifuged, decanted into a 100 mL volumetric flask, and washed twice with 25 mL of saturated NaCl. Next, 40 mL of 0.3 *M* $Na_3C_6H_5O_7$ and 5.0 mL of 1 *M* $NaHCO_3$ were added to the residue and the suspension was heated in a water bath at 85°C. Once the suspension reached 85°C, 1.0 g of sodium dithionate, $Na_2S_2O_4$, was added and stirred rapidly to extract the reductant-soluble phosphorus (RS-P). The solution was allowed to heat for 15 minutes, then centrifuged. The supernatant was poured into a 100 mL volumetric and the residue was washed with saturated NaCl. The $Na_3C_6H_5O_7\backslash NaHCO_3\backslash Na_2S_2O_4$ solution was exposed to air to allow the $Na_2S_2O_4$ to oxidize. Finally, 50 mL of 0.25 *M* H_2SO_4 was added to the residue and shaken for 1 hour. The suspension was centrifuged for 10 minutes and transferred to a 100 mL volumetric flask. The residue was washed twice with 25 mL portions of saturated NaCl and combined with the H_2SO_4 extract to determine the calcium phosphate fraction (Ca-P). To determine phosphorus content in all fractions, extracts were analyzed using the colorimetric molybdenum-blue method of Murphy and Riley (1962).

Inorganic P Fractionation for Calcareous Soils

The procedure for calcareous P fractionation is outlined by Kuo (1996). This is an extraction method that has been modified several times from the original fractionation scheme presented by Chang and Jackson (1957). A modification to the scheme was made for the experimentation of this project. No current method provides a LB/S-P extraction for calcareous soils. Therefore, a 0.01 *M* $CaCl_2$ extraction was added for the extraction of the LB/S-P fraction. A dilute $CaCl_2$ solution is commonly used to determine available P in alkaline soils (Pierzynski, 1998). The complete method is summarized graphically on Figure 1b.

One gram of soil sample (dry weight) and 20 mL of 0.01 *M* $CaCl_2$ was added to a centrifuge tube for a 24 hour shake time. The sample was Then centrifuged at 6000 rpm for 10 minutes and filtered through a Whatman No. 42 filter paper to extract the LB/S-P. Then 50 mL of 0.1 *M* NaOH + 1 *M* NaCl were added followed by a 17 hour shake time to extract the Al-P and the Fe-P fractions simultaneously. After centrifuged and decanted into a 100 mL volumetric flask, the residue was washed twice with 25 mL portions of 1 *M* NaCl. The washings and the extract were combined in the same volumetric flask and brought to volume. Next, 40 mL of 0.3 *M* $Na_3C_6H_5O_7$ and 5 mL of 1 *M* $NaHCO_3$ were added to the residue and the suspension was heated in a water bath at 85°C. Once the suspension reached 85°C, 1.0 g of sodium dithionate was added and stirred rapidly to extract the RS-P. The solution was allowed to heat for 15 minutes and then centrifuged. The supernatant was poured into a 100 mL volumetric and the residue was washed 1 *M* NaCl. The $Na_3C_6H_5O_7\backslash NaHCO_3\backslash Na_2S_2O_4$ solution was exposed to air to allow the $Na_2S_2O_4$ to oxidize. Finally, 50 mL of 0.5 *M* HCl was added to the residue and shaken for 1 hour. The suspension was centrifuged for 10 minutes and transferred to a 100 mL volumetric flask. The residue was washed twice with 25 mL portions of saturated NaCl and combined with the HCl extract to determine the Ca-P. As with the non-calcareous soil, all extracts were analyzed using the Murphy and Riley (1962) ascorbic acid method.

Total P Determination

U.S. EPA Method 200.7 was performed on the soils, effluents, and sludge samples for total P determination (Smoley, 1992). Total P was measured to determine the amount of inorganic and organic P in the swine waste and to establish a baseline P concentration for the soils. The extraction and analysis was: 1.0 g of soil or sludge material is weighed into a beaker and 4 mL of 1:1 nitric acid and 10 mL of 1:4 hydrochloric acid were added to the sample. The beaker was heated on a hot plate to approximately 85°C and refluxed for 30 minutes. After cooling, the solution was quantitatively transferred to a 100 mL volumetric flask and diluted with DI water to volume. The extract is analyzed for P by ICP emission spectrometer.

Mehlich III Extractable P

To determine plant available P for each soil treatment, Mehlich III extractable P was measured for all soil incubations. The extraction process consisted of using 2.0 g soil (dry weight) and 20 mL of Mehlich III extractant. The solution is mixed for a 30 minute shake time. The Mehlich III solution was composed of $0.2M$ CH_3COOH, $0.25M$ NH_4NO_3, $0.015M$ NH_4F, $0.013M$ HNO_3, and $0.001M$ EDTA. The soils were immediately filtered through a Whatman 1-filter and analyzed using the Murphy and Riley (1962) colorimetric determination on spectrophotometer at 880 nm.

Soil Physical and Chemical Properties

Soil texture and percent clay were determined for the three soils and particle size distribution was conducted using the bouyoucos hydrometer method (Klute, 1986). Soil carbonates were measured on the soils using the acetic acid dissolution method, which is a titration procedure where a standard curve is established with fine-grained reference calcite. Soil pH was measured by a 1:2 soil water solution and soil CEC was determined for acid/neutral soils by a $BaCl_2$ extraction and the Palemio and Rhodes method for the calcareous soil. Finally, P sorption maximas were measured for the three soils using the Langmuir adsorption isotherm method. A 25 mL standard P solution is added to 1.0 g soil in a centrifuge tube. A 0.0, 0.2, 0.5, 1.0, 5.0, 10.0, 15.0, and 20.0 ppm P standard was utilized for this method. Soils were shaken for 24 hours, centrifuged at 10,000 rpm for 10 min. and measured on the spectrophotometer using the Murphy and Riley (1962) method. Measured values for the physical and chemical properties of the soil and swine effluent/sludge are provided in Tables 1 and 2.

RESULTS AND DISCUSSION

Total Phosphorus of Soils and Swine Effluents/Sludge

The total phosphorus (TP) contents for the three soil series and the three lagoon waste layers are displayed on Tables 1 and 2. The Dennis loam averaged 157 mg P kg^{-1} soil of TP, while the Bethany silt loam had a mean TP content of 346 mg P kg^{-1} soil and the Richfield clay loam of 261 mg P kg^{-1} soil. The P measured for the Bethany and Richfield soils were noticeably higher than the Dennis, likely because these samples were obtained from fertility research plots. The mean TP for the 0-2 ft. depth effluent, the 8-10 ft. depth effluent, and the swine sludge were 72.3, 68.3, and 2969 mg P L^{-1} waste, respectively. The effluent values are similar to reported P concentrations commonly found in Oklahoma of 1.2-1.3 lbs P_2O_5/1000 gallons of effluent, equivalent to 63-68 mg P L^{-1} effluent (Zhang and Hamilton, 1996). Two different lagoon effluent layers were utilized to evaluate if differences in chemical composition would alter the P sorption; however, throughout all the analysis performed, the two effluent layers effected the mineral fractions almost identically. The sludge contained higher concentrations of P due to the increased solid portion by which the P is bound chiefly in organic solids. The sludge was composed of approximately 10% solids and the effluents averaged 0.5-0.7% solids.

Soil Phosphorus Fractionation

The mean P fractionation concentration for time zero and 35 day incubated soils are recorded on Figure 2 through Figure 7, while the compiled laboratory data is provided in Tables I-1 through Table I-7. An increase in P fraction concentrations was observed with the addition of the swine wastes (with the exception of the RS-P fraction). In particular, the sludge incubations increased the LB/S-P, Al-P, Fe-P, and Ca-P considerably when compared to the effluent treated soils. These treatments were established on a moisture content basis; therefore, the soils receiving the swine sludge demonstrated increases in P fractions because more P was added to the soils in comparison to the effluent treatments.

The LB/S-P concentrations decreased at the 35 day incubation time for all unsaturated and saturated soils with the exception of the Bethany and Dennis unsaturated control treatments. The saturated Dennis soils treated with 0-2 ft. and 8-10 ft. depth lagoon effluent exhibited a decrease of the LB/S-P to undetectable levels (Figures 2a -d). The LB/S-P concentrations were initially 58.5 and 57.5 mg P kg^{-1} soil for the 0-2 and 8-10 ft. depth effluents respectively. However, following the 35 day incubation the LB/S-P was below the detection limit. The Fe-P and Al-P fractions displayed a significant increase, with little alteration of the Ca-P pool ($\alpha = 0.05$). A similar result was found by Nziguheba et al. (1998), where a decrease in labile P (resin extractable P) was measured in an acid soil over the period of several weeks after the addition of triple super phosphate and organic P sources. Yuan et al. (1960) investigated forms of newly fixed P after an incubation of three acid soils with the addition of various P concentrations. Utilizing the Chang and Jackson fractionation method, over 80% of the added P was retained as Al-P and Fe-P. The sorption of P into Fe-P and Al-P minerals pools (either by precipitation of iron/aluminum phosphates or chemisorption with iron/aluminum oxides) is typical for most acid soils at a pH of 5.5 and below (Sharpley, 1999).

The saturated and unsaturated Bethany silt loam demonstrated a behavior similar to that of the Dennis soils. For the saturated incubations, the LB/S-P decreased significantly for the effluent treated soils and the P was distributed predominantly into the Fe-P fraction. However, the concentrations following the incubation period were still detectable unlike those of the saturated effluent Dennis soils. At time zero, LB/S-P concentrations of 93.8 and 93.0 mg P kg^{-1} soil were measured for the 0-2 ft. effluent and the 8-10 ft. depth effluent treatments respectively. After the incubation period the LB/S-P fell to 24.8 and 16.6 mg P kg^{-1} soil (Figures 4b and 4c). A possible explanation for the differences in the LB/S-P concentrations between the Dennis and Bethany LB/S-P concentrations is the Bethany soil's higher total P concentration and lower sorption maxima (Table 1), which suggests a greater P saturation when compared to the lower total P and higher Langmuir sorption maxima of the Dennis loam. Similar results were found by Indiati et at. (1995) in a three month incubation with three Italian soils. As P additions and P contents increased, available P increased dramatically for the soils possessing a lower sorption capacity.

The calcareous Richfield clay loam soil exhibited a decline in LB/S-P after the 35 day incubation. For the saturated two effluent treatments and the sludge treatment, the initial concentrations of the LB/S-P fraction were 45.5, 48.9, and 240.9 mg P kg^{-1} soil, respectively. The concentrations decreased to 0.9, 0.3, and 29.1 mg P kg^{-1} soil after 35 days as in illustrated Figures 6b-d. There was also a significant increase in the Fe+Al-P fraction as a result of the P in the swine wastes for the soils. The unsaturated treatments reflected a similar trend as the saturated Richfield soils by a reduction of LB/S-P and an increase of the Fe+Al-P fraction. The Richfield contained more P in the Ca-P pool than any other fraction which is similar to other calcareous soils (Chang and Jackson, 1958; Williams et al., 1971; Hedley et al., 1982; Sharpley and Smith, 1985).

Shifts in P for the unstaturated and saturated conditions were also of interest for this study. These conditions would occur as lagoon levels rise and fall or as sub-surface irrigation systems are turned on or off. A comparison of how P sorption differed between the saturated and unsaturated treatments is documented on Table 3. The percentages represent the amount of each individual fraction from the summation of all extracted fractions. When comparing the two moisture contents within each P fraction of the individual soils, the LB/S-P exhibited large percentage changes in the Dennis and Bethany soils that were significant at the $\alpha = 0.05$ probability level. Under saturated conditions, less LB/S-P was measured for all control and effluent treated soils while more LB/S-P was detected for the unsaturated soils. The Dennis soil displayed the greatest decrease in LB/S-P for the saturated treatments with declines ranging between 15.1-35.9% (Table 3). This is similar to a field experiment investigated by Khalid et al. (1977) in which 14 of 20 Louisiana soils studied sorbed more available P under saturated conditions than unsaturated conditions. Khalid et al. (1977) concluded that a greater surface area generated by the transformation of ferric soil minerals to more reactive ferrous minerals may be responsible for the greater P sorption. The Richfield displayed relatively no difference in the LB/S-P fraction ratio under saturated and unsaturated conditions (Table 3). The low to negligible concentrations of LB/S-P before and after the 35 day incubation suggests that the P was sorbed into stable mineral pools (Fe+Al-P and Ca-P), thus reducing the environmental impact potential in the case of a lagoon failure.

For all soils, the Al-P and/or the Fe-P fractions were significantly different for the saturated and unsaturated moisture contents. The Al-P and Fe-P fractions presented the greatest percentage change of the five measured fractions when evaluating the differences in P partitioning between the saturated and unsaturated soils. For instance, the Fe-P fraction for the Dennis loam was substantially altered when the soil was saturated. With Fe-P percentages of 38.2%, 46.0%, and 38.1% of the extracted P for the unsaturated control, 0-2, and 8-10 ft. effluent treatments; the percentages rose to 69.2%, 66.9%, and 71.4% for the saturated treatments, respectively (Table 3). This demonstrates that when higher P concentrations are added in saturated conditions, soils containing Fe and Al mineral pools could sorb large amounts of P from solution. This is consistent with Amer et al. (1991) who observed that P sorption capacity of saturated soils increased with time of submergence and was highly correlated with the presence of iron minerals. Sah and Mikkelsen (1986) also found a similar outcome in an incubation study of three flooded soils. Of the P added to the soils, most was recovered in the Al-P and Fe-P fractions, which indicated that Fe and Al mineral sinks were the most reactive soil components of P.

For the majority of the Dennis, Bethany, and Richfield soils, the Ca-P pool was also significantly different for the saturated and unsaturated moisture contents. The Ca-P percentages for the saturated soils were significantly lower than the unsaturated soils (Table 3). In particular, the calcareous Richfield clay loam exhibited the greatest difference in Ca-P between the moisture contents with 15.5%, 17.5%, and 14.6% less for the saturated 0-2, 8-10 ft., and sludge treated soils, respectively. This decrease in Ca-P has not been previously reported, it could be a dissolution of Ca-P minerals during the flooding process with a decrease in system pH. In the conditions Ca minieral dissolution would be affected by pH and the pCO_2 in the system. Another explanation could be attributed to the greater sorption ability of the Fe-P and Al-P fractions in comparison to the Ca-P. In the case of the saturated soils where more P was added to the soils than the unsaturated soils, this effect could be observed more noticeably. Previous work clearly demonstrates that Fe and Al mineral pools are the major source of P sorption capacity in most soils (Sparks, 1995; Sposito, 1989).

When comparing the individual P fraction percentages among the different swine waste treatments, often the control and effluent treated soils are significantly different than the fraction percentages of the sludge treated soils. In Table 4, note that the sludge incubated soils were statistically different ($\alpha = 0.05$) than the three other treatments, with the exception of the Bethany unsaturated soil. As stated previously, the LB/S-P fraction appears to be the fraction most greatly affected by the sludge addition by substantially increasing this labile and potentially mobile pool. Furthermore, with a large increase in the LB/S-P fraction percentage, the other P fraction percentages decrease with the Fe-P displaying the most significant decline within the noncalcareous soils. A reason for this consistent decrease in the Fe-P pool cannot be determined.

As more P is supplied in the calcareous Richfield soil, the Ca-P fraction decreases markedly. The saturated Richfield control contained a Ca-P extractable P percentage of 91.3%; however, the Ca-P pool decreased to 72.5%, 70.2%, and 55.7% for the 0-2 ft. effluent, 8-10 ft. effluent, and sludge treatments (Table 4). The majority of the P appears to be distributed into the Fe+Al-P fraction with a statistically significant increase in the LB/S-P fraction as well.

Environmental Implications

The results of this incubation study can be applicable in the circumstance of a subsurface lagoon failure. The seeping waste would create a wetting front where a saturated moisture condition would be present. Behind the wetting front would be an unsaturated moisture condition. Downward movement of the waste would be facilitated by the presence of a hydraulic head from the above lagoon effluent. Under saturated and unsaturated conditions all three soils demonstrated the ability to significantly reduce the concentration of LB/S-P over the 35 day period and distribute the added P into stable Al-P, Fe-P, and Ca-P fractions (Figure 2 through Figure 7). The zero time incubations also showed the capacity to quickly sorb the swine waste P into mineral pools beside the LB/S-P. For instance, Figure 4 shows a marked increase in Al-P and Fe-P fractions for the Bethany soil and each swine waste source. The Al-P fraction of the Bethany displayed the most significant increase with a control concentration of 93.0 mg P kg^{-1} soil and subsequent concentrations of 149.2, 150.0, and 460.7 mg P kg^{-1} soil for the 0-2 ft., 8-10 ft. effluent, and sludge treated soils.

As stated earlier, a difference in P distribution was observed for the saturated and unsaturated soils. Substantially greater concentrations of LB/S-P were detected for the unsaturated effluent soils than the saturated effluent soils for the Dennis and Bethany. The larger concentrations in potentially mobile LB/S-P in an unsaturated soil could pose a threat to groundwater quality. However, the concern is inconsequential due to observed ability of the saturated soils to sorb large amounts of soluble P (saturated conditions would be present if the waste leached to the groundwater table). The saturated and unsaturated soils distributed the manure P chiefly into the stable and immobile Fe-P and

Al-P fractions. In areas where swine facilities are located near P sensitive waterbodies, the presences of soils rich in Fe and Al mineral sinks would be most suitable to underlie waste lagoons.

The sludge treated soils all exhibited much larger fraction concentrations than the 0-2 and 8-10 ft. effluent treated soils. Though the concentrations are very high, the risk of the LB/S-P to groundwater quality is minimal. Past studies have proven that a structural failure near the bottom of the lagoon would essentially repair itself by plugging the crack or rip of the lagoon bed with the organic solids in the sludge (Barrington, 1985; Ritter et al., 1984). The elevated concentrations of LB/S-P would be localized with little threat of downward movement due to the self-sealing of the lagoon.

Agricultural Implications

From an agricultural production perspective, the unsaturated treatments would represent an application equivalent to 1.1 to 1.4 acre-inches of the swine waste (assuming an acre-furrow slice weighs 2.0×10^8 lbs). From the results, the unsaturated sludge treatments would add sufficient amounts of LB/S-P to soils that would be immediately available for growing crops. The swine effluent treatments altered the LB/S-P concentrations of the three soils slightly and would required continuous applications to supply crops with adequate concentrations of LB/S-P. Although most applied P is sorbed into mineral fractions that are nonlabile, as the Al-P, Fe-P, and Ca-P fractions increase so the LB/S-P will increase as well. Figures 8a and 8b demonstrate the relationship between total sorbed P and LB/S-P fractions. The linear regression plots display a trend of increasing LB/S-P with an increase in Al, Fe, and Ca-P pools for both the 35 day and the zero incubation time (R^2= 0.68 for the 35 day and an R^2= 0.65 for time zero treatments). The differences in slope coefficients of the two linear equations are due to the increased sorption of P after the 35 day incubation. With a slope of 2.11 for the 35 day and 0.98 for the time zero, more than a two-fold increase in P sorption occurred after the incubation period. The observed positive relationship between the LB/S-P and Al, Fe, and Ca-P pools is likely due to the saturation of these fractions; therefore, as the inorganic P sinks begin to saturate, more labile P will be available for plant uptake.

Mehlich III extractable P was measured for all soils to measure the plant available P. The Mehlich III extractable P (M3-P) concentrations followed the same trend as the fractionation extractions. Table 4 and Table 5 demonstrate how the soils treated with the effluents and sludge increased in M3-P when compared to the controls. Also, over the incubation period the extractable P decreased for all treatments except for two control soils. More M3-P was measured for the saturated and unsaturated Bethany soil treatments than the Dennis and Richfield soils. This observation coincides with the trend found for the fractionation extraction where more P was detected from the summation of the LB/S-P, the Al-P, Fe-P, RS P, and the Ca-P. For example on Figure 2 through Figure 7, greater concentrations of total extractable P, the summation of all fractions, were measured for the Bethany soils. Although the control P concentrations were nearly twice of that found for the other two control soils, the total extractable P was two to three times of that extracted in the Dennis or Richfield.

CONCLUSIONS

The sorption of swine waste P by soils can differ depending on soil properties (soil pH, adsorption potential, texture, etc.), moisture content, and residence time of applied P. All three soils used in the project demonstrated the potential to sorb high concentrations of P which was distributed into chemically stable fractions like Al-P, Fe-P, and Ca-P. After a 35 day incubation time, the LB/S-P pool decreased for all saturated and unsaturated soils treated with swine effluent and sludge. The unsaturated soils actually possessed more LB/S-P than the saturated soils following the incubation, despite the fact that the saturated soils received more than four times the added P than the unsaturated.

Although phosphorus fractionation methods are not extremely precise, the methods are helpful in examining what mineral pools are affected by the addition of P. Consistently throughout the project, the Al-P and/or the Fe-P fractions significantly increased after the incubation for nearly every soil treated with the swine wastes. Even the calcareous Richfield soil exhibited this trend. The Al-P and Fe-P pools appear to be the fractions that contribute the most to a soil's P sorption potential.

As swine production rises in Oklahoma and throughout the United States, waste management issues dealing with P will become increasingly more important. From a crop production standpoint, applications of swine sludge would provide a substantial amount of immediate available P to growing plants. Repeated applications of swine effluent would likely be required to increase LB/S-P soil levels in P deficient soils. In addition, the soils ability to distribute P into immobile mineral sinks is beneficial in the circumstance of a lagoon failure. The threat to groundwater quality is minimal when soils underlying swine lagoons have the ability to sorb P into the Al-P, Fe-P, and Ca-P pools. The calcareous Richfield soil displayed the ability to sorb virtually all LB/S-P into stable mineral pools (Fe+Al-P and Ca-P) under saturated and unsaturated conditions, thus reducing the environmental impact potential in the case of a lagoon failure. The Richfield and other similar calcareous soils exist throughout the Oklahoma Panhandle; therefore, the risk of P leaching into deep groundwater formations in this region is highly unlikely.

REFERENCES

Amer, F., M.E. Saleh, and H.E. Mostafa. 1991. Phosphate behavior in submerged calcareous soils. Soil Sci. 151:306-311.

Barrington, S. 1985. The sealing of soils by manure. Ph.D. thesis, p. 430. McGill Univ., Montreal, Quebec.

Calvert, D.V. 1975. Nitrate, phosphate, and potassium movement into drainage lines under three soil management practices. J. Environ. Qual. 4:183-186.

Chang, S.C., and M.L. Jackson. 1957. Fractionation of soil phosphorus. Soil Sci. 84:133-144.

Chang, S.C., and M.L. Jackson. 1958. Soil phosphorus fractions in some representative soils. J. Soil Sci. 9:109-119.

Ciravolo, T.G., D.C. Martens, D.L. Hallock, E.R. Collins, Jr., E.T. Kornegay, and H.R. Thomas. 1979. Pollutant movement to shallow ground water tables from anaerobic swine waste lagoons. J. Environ. Qual. 8:126-130.

Hedley, M.J., J.W.B. Stewart, and B.S. Chauhan. 1982. Changes in inorganic and organic soil phosphorus fractions induced by cultivation practices and by laboratory incubations. Soil Sci. Soc. Am. J. 46:970-976.

Hieltjes, A. H. M., and L. Lijklema. 1980. Fractionation of inorganic phosphates in calcareous sediments. J. Environ. Qual. 9:405-407.

Indiati, R., A.N. Sharpley, C. Izza, A. Figliolia, B. Felici, and P. Sequi. 1995. Soil phosphorus sorption and availability as a function of high phosphorus fertilizer additions. Commun. Soil Sci. Plant Anal. 26:1863-1872.

Izuno, F.T., C.A. Sanchez, F.J. Coale, A.B. Bottcher, and D.B. Jones. 1991. Phosphorus concentrations in drainage water in the Everglades Agricultural Area. J. Environ. Qual. 20:608-619.

Khalid, R.A., W.H. Patrick, Jr., and R.D. DeLaune. 1977. Phosphorus sorption characteristics of flooded soils. Soil Sci. Soc. Am. J. 41:305-310.

Klute, A. 1986. Methods of soil analysis: Part 1-physical and mineralogical methods. ASA and SSSA, Madison, WI.

Kuo, S. 1996. Phosphorus. p. 869-919. *In* D.L. Sparks et al. (ed.) Methods of soil analysis, Part 3. 2nd ed. ASA and SSSA, Madison, WI.

Mallin, M.A., J.M. Burkholder, M.R. McIver, G.C. Shank, H.B. Glascow, Jr., B.W. Touchette, and J. Springer. 1997. Comparative effects of poultry and swine waste lagoon spills on the quality of receiving streamwaters. J. Environ. Qual. 26:1622-1631.

Murphy, J., and J.P. Riley. 1962. A modified single solution method for the determination of phosphate in natural waters. Anal. Chim. Acta. 27:31-36.

Nziguheba, G., C.A. Palm, R.J. Buresh, and P.C. Smithson. 1998. Soil phosphorus fractions and adsorption as affected by organic and inorganic sources. Plant and Soil. 198:159-168.

Peterson, G.W., and R.B. Corey. 1966. A modified Chang and Jackson procedure for routine fractionation of inorganic soil phosphates. Soil Sci. Soc. Am. Proc. 30:563-565.

Pierzynski, G.M. (ed.) 1998. Methods of phosphorus analysis for soils, sediments, residuals, and waters. USDA-CSREES Regional Committee, Bull. 30. Aug. 1, 1998.

Ritter, W.F., E.W. Walpole, and R.P. Eastburn. 1984. Effect of anaerobic swine lagoon on groundwater quality in Sussex County, Delaware. Agric. Wastes. 10:267-284.

Sah, R.N., and D.S. Mikkelsen. 1986. Transformations of inorganic phosphorus during the flooding and draining cycles of soil. Soil Sci. Soc. Am. J. 50:62-67.

Sharpley, A.N. 1999. Phosphorus availability. p. D18-D38. *In* M.E. Sumner (ed.) Handbook of Soil Science. CRC Press, Boca Raton, FL.

Sharpley, A.N., and S.J. Smith. 1985. Fractionation of inorganic and organic phosphorus in virgin and cultivated soils. Soil Sci. Soc. Am. J. 49:127-130.

Smoley, C.K. 1992. Methods for the determination of metals in environmental samples. Environ. Monitoring Systems Lab. U.S. EPA. CRC Press, Boca Raton, FL.

Sparks, D.L. 1995. Environmental Soil Chemistry. Academic Press, San Diego, CA.

Sposito, G. 1989. The Chemistry of Soils. Oxford University Press, New York, NY.

USDA-NRCS Soil Survey Division. 1996. Official soil series descriptions data access [Online]. Available at http://www.statlab.iastate.edu/soils/osd/ (verified 19 May, 1998.)

Westerman, P.W., R.L. Huffman, and J.S. Feng. 1995. Swine-lagoon seepage in sandy soil. Trans. ASAE 38:1749-1760.

Williams, J.D.H., J.K. Syers, R.F. Harris, and D.E. Armstrong. 1971. Soil Sci. Soc. Proc. 35:250-255.

Withers, P.J.A., H.G. McDonald, K.A. Smith, and C.G. Chumbley. 1998. Behaviour and impact of cow slurry beneath a storage lagoon: 1. Groundwater contamination 1975-1982. Water Air Soil Pollut. 107:35-49.

Yuan, T.L., W.K. Robertson, and J.R. Nellar. 1960. Forms of newly fixed phosphorus in three acid soils. Soil Sci. Soc. Proc. 24:447-450.

Zhang, H.A., and D.W. Hamilton. 1996. Using lagoon effluent as fertilizer. Div. Ag. Sci. & Natural Resources. F-2245. Oklahoma State Univ., Stillwater, OK.

Table 1. Selected physical and chemical properties for the Dennis, Bethany, and Richfield soils.

Soil	Texture[a]	Clay	pH[b]	CEC[c]	$CaCO_3$ equiv.[d]	Total P[e]	P sorption max[f]
		%		$cmol_c\ kg^{-1}$		$mg\ kg^{-1}$	$mg\ kg^{-1}$
Dennis	loam	25	5.7	10.4	0.1	157	238
Bethany	silt loam	20	7.2	14.8	0.7	346	209
Richfield	clay loam	45	8.2	33.5	6.8	261	183

[a] texture determined using bouyoucos hydrometer method

[b] pH determined using 1:2 soil water solution

[c] $BaCl_2$ extraction for acid/neutral soil; Palemio and Rhodes method for calcareous soil

[d] acetic acid dissolution method

[e] EPA Method 200.7 using HNO_3/HCl digestion

[f] Pmax determined from Langmuir adsorption isotherm

Table 2. Total P and pH for the 0-2 ft. effluent, 8-10 ft. depth lagoon effluent, and swine sludge.

Swine Waste Source	Total P[a]	pH[b]
	$mg\ kg^{-1}$	
0-2 ft. effluent	72.33	6.8
8-10 ft. effluent	68.33	6.7
Sludge	2969	6.9

[a] EPA Method 200.7 using HNO_3/HCl digestion

[b] pH determined using 1:2 soil water solution

Table 3. Changes in phosphorus fractions extracted for saturated and unsaturated soils treated with each swine waste source after 35 day incubation.[†]

Noncalcareous P Fractionation Procedure

Soil Series	Treatment	Loosely Bound/ Soluble P	Al-P	Fe-P	Reductant Soluble P	Ca-P
		----------% of P extracted from the sum of all fractions----------				
Dennis	control, unsat.	31.5%a	21.2%a	38.2%a	0.0%a	9.1%a
	control, sat.	0.0%b	25.3%a	69.2%b	0.0%a	5.4%b
	0-2 ft. effluent, unsat.	15.1%a	35.6%a	46.0%a	0.0%a	8.4%a
	0-2 ft. effluent, sat.	0.0%b	28.9%a	66.9%b	0.0%a	4.2%b
	8-10 ft. effluent, unsat.	35.9%a	26.0%a	38.1%a	0.0%a	8.7%a
	8-10 ft. effluent, sat.	0.0%b	23.9%a	71.4%b	0.0%a	4.7%b
	sludge, unsat.	24.3%a	45.7%a	25.9%a	0.0%a	4.1%a
	sludge, sat.	34.7%b	37.5%a	24.4%a	0.0%a	3.3%a
Bethany	control, unsat.	21.2%a	37.3%a	16.3%a	0.0%a	25.2%a
	control, sat.	3.2%b	41.8%a	26.4%b	0.0%a	28.6%a
	0-2 ft. effluent, unsat.	19.3%a	35.9%a	18.0%a	0.0%a	26.8%a
	0-2 ft. effluent, sat.	7.5%b	45.1%b	25.7%b	0.0%a	21.7%a
	8-10 ft. effluent, unsat.	19.3%a	37.4%a	17.2%a	0.0%a	26.1%a
	8-10 ft. effluent, sat.	4.8%b	49.6%b	26.6%b	0.0%a	19.0%b
	sludge, unsat.	23.8%a	50.7%a	12.2%a	0.0%a	13.3%a
	sludge, sat.	45.4%b	37.9%b	10.7%a	0.0%a	6.0%b

Calcareous P Fractionation Procedure

Soil Series	Treatment	Loosely Bound/ Soluble P	Al+Fe-P	Reductant Soluble P	Ca-P
		----------% of P extracted from the sum of all fractions----------			
Richfield	control, unsat.	0.0%a	6.7%a	0.0%a	93.3%a
	control, sat.	0.0%a	8.7%b	0.0%a	91.3%b
	0-2 ft. effluent, unsat.	0.0%a	11.9%a	0.0%a	88.0%a
	0-2 ft. effluent, sat.	1.5%b	27.4%b	0.0%a	72.5%b
	8-10 ft. effluent, unsat.	0.4%a	11.9%a	0.0%a	87.7%a
	8-10 ft. effluent, sat.	0.2%a	29.6%b	0.0%a	70.2%b
	sludge, unsat.	6.1%a	23.6%a	0.0%a	70.3%a
	sludge, sat.	5.2%a	39.0%b	0.0%a	55.7%b

[†] Within each soil treatment having unsaturated and saturated moisture contents, the P fraction percentages displaying the same letter are not significantly different ($P < 0.05$) as determined by analysis of variance.

Table 4. Comparison of phosphorus fractions extracted among the different swine waste treatments for the 35 day incubations.[†]

Noncalcareous P Fractionation Procedure

Soil Series	Treatment	Loosely Bound/ Soluble P	Al-P	Fe-P	Reductant Soluble P	Ca-P
		----------% of P extracted from the sum of all fractions----------				
Dennis	control, unsat.	31.5%a	21.2%a	38.2%a	0.0%a	9.1%a
	0-2 ft. effluent, unsat.	15.1%b	35.6%b	46.0%b	0.0%a	8.4%a
	8-10 ft. effluent, unsat	35.9%a	26.0%a	38.1%a	0.0%a	8.7%a
	sludge, unsat.	24.3%c	45.7%c	25.9%c	0.0%a	4.1%b
	control, sat.	0.0%a	25.3%ab	69.2%a	0.0%a	5.4%a
	0-2 ft. effluent, sat.	0.0%a	28.9%b	66.9%a	0.0%a	4.2%ab
	8-10 ft. effluent, sat.	0.0%a	23.9%a	71.4%a	0.0%a	4.7%a
	sludge, sat.	34.7%b	37.5%c	24.4%b	0.0%a	3.3%b
Bethany	control, unsat.	21.2%a	37.3%a	16.3%a	0.0%a	25.2%a
	0-2 ft. effluent, unsat.	19.3%a	35.9%a	18.0%a	0.0%a	26.8%a
	8-10 ft. effluent, unsat	19.3%a	37.4%a	17.2%a	0.0%a	26.1%a
	sludge, unsat.	23.8%a	50.7%b	12.2%b	0.0%a	13.3%b
	control, sat.	3.2%a	41.8%a	26.4%a	0.0%a	28.6%a
	0-2 ft. effluent, sat.	7.5%a	45.1%a	25.7%a	0.0%a	21.7%b
	8-10 ft. effluent, sat.	4.8%a	49.6%b	26.6%a	0.0%a	19.0%b
	sludge, sat.	45.4%b	37.9%c	10.7%b	0.0%a	6.0%c

Calcareous P Fractionation Procedure

Soil Series	Treatment	Loosely Bound/ Soluble P	Al+Fe-P	Reductant Soluble P	Ca-P
		----------% of P extracted from the sum of all fractions----------			
Richfield	control, unsat.	0.0%a	6.7%a	0.0%a	93.3%a
	0-2 ft. effluent, unsat.	0.0%a	11.9%b	0.0%a	88.0%a
	8-10 ft. effluent, unsat	0.4%b	11.9%b	0.0%a	87.7%a
	sludge, unsat.	6.1%c	23.6%c	0.0%a	70.3%b
	control, sat.	0.0%a	8.7%a	0.0%a	91.3%a
	0-2 ft. effluent, sat.	1.5%b	27.4%b	0.0%a	72.5%b
	8-10 ft. effluent, sat.	0.2%a	29.6%b	0.0%a	70.2%b
	sludge, sat.	5.2%c	39.0%c	0.0%a	55.7%c

[†] Within each P fraction, the fraction percentages displaying the same letter are not significantly different ($P < 0.05$) as determined by analysis of variance.

Table 5. Mean Mehlich III extractable P concentrations (mg P kg soil^{-1}) of unsaturated incubations treated with each swine waste type

Soil Series	Soil pH		Time Zero	35 Day
Dennis	5.7	control	5.7a	5.0a
		0-2 ft. depth effluent	13.1a	8.5b
		8-10 ft. depth effluent	12.0a	8.0b
		sludge	261.7a	200.0b
Bethany	7.2	control	111.7a	114.7a
		0-2 ft. depth effluent	133.3a	121.3a
		8-10 ft. depth effluent	136.0a	126.3a
		sludge	529.0a	382.3b
Richfield	8.2	control	16.0a	17.5a
		0-2 ft. depth effluent	32.0a	28.5a
		8-10 ft. depth effluent	32.3a	27.7a
		sludge	419.0a	376.5b

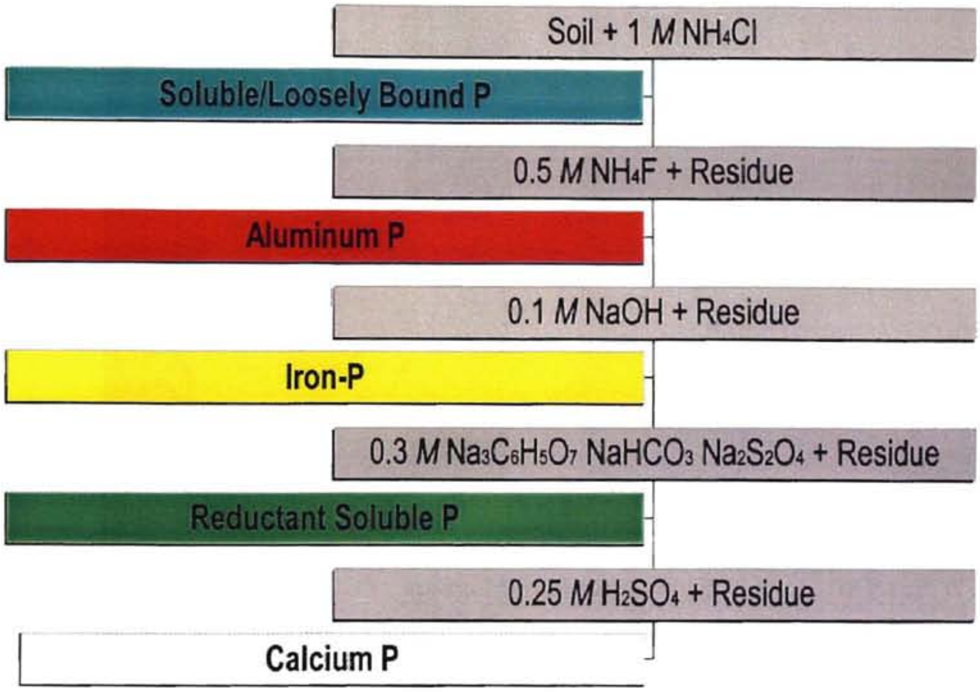
[†] Within each treatment, the P concentration displaying the same letter are not significantly different ($P < 0.05$) as determined by analysis of variance.

Table 6. Mean Mehlich III extractable P concentrations (mg P kg soil^{-1}) of saturated incubations treated with each swine waste type

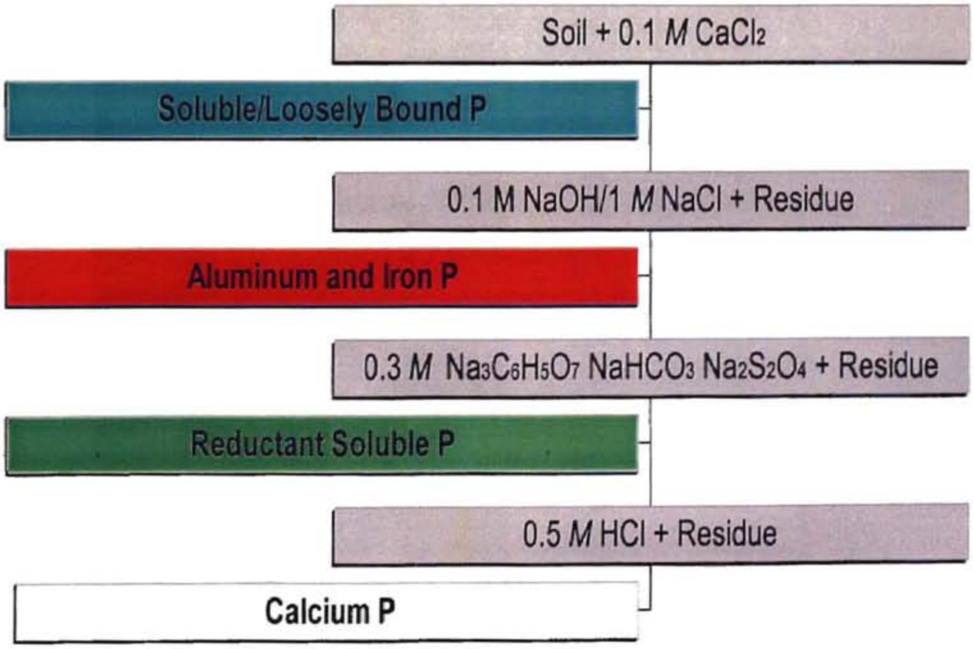
Soil Series	Soil pH		Time Zero	35 Day
Dennis	5.7	control	4.5a	3.0a
		0-2 ft. depth effluent	22.7a	11.0b
		8-10 ft. depth effluent	21.7a	8.0b
		sludge	1225.3a	856.0b
Bethany	7.2	control	110.3a	101.3a
		0-2 ft. depth effluent	192.3a	173.7b
		8-10 ft. depth effluent	194.0a	151.0b
		sludge	1684.0a	1265.7b
Richfield	8.2	control	17.3a	19.0a
		0-2 ft. depth effluent	85.0a	58.0b
		8-10 ft. depth effluent	87.3a	72.3b
		sludge	1638.0a	1490.7b

[†] Within each treatment, the P concentration displaying the same letter are not significantly different ($P < 0.05$) as determined by analysis of variance.

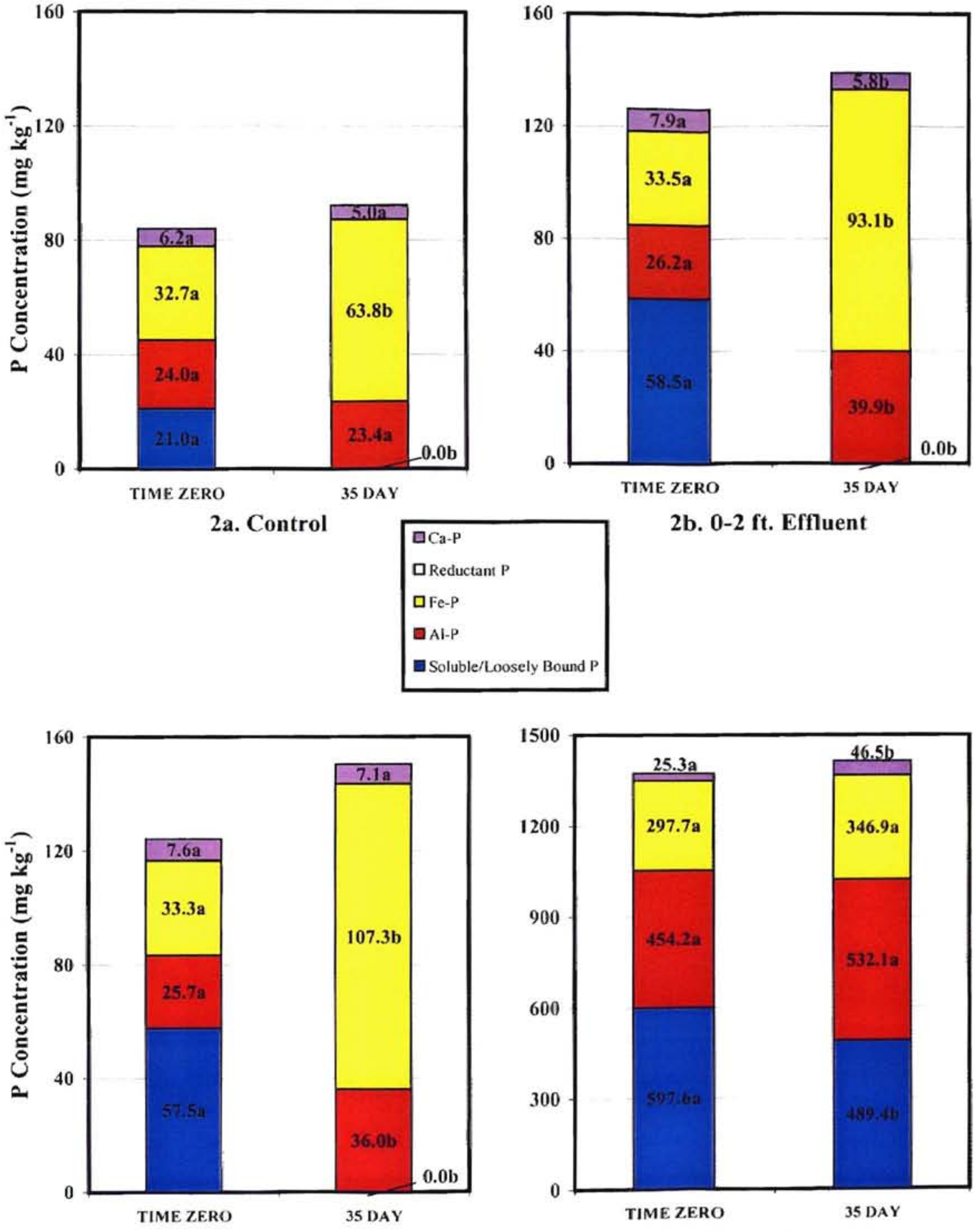
a. Phosphorus Fractionation for Noncalcareous Soils



b. Phosphorus Fractionation for Calcareous Soils



Figures 1a-b. Inorganic phosphorus fractionation schemes for noncalcareous and calcareous soils.



2a. Control

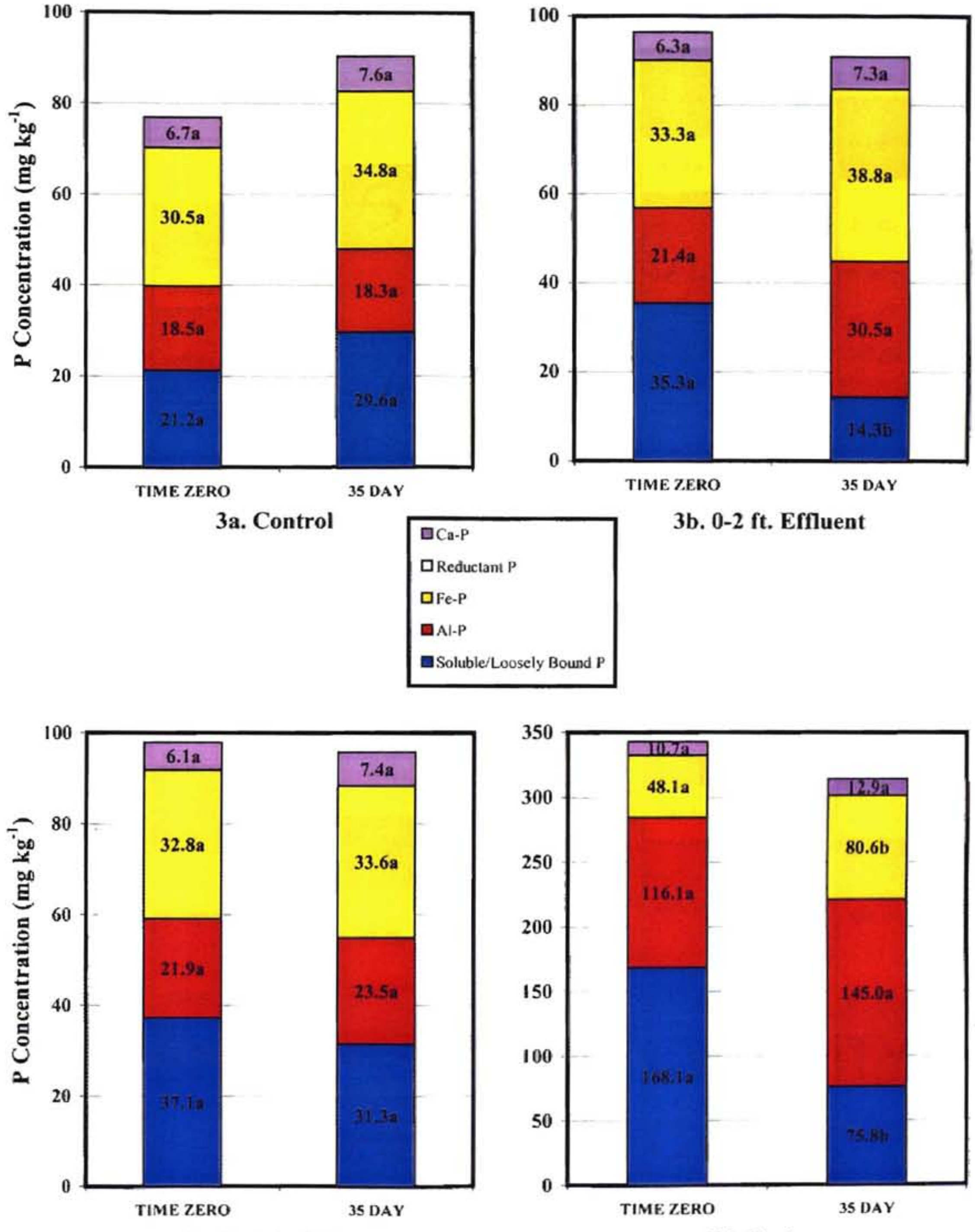
2b. 0-2 ft. Effluent

2c. 8-10 ft. Effluent

2d. Sludge

Figures 2a,b,c,d. P fraction concentration changes for saturated Dennis incubations.[†]

[†] Within each waste source, the P concentration of each fraction pool displaying the same letter are not significantly different ($P < 0.05$).



3a. Control

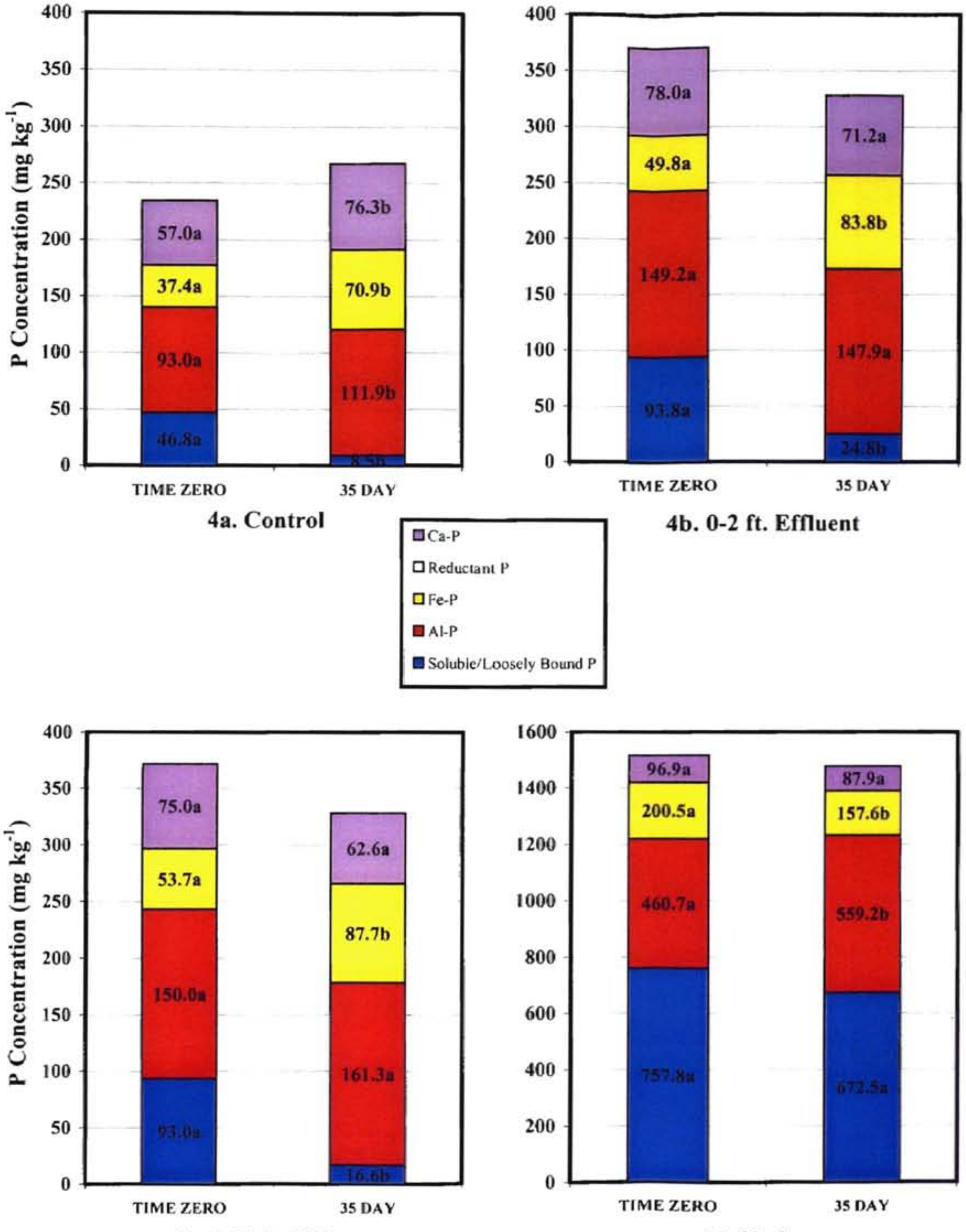
3b. 0-2 ft. Effluent

3c. 8-10 ft. Effluent

3d. Sludge

[†] Within each waste source, the P concentration of each fraction pool displaying the same letter are not significantly different ($P < 0.05$).

Figures 3a,b,c,d. P fraction concentration changes for unsaturated Dennis incubations.[†]



4a. Control

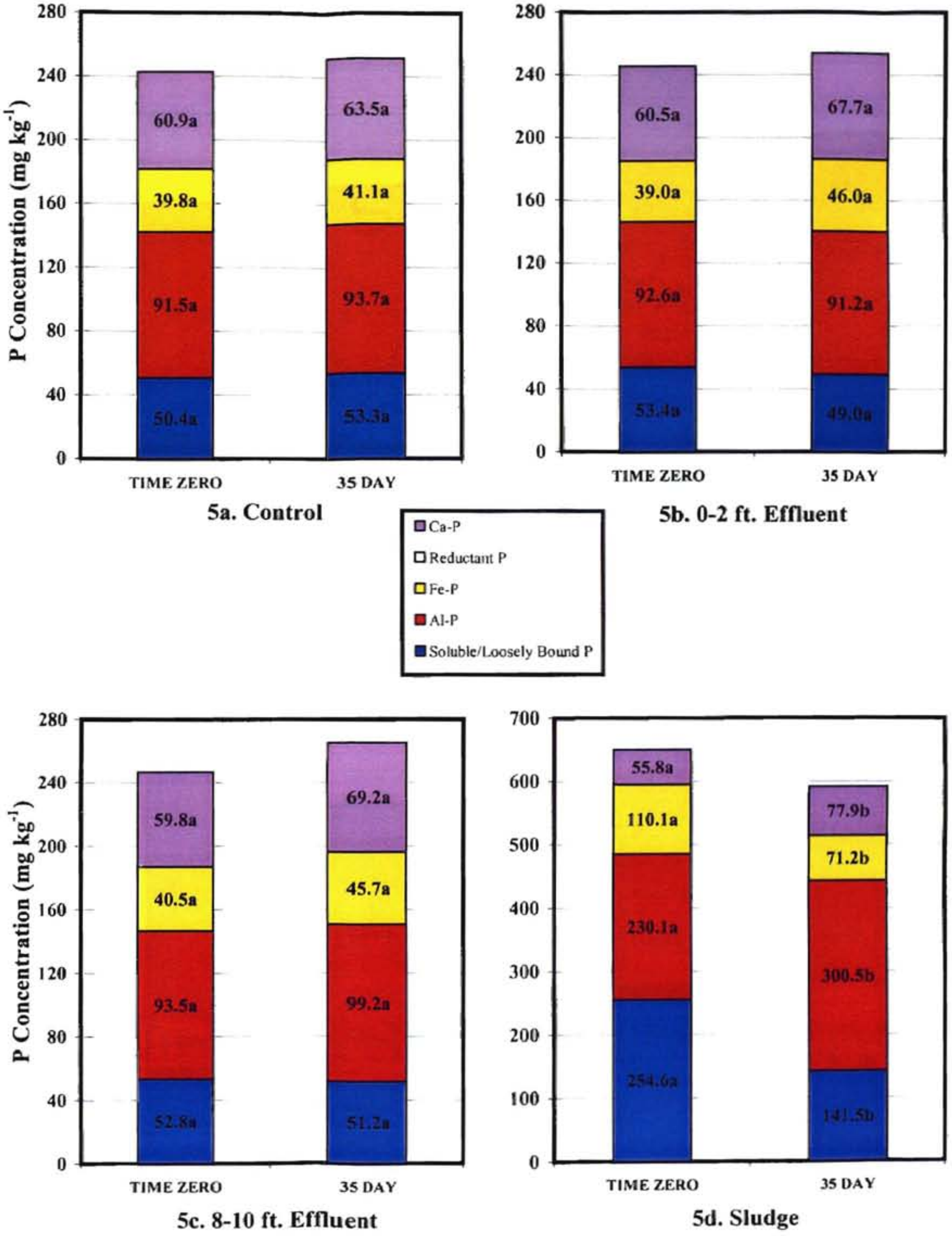
4b. 0-2 ft. Effluent

4c. 8-10 ft. Effluent

4d. Sludge

† Within each waste source, the P concentration of each fraction pool displaying the same letter are not significantly different ($P < 0.05$).

Figures 4a,b,c,d. P fraction concentration changes for saturated Bethany incubations.†



5a. Control

5b. 0-2 ft. Effluent

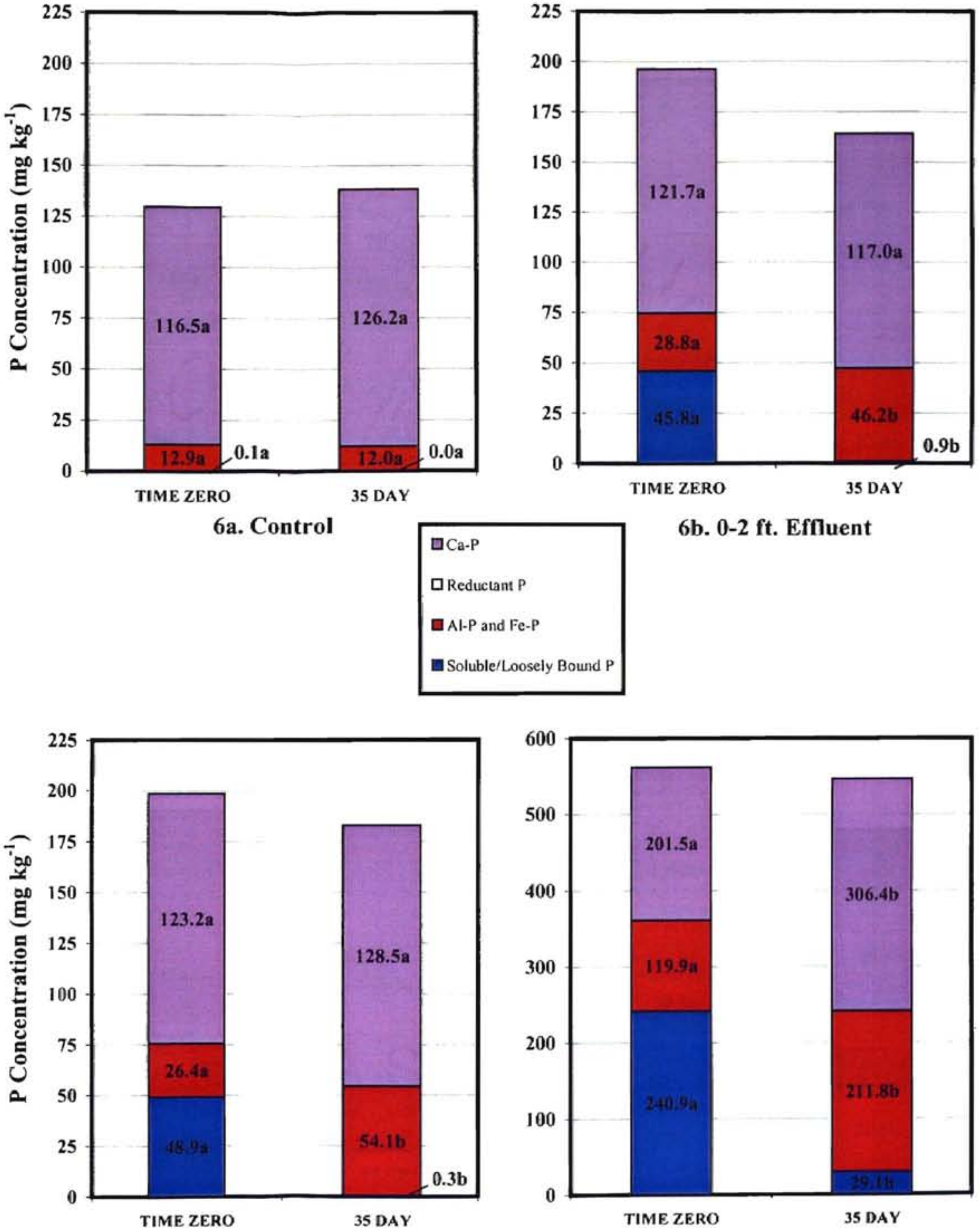
5c. 8-10 ft. Effluent

5d. Sludge

Figures 5a,b,c,d. P fraction concentration changes for unsaturated Bethany incubations.†

---

† Within each waste source, the P concentration of each fraction pool displaying the same letter are not significantly different ($P < 0.05$).



6a. Control

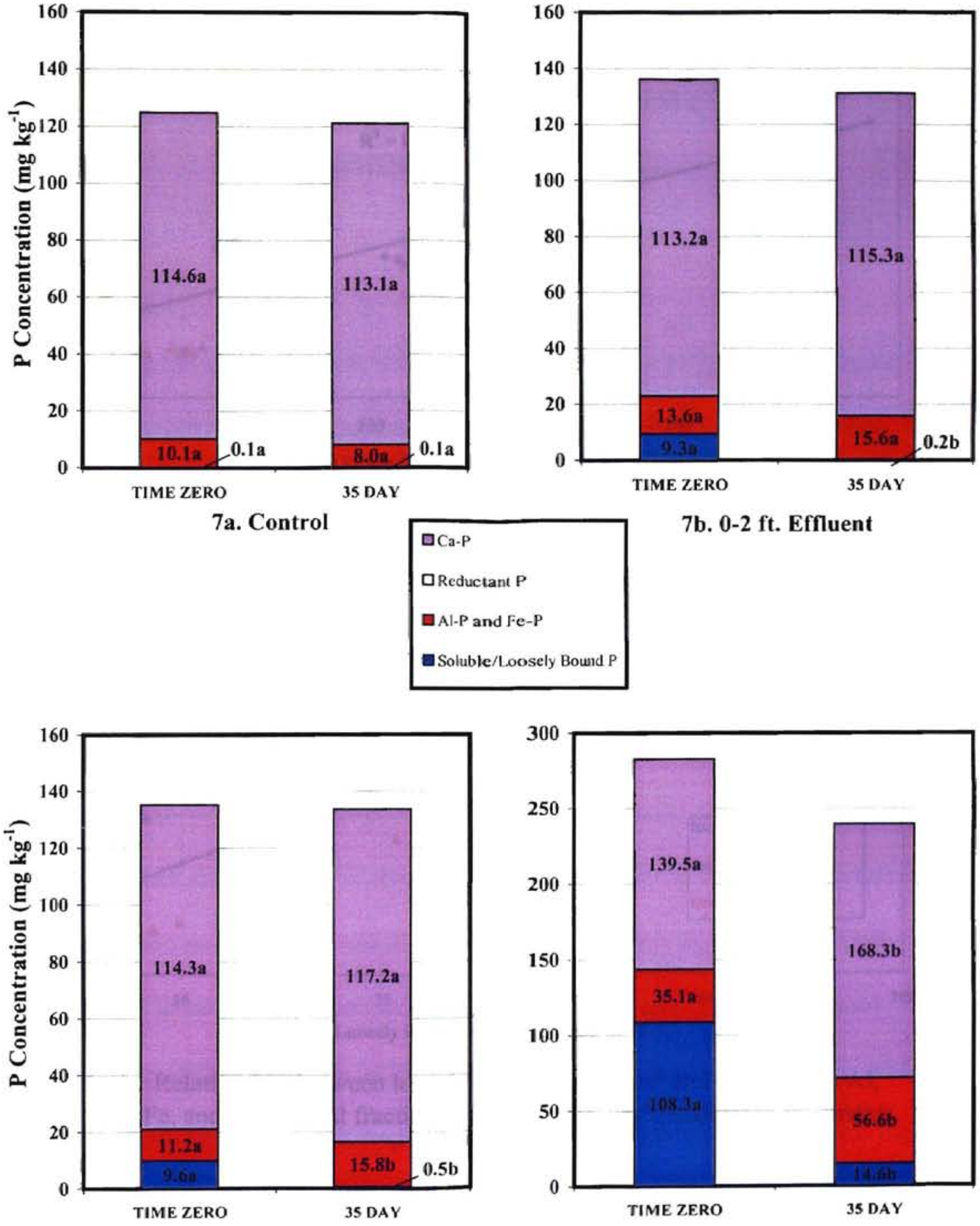
6b. 0-2 ft. Effluent

6c. 8-10 ft. Effluent

6d. Sludge

† Within each waste source, the P concentration of each fraction pool displaying the same letter are not significantly different ($P < 0.05$).

Figures 6a,b,c,d. P fraction concentration changes for saturated Richfield incubations.†



7a. Control

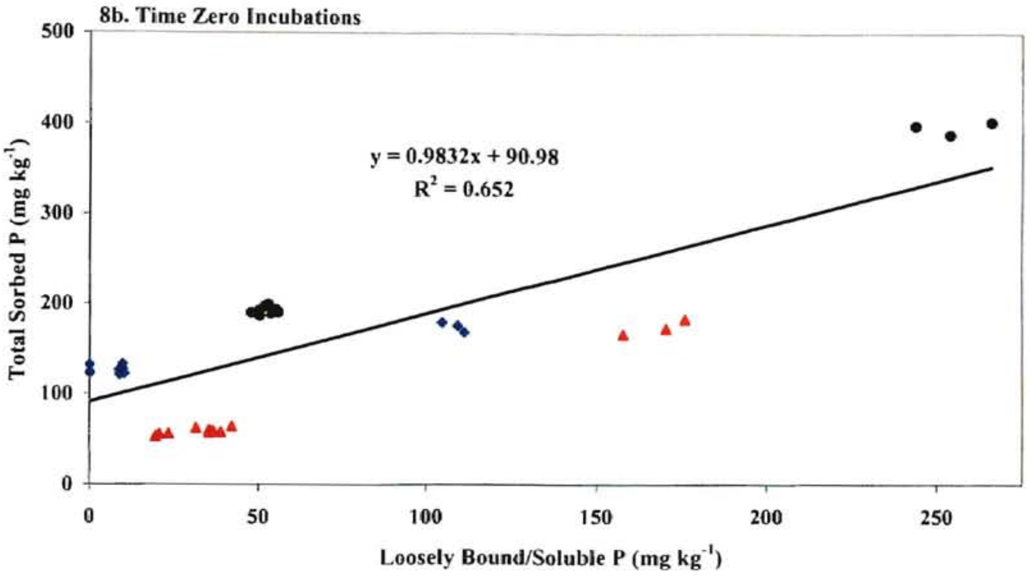
7b. 0-2 ft. Effluent

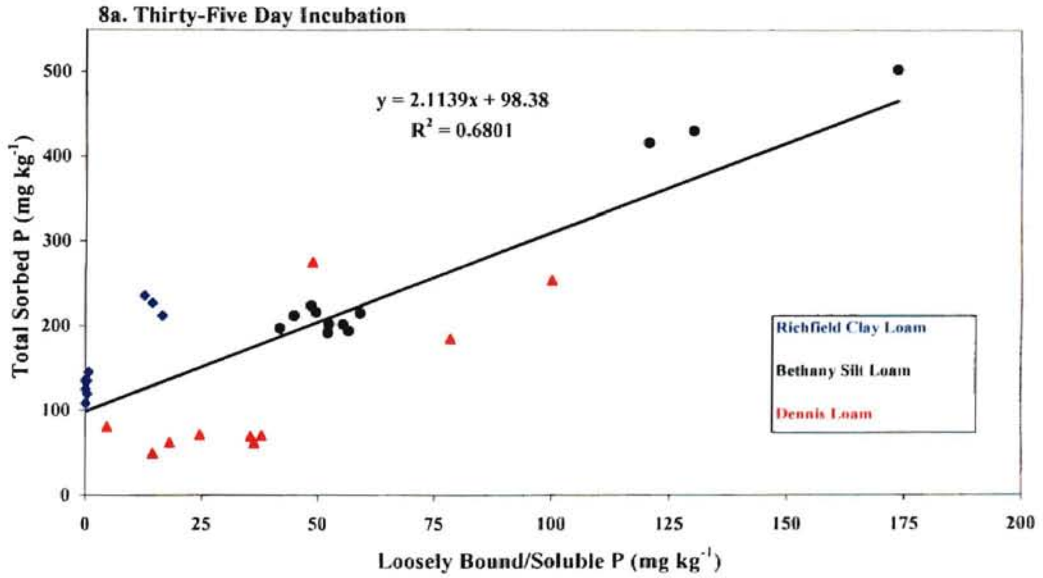
7c. 8-10 ft. Effluent

7d. Sludge

[†] Within each waste source, the P concentration of each fraction pool displaying the same letter are not significantly different ($P < 0.05$).

Figures 7a,b,c,d. P fraction concentration changes for unsaturated Richfield incubations.[†]





Figures 8a, b. Relationship between loosely bound/soluble P and P sorbed in Al, Fe, and Ca mineral fractions for 0 and 35 day unsaturated treatments.

CHAPTER III

PHOSPHORUS MOBILITY IN SOILS FROM LEACHING OF SWINE LAGOON WASTE

ABSTRACT

Leaching potential of swine waste phosphorus (P) in soils was investigated using column leaching tests and batch adsorption experiments. Eight Oklahoma soils possessing a range of chemical and physical properties were used in the experimentation. Batch adsorption tests were prepared using inorganic P concentrations ranging from 0-20 mg L^{-1}. From linear Langmuir adsorption isotherms, distribution coefficients and retardation factors (R) were calculated to predict P mobility. For the leaching column experiment, swine lagoon waste was leached through disturbed soil columns under a constant hydraulic gradient. A range of 6 to 7.5 pore volumes were leached over a 100 day period with no measurable P in the column effluent for the five finest textured soils. The columns were destructively sampled following the leaching period. In three coarse textured soils, P breakthrough was measured over a range of 6-12 pore volumes. To verify the results of the column experiment, the transport model WCOLUMN was utilized with the breakthrough/destructive column data to calculate an R factor for each soil. In the fine textured soils, the batch isotherms and the leaching columns yielded similar estimates for R. For the coarse textured soils, the batch isotherms estimated R factors that were significantly greater than those determined by the column test fit. Batch tests generated R factors of 7.5 to 129, while column tests yielded R values from 1.4 to

109. The sorption of P in the columns was strongly correlated with ammonium oxalate extractable aluminum and iron ($r^2 = 0.967$). These results are applicable in determining P transport through soils underlying swine lagoons and may aid in the selection of sites for lagoon installation.

INTRODUCTION

In the past decade, swine production in Oklahoma has increased dramatically. According to the National Agricultural Statistics Service of the United Stated Department of Agriculture, the swine industry within Oklahoma has grown from 215,000 total hogs in 1990 to 2,310,000 hogs in 2000 (National Agricultural Statistics Service, 2001). Most swine production systems use earthen lagoons for the storage and treatment of manure; in particular, large corporate farms utilize lagoons almost exclusively as a waste management option. Public concern has arisen about the effectiveness of earthen basins to store large volumes of liquid manure because of an increasing number of seepage incidents being reported. In June 2001, concern heightened when the Environmental Protection Agency directly linked groundwater contamination of a drinking water aquifer to local swine lagoons located in Kingfisher and Major Counties in central Oklahoma (Environmental Protection Agency, 2001).

Lagoon size can range from 0.2 hectare to several hectare in surface area depending on the number of hogs and amount of waste produced at the facility. As lagoon size increases, the risk of sidewall and/or liner failure rises due to mismanagement, poor construction, or from the burrowing of animals. Though most waste storage and treatment systems work properly, a number of lagoon failures have been reported where groundwater quality has been impacted. Ciravolo et al. (1979) discovered low levels of groundwater contamination due to the rupture of lagoon seals from the drying of exposed subsoil during a low containment period. Ritter et al. (1981) found elevated concentrations of chlorides and ammonia in groundwater due to localized seepage of an anaerobic swine lagoon. Westerman et al. (1995) investigated groundwater impact from two newly-formed clay lined lagoons underlied by a coarse textured material. Significant contamination to groundwater was measured 3.5 to 5 years after lagoon filling as a result of seeping liquid manure. Finally, Withers et al. (1998) found significant increases in dissolved PO_4^-, K^+, NH_4^+, Na^+, Cl^-, SO_4^-, and *Escherichia coli* in groundwater due to a dairy lagoon fissure.

Numerous investigators have concluded that most earthen lagoons effectively "self-seal" from the clogging of soil pores with organic material and/or from microbial activity (Barrington and Broughton, 1988). However, a prior study conducted by Perschke and Wright (1998) found that lagoon failures can occur along the sidewall of the basin within the top three feet of the surface. Lagoon liquid levels fluctuate significantly in this failure zone, which often creates cracking for preferential flow pathways. Additionally, fewer organics are located on the sidewall for the clogging of soil pores as demonstrated in Diagram 1. The diagram is provided below:

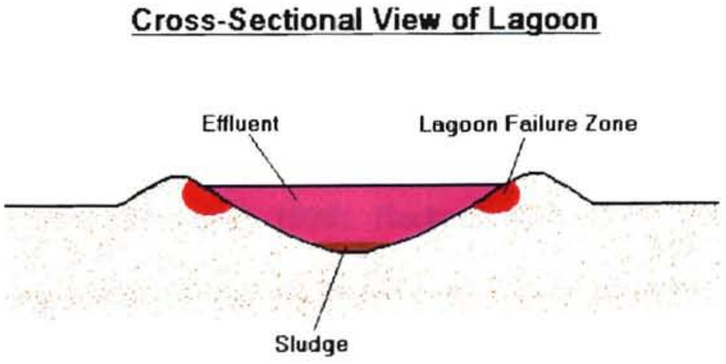


Diagram 1. Failure zone for earthen lagoons used for swine waste treatment. Designed by D. Wright based on data from Perschke and Wright (1998)

An elevated nitrate concentration in groundwater is a common indicator lagoon seepage has occurred. However, phosphates have also been identified to migrate with the effluent plume. Mallin et al. (1997) evaluated a poultry waste lagoon rupture and a swine lagoon leak in North Carolina during the summer of 1995. High concentrations of orthophosphate and total P (>11.0 mg L^{-1} measured for both) were discovered in stream waters several kilometers away from the lagoon sites. Excessive algal bloom, numerous fish and mollusk kills, and high coliform concentrations were observed in these accidents. Eight other accidents were recorded in North Carolina throughout the summer of 1995 where an approximated 4 million liters of waste was discharged from the facilities.

Besides lagoon failures, other circumstances exist where P mobility has been observed in subsurface environments. Historically, phosphate has been considered an immobile nutrient due to the tremendous number of mineral sinks for P within soil; however, soils possess a finite capacity to remove soluble P through chemisorption, nonspecific adsorption, and precipitation reactions. Documented cases of vertical P transport have been reported where large amounts of animal waste or fertilizer additions have resulted in P movement up to several meters below the surface in a range of soil textures (Chardon et al., 1997; Eghball et al., 1996; Lucero, 1995). Particulate-bound P and dissolved reactive phosphate have also been observed to leach in some soil systems (Smith et al., 1998; Beauchemin et al., 1998). Heckrath et al. (1995) proposed that P can leach in forms less susceptible to sorption (organic or colloidal phosphates) but is measured as dissolved reactive P following extraction and analysis procedures. Under reduced conditions, the reduction of iron phosphates through microbial respiration can yield soluble P (de Mello et al., 1998; Vadas and Sims, 1998). Finally, some work suggests that P leaching can occur before the total sorption capacity of a soil is reached. Holford et al. (1997) observed after a 12-year period of effluent application, P sorption capacity and sorption strength decreased by 40% to a 100 cm depth, while the KCl-extractable P (loosely bound/soluble P) levels increased to the same depth. Holford concluded that P leaching can occur well before the P sorption capacity of the soil is depleted.

To predict P movement in soil, several column leaching studies have been performed focusing on fertilizer, animal waste, and sewage sludge additions to surface soil (Chardon at al., 1997; Fox and Kamprath, 1971; Sawhney, 1977; Siddique et al., 2000). However, little research has been conducted which simulates the leaching of swine manure P from waste lagoons. In this investigation, the leaching potential of swine lagoon P will be evaluated for a wide range of soils using leaching columns and mobility factors (K_d and R) determined from the columns tests will be compared to linear equilibrium adsorption modeling. The mobility factors will also be compared to estimates calculated from batch adsorption experiments. The influence of specific soil physical and chemical properties in relation to vertical P migration will also be examined.

MATERIALS AND METHODS

Eight soils representing a wide range of chemical and physical properties were selected from different locations in Oklahoma where swine production is prevalent (Table II-1). The samples were obtained from the B-horizon of each soil profile. Subsurface soils were utilized in the experimentation because of the evidence demonstrating that most effluent leaching occurs on the sidewall of waste lagoons. Following sampling, the soils were air-dried and sieved using a 2 mm screen. A list of the soil series along with the classification and selected physical properties is shown in Table 1. Particle size distribution was determined by the bouyoucos hydrometermethod and saturated hydraulic conductivity (K_{sat}) was measured using the Darcy equation for a uniform cross-sectional area (Klute, 1986). Chosen chemical properties of the eight subsurface soils can be found in Table 2. Soil pH was measured by a 1:2 soil water solution using a glass electrode. Organic carbon (C) and total nitrogen (N) were analyzed via the dry combustion method. Amorphous iron and aluminum oxide were extracted using acid ammonium oxalate then analyzed by inductively coupled plasma-atomic emission spectroscopy (ICP-AES). Water soluble calcium was measured by shaking 1.0 g soil with 25 mL DI water for 30 minutes and analyzing the solution for calcium using ICP-AES. Finally, total P was measured using U.S. EPA Method 200.7, a nitric/hydrochloric acid digestion followed by analysis using an ICP-AES.

Phosphorus adsorption isotherms were used to estimate P sorption maximum (P_{max}) and predict P mobility in the soils. The linear Langmuir model was used to calculate P_{max}, distribution coefficients (K_d), and retardation factors (R). K_d and R are terms that predict the rate of P movement through soil and have been used in a number of past experiments evaluating solute movement (Faust and Mercer, 1980; Prickett et al., 1981; Srinivasan and Mercer, 1988). The K_d is equal to the slope of the linear Langmuir isotherm and is important in that K_d gives a relative indication of sorption potential. The R factor estimates the pore volumes required to attain half of the breakthrough curve and shown below as Eqn. 1:

$$R = 1 + (\rho_b/\theta) * K_d \qquad \text{Eqn. (1)}$$

where R: retardation factor

ρ_b: soil bulk density

θ: soil porosity

K_d: distribution coefficient.

Batch adsorption isotherms were performed using one gram of soil sample in 25 mL of varying concentration P solution (a K_2HPO_4 standard). Solution concentrations of 0.0, 0.5, 1.0, 1.5, 2.0, 2.5, and 5.0 mg L^{-1} P were shaken with the four coarse textured soils in a 50 mL centrifuge tube. P solution concentrations of 0.0, 1.0, 2.0, 5.0, 10.0, and 20.0 mg L^{-1} P were mixed with the fine textured samples. A wider range of P standards was used for the fine textured soils because of their ability to sorb greater concentrations of P in comparison to the coarse soils. After mixing the soil and P standard solution, the tubes were shaken for 24 hours on an end-to-end shaker at 150 oscillations per minute. The samples were then centrifuged for 10 minutes at 10,000 rpm and decanted for P analysis. The Murphy and Riley (1962) colorimetric method was utilized and the amount of P adsorbed was determined by the difference between the initial and the final amounts of P in solution. All soil adsorption isotherms were run in duplicate treatments.

The disturbed soil column experiments were conducted using acrylic plastic cylinders 100 cm in length and 4.34 cm in diameter. Column preparation involved placing filter paper (Whatman #2) at the bottom of each tube, then a 2 cm gravel layer (acid washed), and then another filter paper on top of the gravel. This step ensures uniform flow from the bottom of the sample column and inhibits the siltation of the effluent outlet. The columns were filled to a depth of 20 cm with soil and packed to selected bulk densities (coarse textured soils ρ_b= 1.50 g cm^{-3}; fine textured soils ρ_b= 1.33 g cm^{-3}). A 400 g weight was placed on top of the packed soil to prevent sidewall separation due to soil shrink-swell properties. The weights were chosen based on size (the greatest mass that could fit into the 4.34 cm column diameter) and their ability to distribute pressure evenly across the surface area of the soil column. The swine effluent delivery system was constructed using three plastic tanks connected to the columns using 1.25 cm Tygon tubing. The tanks held approximately 28 L of lagoon effluent each and were placed above the columns to create a hydraulic head of 1.5 m. The tanks were filled regularly with effluent to retain a constant hydraulic gradient throughout the experiment. Columns were randomly placed in racks and covered to prevent algae or other photosynthetic biological growth.

The leachate from each soil column was captured in polyethylene Nalgene bottles (acid washed before used) marked with the appropriate pore volume. Every 0.25 pore volume of leachate was collected for the coarse textured soils and every 0.5 pore volume for the fine textured soils. The fine textured soils were run in triplicate due to the anticipated destructive sampling required because of no P breakthrough, while no replications were created for the coarse textured soils in which P breakthrough was expected. After the desired pore volume was collected, the leachate was immediately analyzed for dissolved reactive P by the Murphy and Riley (1962) colorimetric method. After the leaching period, destructive sampling occurred by sectioning the columns at five depths (0-1 cm, 1-5 cm, 5-10 cm, 10-19 cm, and 19-20 cm column depth). Soils from each depth were air-dried and analyzed for total P by EPA Method 200.7. To determine P movement through the soil column, the initial total P concentration was subtracted from the P extracted from the analysis of the sectioned columns (referred to as effluent residual P). Additionally, all soil columns were analyzed for total N, total C, and pH were measured for the sectioned column soils. Photographs displaying the leaching column apparatus and design before and after the swine lagoon effluent experiment began are provided in the appendix as Figure II-1.

The leaching duration for the coarse and fine textured soils ranged from 22 hours to 100 days (Table 3). For each soil series, leaching ceased when P breakthrough was observed or when the 100 day period was completed. Hydraulic conductivity decreased over the course of the leaching experiment possibly due to organics in the swine effluent clogging pores in the upper region of the soil columns. The blockage of soil pores by bacterial cells and fine organic matter in effluent is well documented in numerous field studies (DeTar, 1979; Rowsell et al., 1985, Barrington et al., 1987).

Swine effluent used in the leaching column experiments was obtained immediately before the column leaching began from a facultative anaerobic lagoon located in Stillwater, Oklahoma. Some of the pertinent characteristics of the swine waste are as follows: pH= 7.94; percent solids = 0.1%; total N= 0.019%; total C= 0.060%; dissolved reactive P= 40.1 mg L^{-1}; total P= 70.3 mg L^{-1}.

Following P analysis on captured leachate and sectioned soil columns, data was input into the transport model WCOLUMN to compare to the results obtained from the column experiment. WCOLUMN is a Windows-based computer program for fitting model parameters to column flow breakthrough curves. The program is a one-dimensional linear equilibrium adsorption model for pulse injection with first order decay. WCOLUMN has been validated through work performed by Parker and van Genuchten (1984) and Budge (1996). For each soil, specific model parameters were input into WCOLUMN to predict P transport by calculated R factors. Soil porosity, Darcy flux, effluent P concentration, P dispersion, and duration of leaching period were parameters required for the model fitting of P breakthrough. A value of one was used for the P dispersion factor due to the insensitivity of the model for this parameter and the absence of swine effluent P dispersion data. No additional assumptions or estimates were used for any other model input data.

WCOLUMN offers two boundary condition options for model fitting, flux out of the column or resident movement within the column. The ability of the model to estimate in-column and out-of-column P movement is important due to the variability of P sorption capacities in soils. Resident solute movement is applicable for fine textured soils with high sorption capacities and the flux boundary condition is applied to course textured soils with lower sorption capacities. In summary, WCOLUMN can validate P mobility observed in columns with P breakthrough and can estimate a model curve for columns in which no breakthrough is observed.

RESULTS AND DISCUSSION

Langmuir Adsorption Isotherms

Linear Langmuir P adsorption isotherms for the eight B-horizon soils are shown in Figure 1. The soils displayed a wide distribution of P sorption potentials, which can be observed from the ranges of P_{max} on Table 2 where the batch equilibrium experiment indicates the Norge clay loam has the greatest ability to sorb P with a P_{max} of 147 mg kg^{-1}. The Eufaula sand, Pratt sand, Dougherty loamy sand, and the Yahola sandy loam exhibited characteristics of most coarse textured soils by sorbing significantly lower P concentrations than the finer soils. The P_{max} for the coarse soils ranged between 11-72 mg kg^{-1}, in comparison to a 65-147 mg kg^{-1} P_{max} range for the finer textured soils. These results are similar to P isotherm results found by Furman (2000) for twenty-eight benchmark soils in Oklahoma.

Soil Leaching Column Study

For the leaching column experiment, dissolved reactive P breakthrough was achieved in three of the eight soils. The Eufaula sand, Pratt sand, and Dougherty loamy sand demonstrated P breakthrough ranging from 6-12 pore volumes over the course of a 34 hour, 22 hour, and 21 day leaching period, respectively. Additionally, these three soils contained the highest sand content of the eight soils used in the study. The dissolved reactive P breakthrough data along with the corresponding WCOLUMN model fit curves are displayed for these coarse textured soils in Figure 2a-c. Notably from the three graphs, the leachate P approached the 40.1 mg L^{-1} mean dissolved P concentration in the swine effluent, but the influent concentration was not attained in the column effluent. This observation is typical of most P leaching studies, even after several hundred pore volumes have been passed through the column (Sawhney, 1977; Siddique et al., 2000). WCOLUMN model outputs with corresponding chi-squared values are provided in Table II-2, Table II-3, and Table II-4.

Dissolved P breakthrough for the four fine textured soils and the sandy loam soil (Yahola) did not occur over the 100 day leaching duration; therefore, the columns were destructively sampled to determine P movement with depth. No dissolved reactive P was measured in any of the leachate collected from the Port, Norge, Dennis, or Richfield soil columns. Swine effluent P was measured in the leachate of the Yahola column, but complete P breakthrough did not occur within the 100 days (a final dissolved reactive P concentration of 22.1 mg L^{-1} was measured in the 7.5 pore volume). For the destructively sampled columns, a range of 6-7.5 pore volumes was captured during the 100 day period. The WCOLUMN model outputs for the five destructively sampled columns are provided in Table II-5 through Table II-9.

Figure 3a-e displays the effluent residual P concentration measured for the Port loam, Norge clay loam, Dennis clay loam, Richfield clay, and Yahola sandy loam at the 0-1 cm, 1-5 cm, 5-10 cm, 10-19 cm, and 19-20 cm column depth. The graphs also exhibit the model fit curves calculated from the analytical results. All of the fine textured soils demonstrated the same general trend of a sharp decrease in P below the top section(s) of the column. These results are similar to findings by King et al. (1990) in which swine lagoon effluent was applied to bermudagrass over an 11 year period. In the field study, swine lagoon P accumulated near the soil surface and decreased rapidly to a 30-40 cm depth.

When comparing P movement among the fine textured soils, more effluent residual P was measured at a greater depth for the calcareous Richfield clay (Figure 3a-e). From the 5-10 cm column depth, the Richfield contained approximately four times more effluent residual P (4.15 mg kg^{-1}) than any other fine textured column at that depth and more than twice the concentration of any other column at the 10-19 cm depth. In spite of the increased P mobility, the Richfield possessed a greater clay percentage than the Dennis, Norge, or Port (a higher clay content of 7.5%, 7.5%, and 22.5%, respectively), a high concentration of amorphous iron and aluminum oxide (a sum of 1,742.2 mg kg^{-1}), and a relatively high P sorption potential with a P_{max} of 125 mg kg^{-1}. A possible explanation of the increased P mobility in the Richfield clay could be due to a greater P saturation. With 424 mg kg^{-1} total P, the initial total P of the Richfield was more than twice the concentration than any other soils used in the experiment. Furthermore, the Norge and the Richfield soils possessed similar amorphous iron and aluminum oxide concentrations (1,853 and 1,742 mg kg^{-1}, respectively), yet the P mobility in the Norge column was far less in comparison to the Richfield. Similar past research displaying P saturation in near-surface soil causing increased P mobility into lower portions of the profile has been performed by Holford et al. (1997), Eghball et al. (1996), and Lucero et al. (1995). Furthermore, the potential for P loss from Dutch soils is classified by the degree of saturation of P sorption (Sharpley et al., 1994).

Swine lagoon P sorption in the fine and coarse textured soil columns appeared to be strongly influenced by amorphous iron and aluminum oxide content. Table 2 displays the wide range of ammonium oxalate extractable iron and aluminum for the eight soils with concentrations from 102.3-1,005.8 mg kg^{-1} for amorphous iron oxide and 60.4-851.2

mg kg^{-1} amorphous aluminum oxide. The total P measured in the 0-1 cm column section in relationship to the summation of amorphous iron and aluminum oxide concentration for each soil is presented in Figure 4. A strong positive linear correlation (r^2= 0.9673) exists between P sorbed and amorphous iron and aluminum oxide content, which reflects that specific adsorption is a key mechanism in rendering P immobile (McLean and Bledsoe, 1992). Percent clay content and P sorption maximum also displayed positive correlations with coefficients of r^2= 0.775 and r^2= 0.680, respectively, provided on Figure II-2 and II-3. Poor correlations were observed between total P extracted from the 0-1 cm segment and water soluble calcium or percent organic carbon with r^2= 0.132 and r^2= 0.3775, respectively.

Soil pH values displayed a wide degree of variability with depth in the sectioned leaching columns (Figure 5a-b). In the majority of the columns, pH decreased in the 0-1 cm and 1-5 cm sections following the leaching experiment (statistically significant at the $\alpha = 0.05$ probability level), with an overall mean decrease of 0.5 pH units. The Pratt sand and Eufaula sand pH values declined by 1.1 and 1.0 units, respectively, in comparison to initial pH. Many of the soils equilibrated closer to their initial pH values toward the bottom of the columns. Biological activity and the decomposition of organics trapped in the top portion of the column is a potential explanation for the increased acidity. This decrease in soil pH could aid in the sorption of higher effluent P concentrations from the precipitation of iron and aluminum phosphates. Another general trend observed was the shifting of the fine textured soil pH values in the 0-1 cm section toward the lagoon waste's pH of 7.9. Increased residence time (the 100 day leaching period) between the slightly alkaline swine effluent and the Norge, Port, Richfield, and Dennis soils is the likely cause of this pH change noted in the top segment of the fine textured columns.

Leaching Column Results vs. Batch Adsorption Isotherm Mobility Estimates

Phosphorus transport in the leaching column study was compared to the batch test mobility estimates. This comparison between the two experiments was performed to evaluate if the adsorption isotherm estimates of P mobility correlate to actual P migration in leaching columns. The mobility factors K_d and R obtained from the column experiment and the batch adsorption isotherms are displayed in Table 4 along with a statistical comparison of the mobility factors obtained from the fine textured soils. Since no replications were created for the coarse textured leaching columns, statistical analysis could not be performed to compare the mobility factors calculated from the WCOLUMN model results and the adsorption isotherm mobility estimates. However, a simple comparison of the mean difference in retardation factors clearly displayed obvious distinctions between the results of the two experiments.

As seen in Table 4, the batch isotherms for the fine textured soils yielded similar K_d and R factors when compared to the actual P mobility in the leaching columns for three of the four soil series. No statistical difference was found in the mobility factors for the Dennis clay loam, Norge clay loam, or Port loam ($\alpha = 0.05$). A mean difference in R ranged from 9.95%, 15.2%, 8.92%, and 25.6 % for the Dennis, Norge, Port, and Richfield soils, respectively. The coarse textured soils exhibited a significantly greater difference in R factors with mean differences of 72.3%, 81.3%, 78.9%, and 39.4% for the Dougherty, Eufaula, Pratt, and Yahola soils, respectively.

For the sandy soils, the variation between the R values of the batch test and column study could be due to several potential reasons. Historical research has shown adsorption isotherms typically sorb greater concentrations of P than that observed in leaching column studies. For example, Jardine et al. (1988) found that solute retardation for several cationic and anionic tracers was overestimated for batch isotherms in relation to soil columns. Jardine et al. (1988) concluded that the use of batch adsorption measurements for transport parameters might result in erroneous estimates. Explanations for this occurrence are beyond the scope of this work.

Another explanation for the wide variability in R factors observed in the sandy soils could be due to differences in the sorption of organic phosphates versus inorganic phosphates. The leaching column study utilized swine lagoon effluent as a P source containing organic P (approximately 40% of the P was organic, based on the dissolved reactive P concentration of 40.1 mg L^{-1} and a total P analysis of 70.3 mg L^{-1}), while the isotherms used an inorganic P source (K_2HPO_4). Numerous prior studies have demonstrated that organic P, bound up in colloidal phosphates, is more mobile than inorganic P (Beauchemin et al., 1998; Chardon et al., 1997; Eghball et al., 1996; James et al., 1996; Lucero et al., 1995). Due to this higher mobility of organic P, the isotherms could have underestimated the P transport factors in comparison to the leaching columns model output. Additionally, the non-Gausian shape of the Pratt and Eufaula breakthrough curves also supports the hypothesis that colloidal P could have been transported more rapidly, particularly in the larger pores of the coarse textured soils. Both liquids had similar pH values (swine effluent: 7.9; K_2HPO_4 solutions: 7.7), so the possibility of the solutions altering the sorption and precipitation potential of the soils was ruled out.

Of the fine textured soils, the Norge and Richfield soils demonstrated the greatest differences in R values estimated from the isotherms and leaching column results. Although the predicted P mobility factors appeared to be more accurate for the fine soils in relation to the sandier soils, mean differences in R of 15.2% and 25.6% were calculated for the Norge and Richfield soils, respectively (Table 4). Additionally, these two soils possessed the highest concentrations of amorphous iron and aluminum oxides of the eight soils used in the experiments. The deviation in R factors could be due to the release of phosphate from iron/aluminum oxides under reducing conditions present in the soil columns (redox values of –146 mV and –118 mV were measured for the Norge and Richfield soils, respectively, following the 100 day leaching period). The 24 hour shake time for the adsorption isotherms would not be expected to create conditions this reduced and therefore could not account for the P mobility attributed to desorbed phosphate from the sesquioxides. Phillips (2001), in a similar soil column study using an inorganic P addition to a reduced soil rich in iron oxide clay, also concluded that increased water soluble P was attributed to solubilized phosphates under reducing conditions. Finally, mineralization of P within the swine effluent could also have influenced the downward migration of phosphates within the columns.

In summary, the implementation of adsorption isotherm analysis for lagoon liners and/or underlying lagoon soils would give a good indication of P leaching potential, assuming the material is a fine textured soil. Column leaching studies would provide the most accurate measure of P mobility, but this procedure is time consuming, labor intensive, and economically impractical. As seen in the column tests and the batch isotherm data, coarse textured soils are not suitable for lagoon placement due to their inability to retard vertical P movement and the limited sorption potential of sands in comparison to silts and clays. State regulatory agencies responsible for lagoon construction should consider adopting stricter parameters on the material used for lagoon bottoms, especially in P sensitive watersheds. Specifically, soil properties such as percent sand, percent clay, amorphous iron oxide content, amorphous aluminum oxide content, and P sorption maximum would supply pertinent information on soil factors important on the immobility of P. Additional studies need to be performed which focus on the minimum concentrations of amorphous iron and aluminum oxides necessary to sorb swine effluent P without threatening groundwater quality.

CONCLUSIONS

Phosphorus leaching from swine lagoon effluent can be affected by several soil chemical and physical properties. Characteristics such as soil texture, clay content, P sorption maximum, and amorphous iron and aluminum oxide content had the greatest influence on restricting vertical P movement. Additionally, linear equilibrium adsorption modeling appears to adequately estimate swine lagoon P breakthrough and transport in a broad range of soils.

The results from this project are applicable for determining P transport through soils underlying swine lagoons and could aid in the selection of sites for lagoon installation. As seen from the leaching column data, phosphate movement in a lagoon failure scenario can result in significant P impact to groundwater quality if the subsurface lithology consists of coarse textured soils. By contrast, fine textured soils have the ability to retard infiltration and sorb much greater quantities of P. Additionally, soils possessing high concentrations of amorphous iron and aluminum oxides are best suited for lagoon placement in environmentally sensitive areas susceptible to P enrichment. As stated previously, an adsorption isotherm test or leaching column study is highly recommended as a screening tool for future lagoon placement sites.

The selection of adequate locations for earthen basins would significantly curtail the risk associated with groundwater contamination. Currently in the United States, most state regulatory agencies have adopted guidelines or standards for lagoon construction specifications. Investigating potential lagoon placement sites for subsurface coarse textured material (sand lenses, alluvial sands or gravels, or buried horizons) would screen out locations where a lagoon leak could impact surface or groundwater quality. Finally, lagoon construct should be strictly regulated in areas of high water tables or in regions where seepage could discharge into surface water bodies.

REFERENCES

Barrington, S.F., P.J. Gutras, and R.S. Broughton. 1987. The sealing of soil by manure. I. Preliminary investigations. Can. Agric. Eng. 29:99-103.

Barrington, S.F. and R.S. Broughton. 1988. Designing earthen storage facilities for manure. Can. Agric. Eng. 30:289-292.

Beauchemin, S., R.R. Simard, and D. Cluis. 1998. Forms and concentration of phosphorus in drainage water of twenty-seven tile-drained soils. J. Environ. Qual. 27:721-728.

Budge, K.G. 1996. Results of fitting column breakthrough curves. Memorandum to D.A. Lucero, Feb. 2, 1996. Sandia National Laboratories, Albuquerque, NM.

Chardon, W.J., O. Oenema, P. del Vriesema, J. Japenga, and D. Blaauw. 1997. Organic phosphorus in solutions and leachates from soils treated with animal slurries. J. Environ. Qual. 26:372-378.

Ciravolo, T.G., D.C. Martens, D.L. Hallock, E.R. Collins, Jr., E.T. Kornegay, and H.R. Thomas. 1979. Pollutant movement to shallow ground water tables from anaerobic swine waste lagoons. J. Environ. Qual. 8:126-130.

de Mello, J.W.V., V. Barron, and J. Torrent. 1998. Phosphorus and iron mobilization in flooded soils from Brazil. Soil Sci. 163:122-132.

DeTar, W.R. 1979. Infiltration of liquid dairy manure into soil. Trans. ASAE 22:520-528.

Eghball, B., G.D. Binford, and D.D. Baltensperger. 1996. Phosphorus movement and adsorption in a soil receiving long-term manure and fertilizer application. J. Environ. Qual. 25:1339-1343.

Environmental Protection Agency. 2001. EPA orders three companies to provide safe drinking water to homes [Online]. Available at http://www.epa.gov/Region06/6xa/seaboard.pdf (verified in Oct. 2001).

Faust, C.R., and J.W. Mercer. 1980. Ground-water modeling: recent developments. Ground Water. 6:569-577.

Fox, R.L., and E.J. Kamprath. 1971. Adsorption and leaching of P in acid organic soils and high organic matter sand. Soil Sci. Soc. Amer. Proc. 35:154-156.

Furman, J.K. 2000. Phosphorus sorption and desorption characteristics of selected Oklahoma soils. M.S. thesis. Oklahoma State Univ.

Heckrath, G., P.C. Brookes, P.R. Poulton, and K.W.T. Goulding. 1995. Phosphorus leaching from soils containing different phosphorus concentrations in the broadbulk experiment. J. Environ. Qual. 24:904-910.

Holford, I.C.R., C. Hird, and R. Lawrie. 1997. Effects of animal effluents on the phosphorus sorption characteristics of soils. Aust. J. Soil Res. 35:365-373.

James, D.W., J. Kotuby-Amacher, G.L. Anderson, and D.A. Huber. 1996. Phosphorus mobility in calcareous soils under heavy manuring. J. Environ. Qual. 25:770-775.

Jardine, P.M., G.V. Wilson, R.J. Luxmoore. 1988. Modeling the transport of inorganic ions through undisturbed soil columns from two contrasting watersheds. Soil Sci. Soc. Am. J. 52:1252-1259.

King, L.D., J.C. Burns, and P.W. Westerman. 1990. Long-term swine lagoon effluent applications on 'Coastal' bermudagrass: II. Effect on nutrient accumulation in soil. J. Environ. Qual. 19:756-760.

Klute, A. 1986. Methods of soil analysis: Part 1-physical and mineralogical methods. ASA and SSSA, Madison, WI.

Lucero, D.W., D.C. Martens, J.R. McKenna, and D.E. Starner. 1995. Accumulation and Movement of phosphorus from poultry litter application on a Starr clay loam. Commun. Soil Sci. Plant Anal. 26:1709-1718.

Mallin, M.A., J.M. Burkholder, M.R. McIver, G.C. Shank, H.B. Glascow, Jr., B.W. Touchette, and J. Springer. 1997. Comparative effects of poultry and swine waste lagoon spills on the quality of receiving streamwaters. J. Environ. Qual. 26:1622-1631.

McLean, J.E., and B.E. Bledsoe. 1992. Behavior of metals in soils. USEPA Rep. 540/S-92/018. USEPA Center for Environmental Research Information, Cincinnati, OH.

Murphy, J., and J.P. Riley. 1962. A modified single solution method for the determination of phosphate in natural waters. Anal. Chim. Acta. 27:31-36.

National Agricultural Statistics Service. 2001. Published estimates data base for hogs and pigs in Oklahoma [Online]. Available at http://www.nass.usda.gov:81/ipedb/ (verified in Oct. 2001).

Parker, J.C., and van Genuchten. 1984. Determining transport parameters from laboratory and field tracer experiments. 84:3. M.S. thesis, Virginia Agricultural Experiment Station.

Perschke, S.P., and P.E. Wright. 1998. Evaluation of soil under an earthen manure storage pond. ASAE Int. Meeting. Orlando, FL. July 12-16, 1998. Paper No. 982145. ASAE, St. Joseph, MI.

Phillips, I.R. 2001. Nitrogen and phosphorus transport in soil using simulated waterlogged conditions. Commun. Soil Sci. Plant Anal. 32:821-842.

Prickett, T.A., T.C. Naymik, and C.G. Lonnquist. 1981. A "random walk" solute transport model for selected groundwater quality evaluations. Illinois State Water Survey, Bull. 65. p. 103.

Sawhney, B.L. 1977. Predicting phosphate movement through soil columns. J. Environ. Qual. 6:86-89.

Sharpley, A.N., S.C. Chapra, R. Wedepohi, J.T. Sims, T.C. Daniel, and K.R. Reddy. 1994. Managing agricultural phosphorus for protection of surface waters: Issues and options. J. Environ. Qual. 23:437-451.

Siddique, M.T., J.S. Robinson, and B.J. Alloway. 2000. Phosphorus reactions and leaching potential in soils amended with sewage sludge. J. Environ. Qual. 29:1931-1938.

Smith, K.A., A.G. Chalmers, B.J. Chambers, and P. Christie. 1998. Organic manure phosphorus accumulation, mobility and management. Soil Use and Management. 14:154-159.

Ritter, W.F., E.W. Walpole, and R.P. Eastburn. 1981. An anaerobic lagoon for swine manure and its effect on groundwater quality in sandy-loam soils. p. 244-246. *In* Livestock waste, a renewable resource. ASAE, St. Joseph, MI.

Roswell, J.G., M.H. Miller, and P.H. Groenevelt. 1985. Self sealing of earthen liquid manure storage ponds. II. Rates and mechanisms of sealing. J. Environ. Qual. 14:539-543.

Srinivasan, P., and J.W. Mercer. 1988. Simulation of biodegradation and sorption processes in ground water. Ground Water. 4:475-487.

Vadas, P.A., and J.T. Sims. 1998. Redox status, poultry litter, and phosphorus solubility in Atlantic Coastal Plain Soils. Soil Sci. Soc. Am. J. 62:1025-1034.

Westerman, P.W., R.L. Huffman, and J.S. Feng. 1995. Swine-lagoon seepage in sandy soil. Trans. ASAE 38:1749-1760.

Withers, P.J.A., H.G. McDonald, K.A. Smith, and C.G. Chumbley. 1998. Behaviour and impact of cow slurry beneath a storage lagoon: 1. Groundwater contamination 1975-1982. Water Air Soil Pollut. 107:35-49.

Table 1. Classification, percent separates, and texture for eight B-horizon soils used in column study and batch adsorption isotherm tests.

Soil Series	Classification	% Sand	% Silt	% Clay	Textural Class
Dennis	fine, mixed, thermic Aquic Argiudoll	37.5	27.5	35.0	clay loam
Dougherty	loamy, mixed, thermic Arenic Haplustalf	87.5	7.5	5.0	loamy sand
Eufaula	sandy, siliceous, thermic Psammentic Paleustalf	95.0	5.0	0.0	sand
Norge	fine-silty, mixed, thermic Udic Paleustoll	27.5	37.5	35.0	clay loam
Port	fine-silty, mixed, thermic Cumulic Haplustoll	32.5	47.5	20.0	loam
Pratt	sandy, mixed, thermic Psammentic Haplustalf	90.0	5.0	5.0	sand
Richfield	fine, smectitic, mesic Aridic Argiustoll	20.0	37.5	42.5	clay
Yahola	coarse-loamy, mixed, thermic, Typic Ustifluvent	70.0	22.5	7.5	sandy loam

Table 2. Selected chemical properties of eight B-horizon soils used in column study and batch adsorption isotherm tests.

Soil Series	pH	Redox Potential* (mV)	Organic Carbon (g kg-1)	Amorphous Iron Oxide (mg kg-1)	Amorphous Aluminum Oxide (mg kg-1)	Water Soluble Calcium (mg L-1)	Total N (mg kg-1)	Total P (mg kg-1)	Pmax# (mg kg-1)
Dennis	7.69	-215	0.14	366.0	363.7	5.8	55.2	40	141
Dougherty	5.50	-60	0.40	317.6	170.1	3.5	94.3	138	72
Eufaula	7.03	70	0.02	102.3	60.4	3.3	2.1	44	11
Norge	6.01	-146	0.88	1001.8	851.2	4.0	200.7	153	147
Port	7.80	-182	1.20	658.7	243.7	8.5	257.5	173	65
Pratt	7.34	77	0.48	119.8	203.9	8.1	110.2	173	29
Richfield	8.29	-118	0.48	1005.8	736.4	32.7	107.6	424	125
Yahola	8.28	-100	0.19	230.5	85.2	11.2	48.0	165	21

*Redox potential was measured immediately after leaching ceased

#Pmax was determined by the inverse slope of the Langmuir adsorption isotherm

Table 3. Bulk density (D_B) of packed soil columns, leaching duration, and saturated hydraulic conductivity (K_{sat}) at the starting and ending of leaching period.

Soil Series	D_B (g cm^{-3})	Leaching Duration	Starting K_{sat} (cm s^{-1})	Ending K_{sat} (cm s^{-1})
Dennis	1.33	100 days	2.23E-07	1.27E-07
Dougherty	1.50	21 days	9.25E-06	1.09E-06
Eufaula	1.50	34 hours	5.55E-04	3.52E-04
Norge	1.33	100 days	2.60E-07	1.59E-04
Port	1.33	100 days	6.24E-07	2.64E-07
Pratt	1.50	22 hours	7.88E-04	6.95E-04
Richfield	1.33	100 days	2.11E-07	1.06E-07
Yahola	1.50	40 days	4.37E-06	2.79E-06

Table 4. Calculated distribution coefficients (K_d), retardation coefficients (R), and mean difference in R between the batch adsorption isotherm and leaching column experiment for each soil.

Soil Series	Experimental Method	K_d [†]	R [‡]	Mean Difference in R [††] (%)
Fine Textured Soils*				
Dennis	batch adsorption isotherms	25.0a	68.8a	9.95
	leaching column/model fit	27.8a	76.4a	
Norge	batch adsorption isotherms	47.0a	128.6a	15.2
	leaching column/model fit	40.0a	109.0a	
Port	batch adsorption isotherms	10.5a	29.6a	8.92
	leaching column/model fit	11.6a	32.5a	
Richfield	batch adsorption isotherms	17.1a	47.7a	25.6
	leaching column/model fit	12.7b	35.5b	
Coarse Textured Soils**				
Dougherty	batch adsorption isotherms	5.0	16.6	72.3
	leaching column/model fit	1.0	4.6	
Eufaula	batch adsorption isotherms	2.1	7.5	81.3
	leaching column/model fit	0.1	1.4	
Pratt	batch adsorption isotherms	5.5	18.0	78.9
	leaching column/model fit	0.8	3.8	
Yahola	batch adsorption isotherms	4.6	17.0	39.4
	leaching column/model fit	2.7	10.3	

[†] K_d was determined in the batch test method by calculating the slope of the linear portion of each soil's adsorption isotherm; in the leaching column experiment K_d was determined using WCOLUMN.

[‡] R was estimated in the batch tests by the following equation: $R= 1+(D_B/\Theta)*K_d$; in the leaching column experiment R was determined using WCOLUMN.

[††] Mean difference in R is determined by dividing the least of the two R factors calculated from the batch test or model fit into the opposite R, then multiplying by 100 (yielding a percent). The product is subtracted from 100.

*Within each fine textured soil series, mobility factors K_d and R are not significantly different ($P < 0.05$) when comparing batch isotherms to leaching column/model fit data.

**Statistical comparison for the mobility factors of the coarse textured soils could not be determined due to no replications of leaching columns tests; therefore, mean difference in R was calculated.

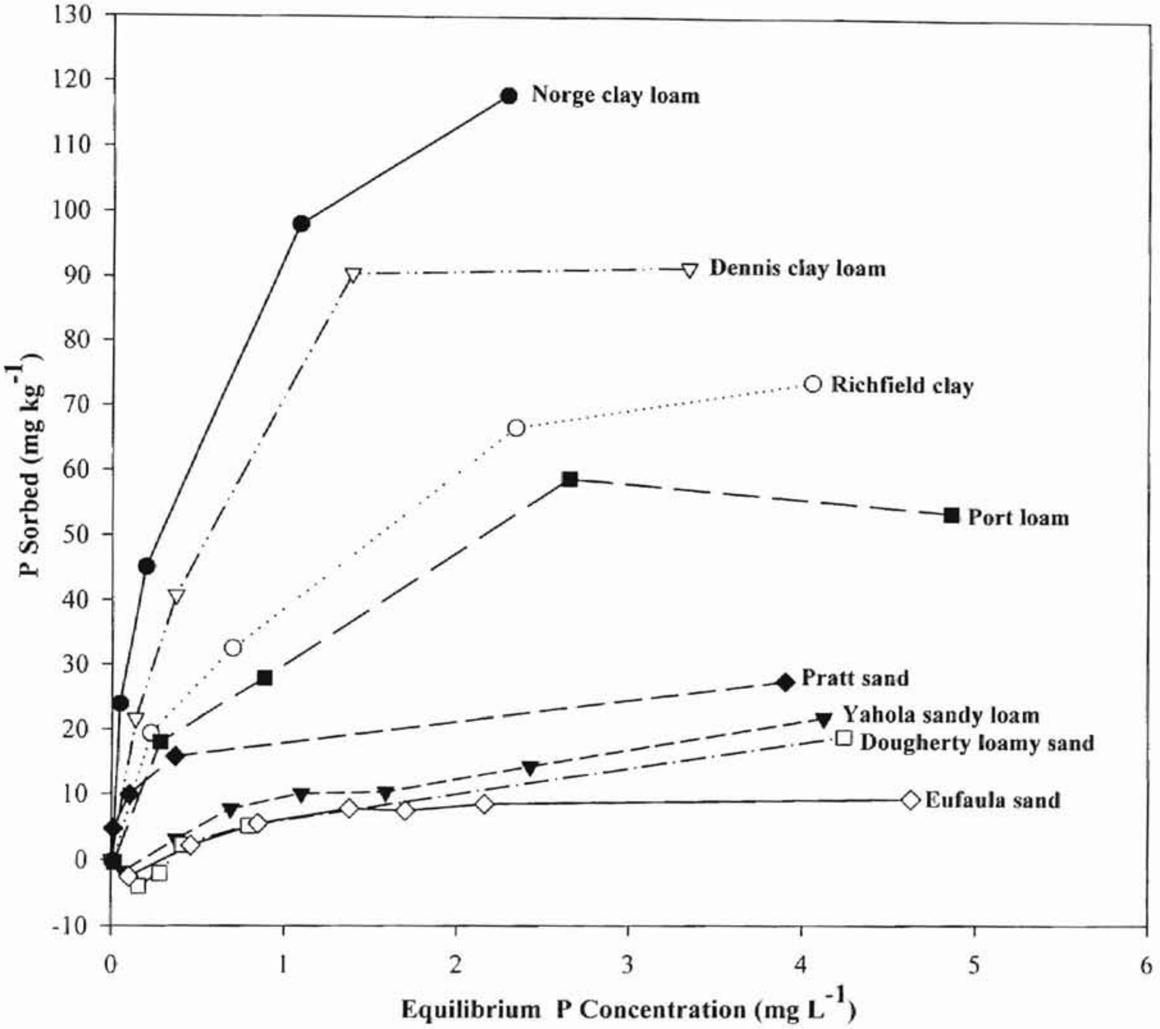


Figure 1. Phosphorus adsorption isotherms for soils used in batch tests and leaching column study.

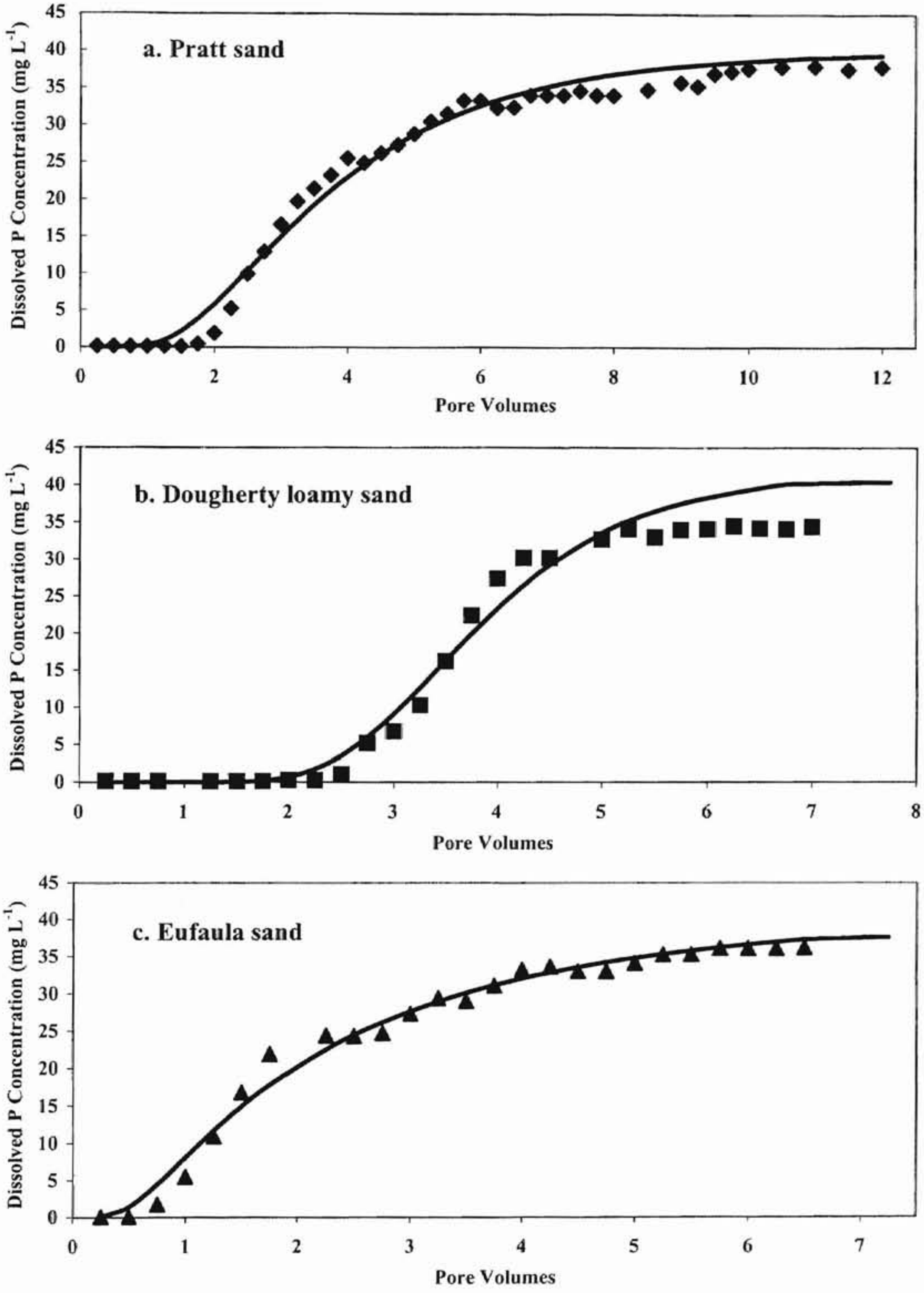


Figure 2a-c. WCOLUMN model fit curves (lines) and dissolved reactive P breakthrough data (points) for the three coarse textured soils from column leaching tests.

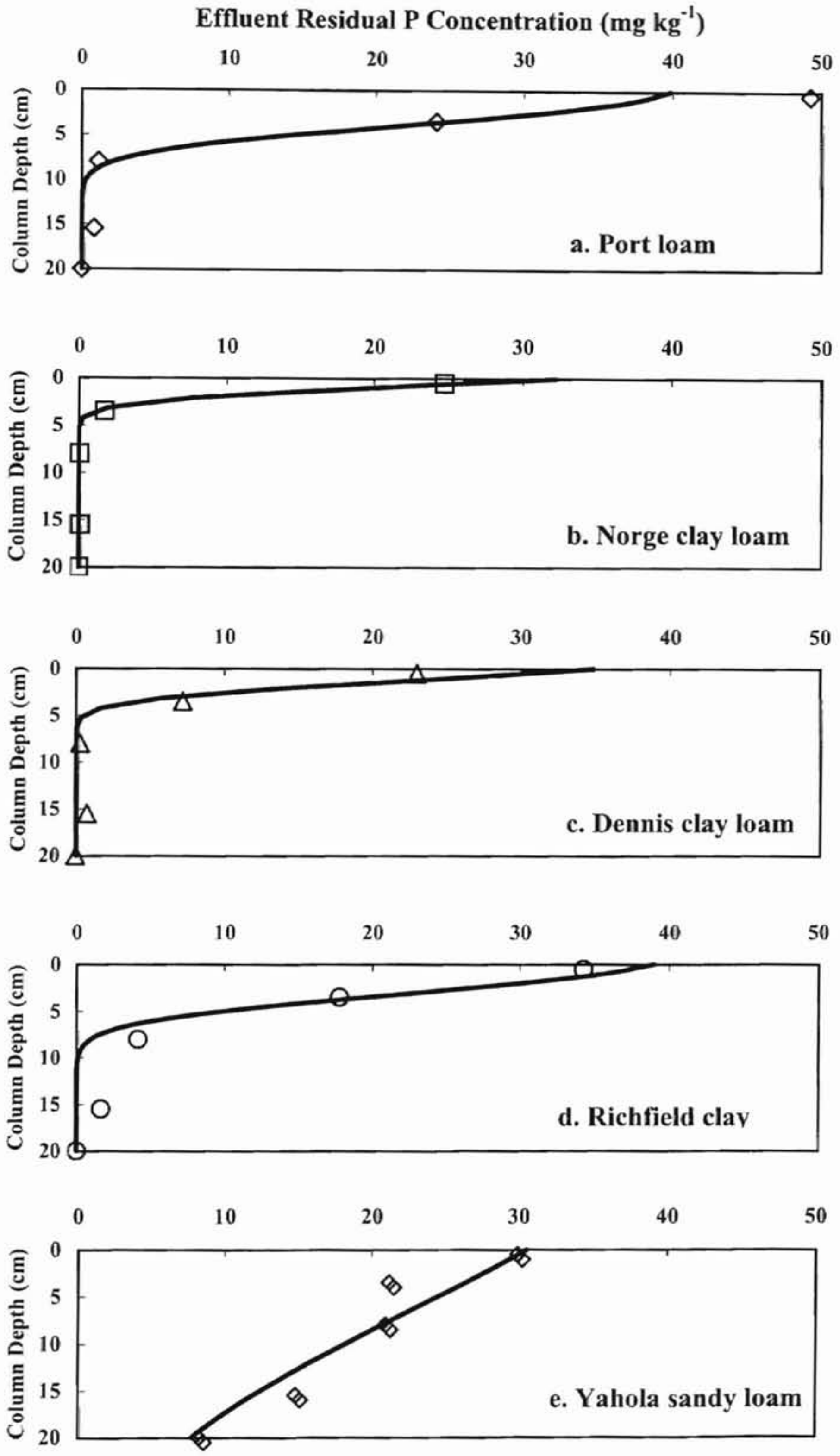


Figure 3a-e. WCOLUMN model fit curves (lines) and total P concentrations with depth from column leaching tests for destructively sampled soils (points).

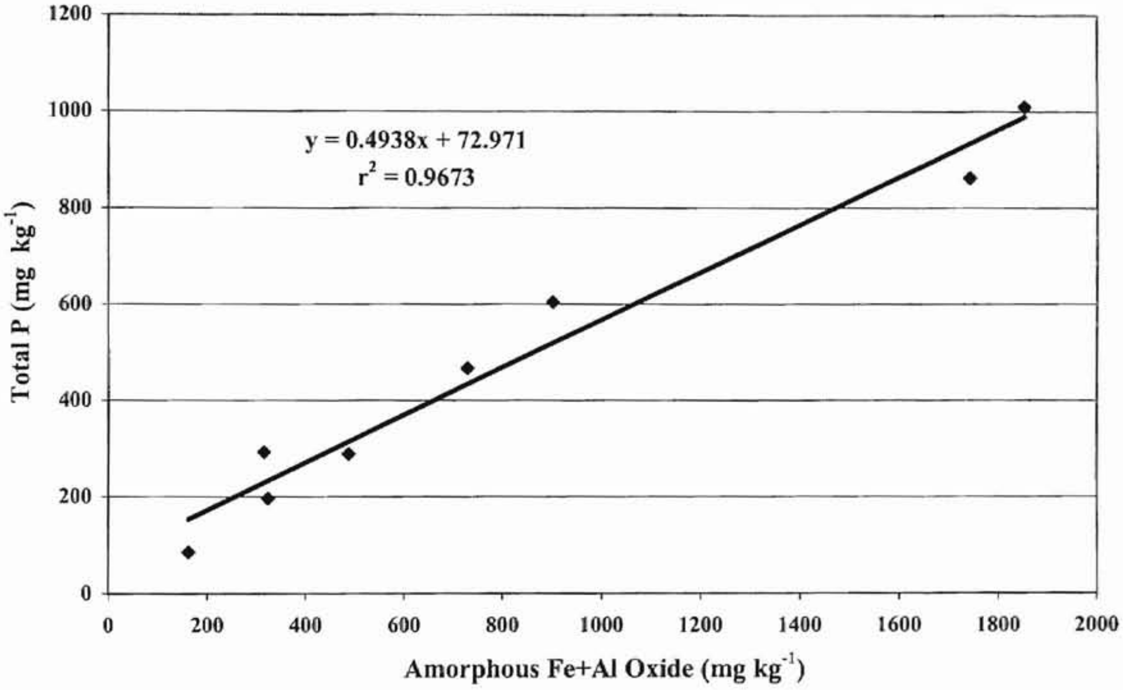


Figure 4. Correlation of amorphous iron and aluminum oxide extracted from soils in relation to total P extracted from the 0-1cm section of destructively sampled columns.

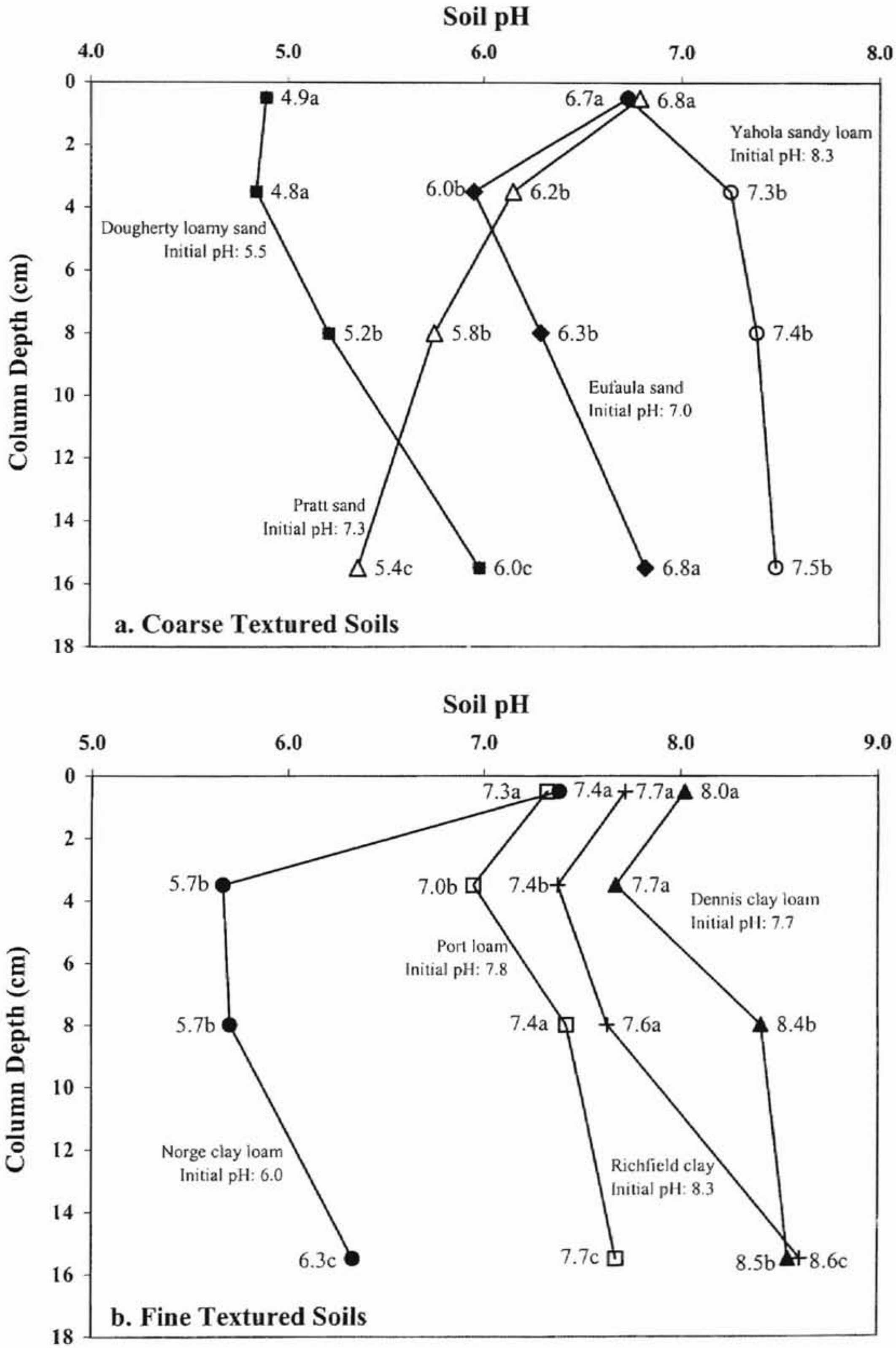


Figure 5a-b. Soil pH with column depth for coarse and fine textured soils leached with swine lagoon effluent (pH 7.9).

APPENDICES

APPENDIX I

Laboratory Analysis From P Fractionation Project

Table I - 1
Dennis loam P fraction concentrations at time zero

Sample #	Treatment	Inorganic P Fractions (mg/kg)					Total P Fractions
		Soluble/ Loosely Bound P	Al-P	Fe-P	Reductant P	Ca-P	
44	Dennis, Check, Unsaturated, Rep. 1	19.5	20.3	27.4	0.0	6.2	73.4
1	Dennis, Check, Unsaturated, Rep. 2	23.4	18.7	31.3	0.0	6.9	80.3
5	Dennis, Check, Unsaturated, Rep. 3	20.7	16.5	32.9	0 0	7.1	77 2
28	Dennis, Check, Saturated, Rep. 1	21.6	21.4	30.3	0.0	5.9	79.2
3	Dennis, Check, Saturated, Rep. 2	20.9	26.9	34.8	0.0	6.1	88.7
20	Dennis, Check, Saturated, Rep. 3	20.4	23.7	33.1	0.0	6.5	83.7
[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]
41	Dennis, 0-2 ft, Unsaturated, Rep. 1	35.4	22.5	32.3	0 0	6.5	96.7
15	Dennis, 0-2 ft, Unsaturated, Rep. 2	31.6	21.7	35.6	0.0	5.8	94.7
13	Dennis, 0-2 ft, Unsaturated, Rep. 3	38.9	19.9	31.9	0.0	6.5	97.2
24	Dennis, 0-2 ft, Saturated, Rep. 1	63.4	27.8	32.5	0.0	7.5	131.2
16	Dennis, 0-2 ft, Saturated, Rep. 2	54.9	24.1	37.0	0.0	8.3	124.3
37	Dennis, 0-2 ft, Saturated, Rep. 3	57.3	26.6	31.1	0.0	7.9	122.9
[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]
33	Dennis, 4-8 ft, Unsaturated, Rep. 1	35.4	21.6	30.3	0.0	6.1	93.4
4	Dennis, 4-8 ft, Unsaturated, Rep. 2	36.7	19.5	34.9	0.0	5.4	96.5
38	Dennis, 4-8 ft, Unsaturated, Rep. 3	42.1	24.7	33.1	0.0	6.9	106.8
2	Dennis, 4-8 ft, Saturated, Rep 1	53.5	23.3	35.1	0.0	6.9	118.8
45	Dennis, 4-8 ft, Saturated, Rep. 2	61.2	29.2	30.8	0.0	8.7	129.9
34	Dennis, 4-8 ft, Saturated, Rep. 3	57.7	24.7	34.1	0.0	7.1	123.6
[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]
19	Dennis, Sludge, Unsaturated, Rep. 1	170.4	111.9	49.5	0.0	12.2	344.0
6	Dennis, Sludge, Unsaturated, Rep. 2	157.9	115.1	41.7	0.0	10.3	325.0
25	Dennis, Sludge, Unsaturated, Rep. 3	176.0	121.3	53.1	0.0	9.5	359.9
47	Dennis, Sludge, Saturated, Rep. 1	564.3	462.5	291.5	0.0	25.7	1344.0
29	Dennis, Sludge, Saturated, Rep. 2	628.9	443.7	302.9	0.0	21.0	1396.5
48	Dennis, Sludge, Saturated, Rep. 3	601.1	459.3	298.7	0.0	29.3	1388.4

Table I - 2
Bethany silt loam P fraction concentrations at time zero

Sample #	Treatment	Inorganic P Fractions (mg/kg)					
		Soluble/ Loosely Bound P	Al-P	Fe-P	Reductant P	Ca-P	Total P Fractions
23	Bethany, Check, Unsaturated, Rep. 2	47.9	89.2	39.4	0.0	61.5	238.0
26	Bethany, Check, Unsaturated, Rep. 3	53.0	93.4	42.5	0.0	63.4	252.3
43	Bethany, Check, Saturated, Rep. 1	46.8	92.3	38.7	0.0	56.6	234.4
14	Bethany, Check, Saturated, Rep. 2	48.0	93.7	37.3	0.0	59.3	238.3
7	Bethany, Check, Saturated, Rep. 3	45.7	92.9	36.2	0.0	55.1	229.9
[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]
18	Bethany, 0-2 ft, Unsaturated, Rep. 1	53.7	90.8	36.5	0.0	62.2	243.2
35	Bethany, 0-2 ft, Unsaturated, Rep. 2	55.2	93.1	39.8	0.0	61.0	249.1
12	Bethany, 0-2 ft, Unsaturated, Rep. 3	51.2	94.0	40.7	0.0	58.3	244.2
11	Bethany, 0-2 ft, Saturated, Rep. 1	93.1	152.6	51.6	0.0	76.7	374.0
31	Bethany, 0-2 ft, Saturated, Rep. 2	91.8	145.2	48.6	0.0	78.1	363.7
32	Bethany, 0-2 ft, Saturated, Rep. 3	96.4	149.9	49.1	0.0	79.3	374.7
[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]
27	Bethany, 4-8 ft, Unsaturated, Rep. 1	50.4	88.6	42.2	0.0	62.3	243.5
17	Bethany, 4-8 ft, Unsaturated, Rep. 2	55.9	95.5	37.4	0.0	57.8	246.6
10	Bethany, 4-8 ft, Unsaturated, Rep. 3	52.1	96.4	41.8	0.0	59.3	249.6
22	Bethany, 4-8 ft, Saturated, Rep. 1	92.4	154.3	58.7	0.0	70.3	375.7
36	Bethany, 4-8 ft, Saturated, Rep. 2	90.9	146.0	51.6	0.0	78.8	367.3
9	Bethany, 4-8 ft, Saturated, Rep. 3	95.7	149.8	50.9	0.0	76.0	372.4
[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]	[redacted]
40	Bethany, Sludge, Unsaturated, Rep. 1	243.7	231.9	113.8	0.0	52.4	641.8
30	Bethany, Sludge, Unsaturated, Rep. 2	266.1	237.5	109.4	0.0	55.3	668.3
21	Bethany, Sludge, Unsaturated, Rep. 3	253.9	221.0	107.2	0.0	59.8	641.9
46	Bethany, Sludge, Saturated, Rep. 1	773.4	465.3	202.9	0.0	89.2	1530.8
39	Bethany, Sludge, Saturated, Rep. 2	742.1	457.1	200.2	0.0	97.6	1497.0
42	Bethany, Sludge, Saturated, Rep. 3	758.0	459.7	198.3	0.0	103.8	1519.8

Table I - 3
Richfield clay loam P fraction concentrations at time zero

Sample #	Treatment	Inorganic P Fractions (mg/kg)				
		Soluble/ Loosely Bound P	Al-P and Fe-P	Reductant P	Ca-P	Total P Fractions
60	Richfield, Check, Unsaturated, Rep. 1	0.0	12.4	0.0	119.7	132.1
64	Richfield, Check, Unsaturated, Rep. 2	0.1	9.4	0.0	113.6	123.1
57	Richfield, Check, Unsaturated, Rep. 3	0.1	8.6	0.0	115.0	123.7
63	Richfield, Check, Saturated, Rep. 1	0.1	11.2	0.0	116.7	128 0
56	Richfield, Check, Saturated, Rep 2	0.1	13.5	0.0	112.4	126 0
50	Richfield, Check, Saturated, Rep. 3	0.1	14.1	0.0	120.3	134.5
	[illegible] 0-2 ft.					
53	Richfield, 0-2 ft, Unsaturated, Rep. 1	9.4	11.6	0.0	115.2	136.2
49	Richfield, 0-2 ft, Unsaturated, Rep. 2	8.7	13.4	0.0	112 7	134.8
68	Richfield, 0-2 ft, Unsaturated, Rep. 3	9.8	15 8	0.0	111 8	137 4
72	Richfield, 0-2 ft, Saturated, Rep 1	45.1	28 3	0.0	125.3	198.7
61	Richfield, 0-2 ft, Saturated, Rep. 2	48.9	30 9	0.0	123.4	203 2
52	Richfield, 0-2 ft, Saturated, Rep. 3	43.5	27.1	0.0	116.8	187.4
	[illegible] 4-8 ft.					
67	Richfield, 4-8 ft, Unsaturated, Rep. 1	8 9	10.1	0.0	111 0	130.0
58	Richfield, 4-8 ft, Unsaturated, Rep. 2	9.8	13.9	0.0	119 3	143.0
59	Richfield, 4-8 ft, Unsaturated, Rep. 3	10.2	9 6	0.0	112.4	132.2
69	Richfield, 4-8 ft, Saturated, Rep 1	49 6	24.3	0.0	122.0	195.9
51	Richfield, 4-8 ft, Saturated, Rep. 2	47 7	29.9	0.0	121.3	198.9
66	Richfield, 4-8 ft, Saturated, Rep. 3	49.5	25.1	0.0	127.1	201.7
	[illegible] Sludge					
62	Richfield, Sludge, Unsaturated, Rep. 1	104.7	40.1	0.0	139.4	284.2
54	Richfield, Sludge, Unsaturated, Rep. 2	108.9	33.8	0.0	142.1	284.8
55	Richfield, Sludge, Unsaturated, Rep. 3	111.2	31.3	0.0	137.0	279.5
65	Richfield, Sludge, Saturated, Rep. 1	241.3	112.3	0.0	198.3	551.9
71	Richfield, Sludge, Saturated, Rep. 2	243.9	127.9	0.0	204.9	576.7
70	Richfield, Sludge, Saturated, Rep. 3	237.5	119.4	0.0	201.2	558.1

Table I - 4
Dennis loam P fraction concentrations after 35 day incubation

Sample #	Treatment	Inorganic P Fractions (mg/kg)					
		Soluble/ Loosely Bound P	Al-P	Fe-P	Reductant P	Ca-P	Total P Fractions
44	Dennis, Check, Unsaturated, Rep. 1	14.5	17.4	23.6	0.0	8.6	64.1
1	Dennis, Check, Unsaturated, Rep. 2	37.9	18.4	44.8	0.0	7.8	108.9
5	Dennis, Check, Unsaturated, Rep. 3	36.3	19.3	35.9	0.0	6.5	98.0
28	Dennis, Check, Saturated, Rep. 1	0.0	22.1	60.3	0.0	5.8	88.3
3	Dennis, Check, Saturated, Rep. 2	0.0	26.6	66.7	0.0	5.4	98.6
20	Dennis, Check, Saturated, Rep. 3	0.0	21.5	64.3	0.0	3.8	89.6
41	Dennis, 0-2 ft, Unsaturated, Rep. 1	lost	24.0	46.6	0.0	6.4	76.9
15	Dennis, 0-2 ft, Unsaturated, Rep. 2	3.9	39.5	34.7	0.0	6.8	84.9
13	Dennis, 0-2 ft, Unsaturated, Rep. 3	24.6	28.0	35.1	0.0	8.6	96.3
24	Dennis, 0-2 ft, Saturated, Rep. 1	0.0	42.9	83.1	0.0	6.1	132.2
16	Dennis, 0-2 ft, Saturated, Rep. 2	0.0	41.5	90.1	0.0	6.6	138.1
37	Dennis, 0-2 ft, Saturated, Rep. 3	0.0	35.2	106.0	0.0	4.8	146.0
33	Dennis, 4-8 ft, Unsaturated, Rep. 1	40.4	0.0	29.8	0.0	8.6	78.8
4	Dennis, 4-8 ft, Unsaturated, Rep. 2	35.5	24.2	39.4	0.0	6.1	105.1
38	Dennis, 4-8 ft, Unsaturated, Rep. 3	18.1	23.3	31.5	0.0	7.6	80.5
2	Dennis, 4-8 ft, Saturated, Rep. 1	0.0	38.6	114.4	0.0	7.6	160.6
45	Dennis, 4-8 ft, Saturated, Rep. 2	0.0	39.5	110.1	0.0	7.7	157.4
34	Dennis, 4-8 ft, Saturated, Rep. 3	0.0	30.0	97.4	0.0	6.0	133.4
19	Dennis, Sludge, Unsaturated, Rep. 1	48.8	165.6	95.9	0.0	14.3	324.6
6	Dennis, Sludge, Unsaturated, Rep. 2	78.4	103.1	70.6	0.0	11.6	263.7
25	Dennis, Sludge, Unsaturated, Rep. 3	100.2	166.3	75.4	0.0	12.7	354.5
47	Dennis, Sludge, Saturated, Rep. 1	496.2	457.1	316.2	0.0	49.2	1318.7
29	Dennis, Sludge, Saturated, Rep. 2	496.5	623.5	388.6	0.0	47.7	1556.2
48	Dennis, Sludge, Saturated, Rep. 3	475.5	515.8	335.8	0.0	42.8	1369.8

Table I - 5
Bethany silt loam P fraction concentrations after 35 day incubation

Sample #	Treatment	Inorganic P Fractions (mg/kg)					
		Soluble/ Loosely Bound P	Al-P	Fe-P	Reductant P	Ca-P	Total P Fractions
8	Bethany, Check, Unsaturated, Rep. 1	52.3	90.1	44.9	0.0	66.9	254.3
23	Bethany, Check, Unsaturated, Rep. 2	52.1	93.9	38.2	0.0	59.4	243.6
26	Bethany, Check, Unsaturated, Rep. 3	55.5	97.2	40.2	0.0	64.1	257.0
43	Bethany, Check, Saturated, Rep. 1	3.5	109.4	62.1	0.0	86.1	261.1
14	Bethany, Check, Saturated, Rep. 2	14.9	108.8	66.0	0.0	68.4	258.0
7	Bethany, Check, Saturated, Rep. 3	7.1	117.4	84.8	0.0	74.4	283.7
18	Bethany, 0-2 ft, Unsaturated, Rep. 1	48.6	100.7	53.8	0.0	69.3	272.3
35	Bethany, 0-2 ft, Unsaturated, Rep. 2	41.8	83.0	37.7	0.0	76.1	238.6
12	Bethany, 0-2 ft, Unsaturated, Rep. 3	56.6	89.8	46.5	0.0	57.6	250.5
11	Bethany, 0-2 ft, Saturated, Rep. 1	41.6	144.0	74.1	0.0	71.2	330.9
31	Bethany, 0-2 ft, Saturated, Rep. 2	23.5	160.6	85.0	0.0	75.6	344.7
32	Bethany, 0-2 ft, Saturated, Rep. 3	9.4	139.0	92.2	0.0	66.9	307.5
27	Bethany, 4-8 ft, Unsaturated, Rep. 1	49.7	110.6	40.5	0.0	64.8	265.6
17	Bethany, 4-8 ft, Unsaturated, Rep. 2	44.9	96.7	49.3	0.0	65.6	256.5
10	Bethany, 4-8 ft, Unsaturated, Rep. 3	59.1	90.3	47.2	0.0	77.2	273.8
22	Bethany, 4-8 ft, Saturated, Rep. 1	6.4	144.8	68.0	0.0	52.4	271.6
36	Bethany, 4-8 ft, Saturated, Rep. 2	19.0	161.9	88.8	0.0	59.9	329.7
9	Bethany, 4-8 ft, Saturated, Rep. 3	24.3	177.2	106.3	0.0	75.4	383.2
40	Bethany, Sludge, Unsaturated, Rep. 1	130.2	274.6	79.5	0.0	75.9	560.2
30	Bethany, Sludge, Unsaturated, Rep. 2	120.7	271.6	64.0	0.0	80.5	536.7
21	Bethany, Sludge, Unsaturated, Rep. 3	173.7	355.4	70.0	0.0	77.3	676.3
46	Bethany, Sludge, Saturated, Rep. 1	558.8	574.6	152.3	0.0	102.3	1387.9
39	Bethany, Sludge, Saturated, Rep. 2	741.4	587.0	174.9	0.0	83.4	1586.7
42	Bethany, Sludge, Saturated, Rep. 3	717.4	516.1	145.7	0.0	77.9	1457.2

Table I - 6

Richfield clay loam P fraction concentrations after 35 day incubation

Sample #	Treatment	Inorganic P Fractions (mg/kg)				Total P Fractions
		Soluble/ Loosely Bound P	Al-P and Fe-P	Reductant P	Ca-P	
60	Richfield, Check, Unsaturated, Rep. 1	0.1	7.5	0.0	117.6	125.1
64	Richfield, Check, Unsaturated, Rep. 2	0.1	8.5	0.0	99.6	108.1
57	Richfield, Check, Unsaturated, Rep. 3	lost	8.0	0.0	122.1	130.1
63	Richfield, Check, Saturated, Rep. 1	0.1	11.5	0.0	122.1	133.7
56	Richfield, Check, Saturated, Rep. 2	0.0	11.5	0.0	124.3	135.9
50	Richfield, Check, Saturated, Rep. 3	0.0	12.9	0.0	132.2	145.1
53	Richfield, 0-2 ft, Unsaturated, Rep. 1	0.2	16.1	0.0	117.6	133.8
49	Richfield, 0-2 ft, Unsaturated, Rep. 2	0.2	16.5	0.0	119.8	136.5
68	Richfield, 0-2 ft, Unsaturated, Rep. 3	lost	14.2	0.0	108.6	122.7
72	Richfield, 0-2 ft, Saturated, Rep. 1	0.2	36.9	0.0	127.7	164.8
61	Richfield, 0-2 ft, Saturated, Rep. 2	0.1	51.0	0.0	106.3	157.5
52	Richfield, 0-2 ft, Saturated, Rep. 3	2.2	50.8	0.0	lost	53.0
67	Richfield, 4-8 ft, Unsaturated, Rep. 1	0.4	15.1	0.0	104.1	119.6
58	Richfield, 4-8 ft, Unsaturated, Rep. 2	0.4	17.2	0.0	117.6	135.2
59	Richfield, 4-8 ft, Unsaturated, Rep. 3	0.8	15.1	0.0	130.0	145.9
69	Richfield, 4-8 ft, Saturated, Rep. 1	0.2	54.7	0.0	118.7	173.6
51	Richfield, 4-8 ft, Saturated, Rep. 2	0.1	55.9	0.0	140.1	196.1
66	Richfield, 4-8 ft, Saturated, Rep. 3	0.5	51.7	0.0	126.6	178.8
62	Richfield, Sludge, Unsaturated, Rep. 1	12.7	60.6	0.0	175.0	248.4
54	Richfield, Sludge, Unsaturated, Rep. 2	14.5	55.5	0.0	171.6	241.5
55	Richfield, Sludge, Unsaturated, Rep. 3	16.5	53.7	0.0	158.1	228.4
65	Richfield, Sludge, Saturated, Rep. 1	33.1	204.2	0.0	351.8	589.1
71	Richfield, Sludge, Saturated, Rep. 2	18.9	214.4	0.0	248.2	481.5
70	Richfield, Sludge, Saturated, Rep. 3	35.4	216.8	0.0	319.2	571.3

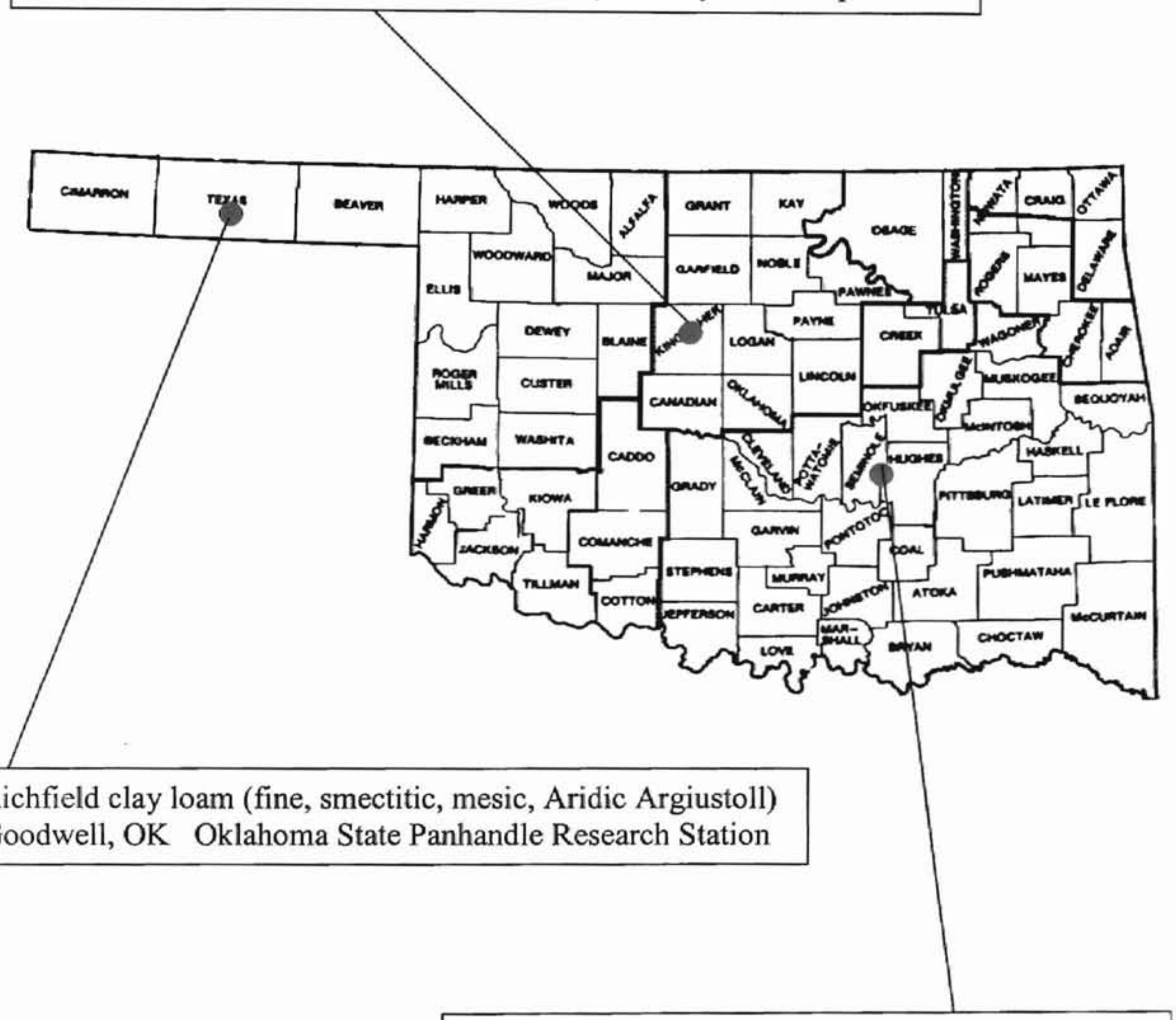
Table I - 7

Total P concentrations and means for soils and swine waste sources

Soil Series	Total P (mg P kg soil^{-1})	Average
Dennis 1	166.00	
Dennis 2	135.00	
Dennis 3	170.00	157.00
Bethany 1	343.00	
Bethany 2	338.00	
Bethany 3	357.00	346.00
Richfield 1	264.00	
Richfield 2	255.00	
Richfield 3	264.00	261.00

Waste Source	Total P (mg P kg soil^{-1})	Average
0-2 ft Effluent 1	74.00	
0-2 ft Effluent 2	71.00	
0-2 ft Effluent 3	72.00	72.33
8-10 ft Effluent 1	70.00	
8-10 ft Effluent 2	65.00	
8-10 ft Effluent 3	70.00	68.33
Sludge 1	3013.00	
Sludge 2	2847.00	
Sludge 3	3047.00	2969.00

Bethany silt loam (fine, mixed, superactive, thermic Pachic Paleustoll)
Hennessey, OK   Oklahoma State University fertility research plots



Richfield clay loam (fine, smectitic, mesic, Aridic Argiustoll)
Goodwell, OK   Oklahoma State Panhandle Research Station

Dennis loam (fine, mixed, thermic Aquic Argiudoll)
Holdenville, OK   Private hog producer's field

Figure I-1. Locations where soil samples were collected for incubation study.

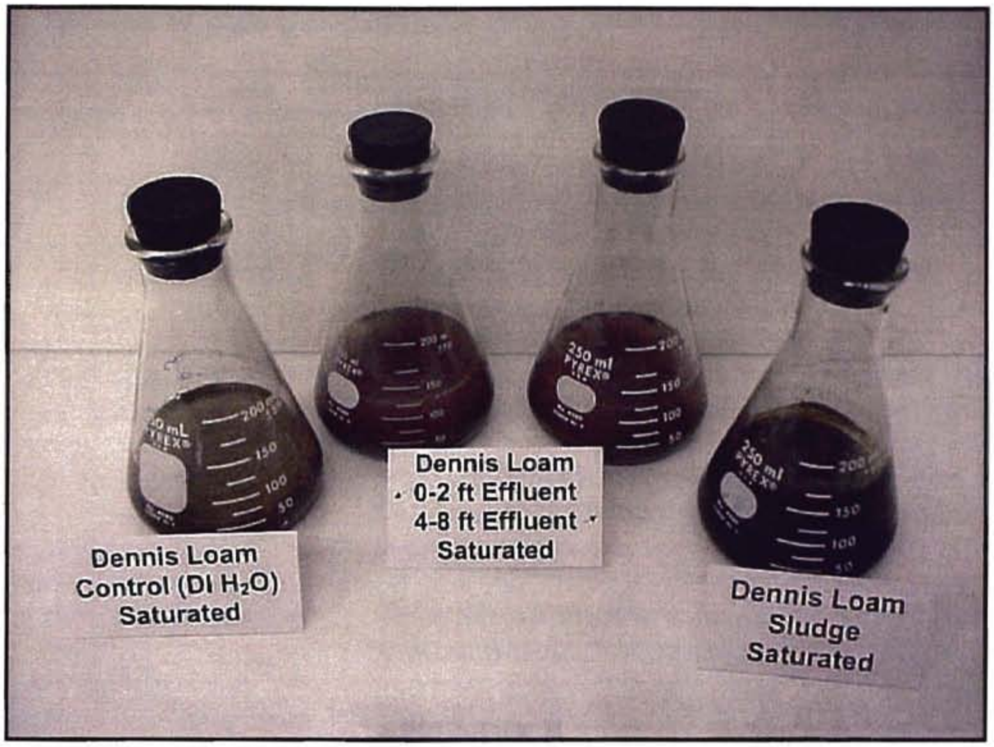


Figure I-2a. Incubation flasks with different treatments applied to Dennis loam.

Figure I-2b. Assorted saturated and nonsaturated treatments.

APPENDIX II

Additional Tables and Figures From Column Study and Batch Isotherm Tests

Table II-1. Location of eight B-horizon soils used in the leaching column project.

Soil Series	County	Legal Land Description
Dennis	Hughes	SW¼, NE¼, SW¼, Sec. 4, R.9 E., T. 9 N. I.M. 13 mi. north and 2.5 mi. east of Holdenville, OK
Dougherty	Payne	SE¼, SE¼, NE¼, Sec. 36, R.2 E., T.18 N. I.M. 8 mi. south and 1 mi. east of Stillwater, OK
Eufaula	Payne	SW¼, SE¼, SE¼, Sec. 4, R.1 E., T.17 N. I.M. 10 mi. south and 8 mi. east of Stillwater, OK
Norge	Payne	SW¼, SE¼, SW¼, NE¼, Sec. 16, R.2 E., T.19 N. I.M. Oklahoma State University Agronomy Research Station in Stillwater, OK
Port	Payne	SW¼, SE¼, SW¼, SW¼, Sec. 2, R.2 E., T.20 N. I.M. 8 mi. north and 0.5 mi. east of Stillwater, OK
Pratt	Kingfisher	SE¼, SW¼, SE¼, Sec. 1, R.9 W., T.19 N. I.M. 12mi. west and 3 mi. north of Hennessey, OK
Richfield	Texas	SW¼, NE¼, SW¼, Sec. 36, R.13 E., T.2 N. C.M. Oklahoma State University Panhandle Research Station in Goodwell, OK
Yahola	Payne	NE¼, SE¼, SW¼, SE¼, Sec. 7, R.3 E., T.17 N. I.M. 1.5 mi. south and 0.75 mi. east of Perkins, OK

Table II-2. WCOLUMN model output for the Dougherty leaching column.

Pore Volume Captured	Observed P Concentration (mg L^{-1})	Model P Concentration (mg L^{-1})	Difference (mg L^{-1})	Chi-Squared
0.25	0.15	5.61E-32	0.15	0.023
0.50	0.15	6.06E-14	0.15	0.023
0.75	0.15	5.51E-08	0.15	0.023
1.25	0.15	2.47E-03	0.15	0.022
1.50	0.15	0.03	0.12	0.014
1.75	0.15	0.19	-0.04	0.001
2.00	0.30	0.66	-0.36	0.133
2.25	0.27	1.70	-1.43	2.038
2.50	1.04	3.45	-2.41	5.787
2.75	5.19	5.93	-0.74	0.546
3.00	6.78	9.03	-2.25	5.060
3.25	10.30	12.54	-2.24	5.013
3.50	16.20	16.22	-0.02	0.001
3.75	22.40	19.87	2.53	6.415
4.00	27.40	23.30	4.10	16.775
4.25	30.20	26.42	3.78	14.260
4.50	30.20	29.17	1.03	1.068
5.00	32.70	33.49	-0.79	0.617
5.25	34.10	35.16	-1.06	1.131
5.50	33.00	36.42	-3.42	11.674
5.75	34.00	37.46	-3.46	12.002
6.00	34.10	38.29	-4.19	17.569
6.25	34.50	39.00	-4.50	17.621
6.50	34.20	39.60	-5.40	18.150
6.75	34.10	40.10	-6.00	18.700
7.00	34.40	40.20	-5.80	18.426
7.25	No Sample	40.30	-	-
7.50	No Sample	40.40	-	-
7.75	No Sample	40.40	-	-

Table II-3. WCOLUMN model output for the Eufaula leaching column.

Pore Volume Captured	Observed P Concentration (mg L^{-1})	Model P Concentration (mg L^{-1})	Difference (mg L^{-1})	Chi-Squared
0.25	0.05	0.05	0.00	0.000
0.50	0.07	1.37	-1.31	1.706
0.75	1.72	4.43	-2.71	7.333
1.00	5.45	8.07	-2.62	6.864
1.25	10.90	11.64	-0.74	0.545
1.50	16.80	14.89	1.91	3.640
1.75	21.90	17.78	4.12	16.999
2.25	24.40	22.52	1.88	3.545
2.50	24.30	24.45	-0.15	0.022
2.75	24.70	26.14	-1.44	2.071
3.00	27.30	27.62	-0.32	0.105
3.25	29.40	28.93	0.47	0.220
3.50	29.00	30.09	-1.09	1.178
3.75	31.10	31.11	-0.01	0.000
4.00	33.20	32.02	1.18	1.396
4.25	33.60	32.83	0.77	0.594
4.50	33.00	33.55	-0.55	0.307
4.75	33.00	34.20	-1.20	1.450
5.00	34.10	34.79	-0.69	0.473
5.25	35.30	35.31	-0.01	0.000
5.50	35.30	35.79	-0.49	0.237
5.75	36.10	36.22	-0.12	0.013
6.00	36.10	36.60	-0.50	0.254
6.25	36.10	36.96	-0.86	0.733
6.50	36.20	37.30	-1.10	1.170
6.75	No Sample	37.40	-	-
7.00	No Sample	37.50	-	-
7.25	No Sample	37.60	-	-

Table II-4. WCOLUMN model output for the Pratt leaching column.

Pore Volume Captured	Observed P Concentration (mg L^{-1})	Model P Concentration (mg L^{-1})	Difference (mg L^{-1})	Chi-Squared
0.25	0.11	2.81E-09	0.11	0.012
0.50	0.12	6.41E-04	0.12	0.014
0.75	0.12	0.04	0.08	0.007
1.00	0.10	0.28	-0.18	0.034
1.25	0.15	0.96	-0.81	0.660
1.50	0.10	2.15	-2.05	4.223
1.75	0.45	3.81	-3.36	11.298
2.00	1.88	5.81	-3.93	15.463
2.25	5.21	8.03	-2.82	7.941
2.50	9.90	10.34	-0.44	0.198
2.75	12.90	12.67	0.23	0.051
3.00	16.60	14.95	1.65	2.717
3.25	19.70	17.14	2.56	6.579
3.50	21.40	19.20	2.20	4.853
3.75	23.20	21.12	2.08	4.315
4.00	25.50	22.91	2.59	6.728
4.25	24.90	24.55	0.35	0.125
4.50	26.20	26.05	0.15	0.023
4.75	27.30	27.42	-0.12	0.014
5.00	28.80	28.66	0.14	0.019
5.25	30.50	29.79	0.71	0.503
5.50	31.50	30.81	0.69	0.474
5.75	33.30	31.73	1.57	2.452
6.00	33.30	32.57	0.73	0.538
6.25	32.30	33.32	-1.02	1.035
6.50	32.40	33.99	-1.59	2.538
6.75	34.00	34.60	-0.60	0.362
7.00	34.00	35.15	-1.15	1.322
7.25	34.00	35.64	-1.64	2.699
7.50	34.60	36.09	-1.49	2.210
7.75	34.00	36.49	-2.49	6.177
8.00	34.00	36.84	-2.84	8.089
8.50	34.80	37.50	-2.70	7.512
9.00	35.80	37.95	-2.15	4.636
9.25	35.30	38.16	-2.86	8.207
9.50	37.00	38.35	-1.35	1.835
9.75	37.30	38.53	-1.23	1.503
10.00	37.70	38.68	1.62	2.623
10.50	37.90	38.94	1.36	1.839
11.00	38.00	39.16	1.14	1.305
11.50	37.60	39.33	0.97	0.938
12.00	37.90	39.47	0.83	0.685

Table II-5. WCOLUMN model output for destructively sampled Dennis column.

Distance in Column (cm)	Leaching Time (days)	Model P Concentration (mg kg^{-1})	Observed P Concentration (mg kg^{-1})
0.00	100	34.90	22.99
1.05	100	24.98	
2.11	100	13.87	
3.16	100	5.68	7.18
4.21	100	1.66	
5.26	100	0.34	
6.32	100	0.05	
7.37	100	4.47E-03	
8.42	100	2.86E-04	0.27
9.47	100	1.22E-05	
10.53	100	3.49E-07	
11.58	100	6.64E-09	
12.63	100	8.38E-11	
13.68	100	7.00E-13	
14.74	100	3.87E-15	0.72
15.79	100	1.41E-17	
16.84	100	3.41E-20	
17.89	100	5.42E-23	
18.95	100	5.68E-26	
20.00	100	3.92E-29	0.00

Table II-6. WCOLUMN model output for destructively sampled Norge column.

Distance in Column (cm)	Leaching Time (days)	Model P Concentration (mg L^{-1})	Observed P Concentration (mg kg^{-1})
0.00	100	32.31	24.73
1.05	100	19.23	
2.11	100	7.71	
3.16	100	1.94	1.71
4.21	100	0.29	
5.26	100	0.03	
6.32	100	1.34E-03	
7.37	100	3.89E-05	
8.42	100	6.37E-07	0.04
9.47	100	5.84E-09	
10.53	100	2.99E-11	
11.58	100	8.49E-14	
12.63	100	1.34E-16	
13.68	100	1.17E-19	
14.74	100	5.62E-23	0.09
15.79	100	1.49E-26	
16.84	100	2.19E-30	
17.89	100	1.76E-34	
18.95	100	7.81E-39	
20.00	100	1.91E-43	0.00

Table II-7. WCOLUMN model output for destructively sampled Richfield column.

Distance in Column (cm)	Leaching Time (days)	Model P Concentration (mg L^{-1})	Observed P Concentration (mg kg^{-1})
0.00	100	38.98	34.26
1.05	100	35.30	
2.11	100	29.49	
3.16	100	22.16	17.79
4.21	100	14.68	
5.26	100	8.43	
6.32	100	4.14	
7.37	100	1.74	
8.42	100	0.61	4.15
9.47	100	0.18	
10.53	100	0.05	
11.58	100	9.35E-03	
12.63	100	1.61E-03	
13.68	100	2.30E-04	
14.74	100	2.73E-05	1.64
15.79	100	2.68E-06	
16.84	100	2.18E-07	
17.89	100	1.47E-08	
18.95	100	8.17E-10	
20.00	100	3.75E-11	0.00

Table II-8. WCOLUMN model output for destructively sampled Port column.

Distance in Column (cm)	Leaching Time (days)	Model P Concentration (mg L-1)	Observed P Concentration (mg kg^{-1})
0.00	100	39.86	49.27
1.05	100	37.43	
2.11	100	33.12	
3.16	100	26.98	24.12
4.21	100	19.81	
5.26	100	12.89	
6.32	100	7.33	
7.37	100	3.63	
8.42	100	1.54	1.14
9.47	100	0.56	
10.53	100	0.17	
11.58	100	0.05	
12.63	100	0.01	
13.68	100	1.89E-03	
14.74	100	2.99E-04	0.84
15.79	100	3.99E-05	
16.84	100	4.48E-06	
17.89	100	4.23E-07	
18.95	100	3.35E-08	
20.00	100	2.23E-09	0.00

Table II-9. WCOLUMN model output for destructively sampled Yahola column.

Distance in Column (cm)	Leaching Time (days)	Model P Concentration (mg L-1)	Observed P Concentration (mg kg^{-1})
0.00	40	30.52	29.90
1.05	40	29.31	
2.11	40	28.06	
3.16	40	26.78	21.19
4.21	40	25.48	
5.26	40	24.16	
6.32	40	22.83	
7.37	40	21.50	
8.42	40	20.17	20.95
9.47	40	18.86	
10.53	40	17.57	
11.58	40	16.30	
12.63	40	15.06	
13.68	40	13.87	
14.74	40	12.71	8.24
15.79	40	11.61	
16.84	40	10.56	
17.89	40	9.56	
18.95	40	8.62	
20.00	40	7.74	

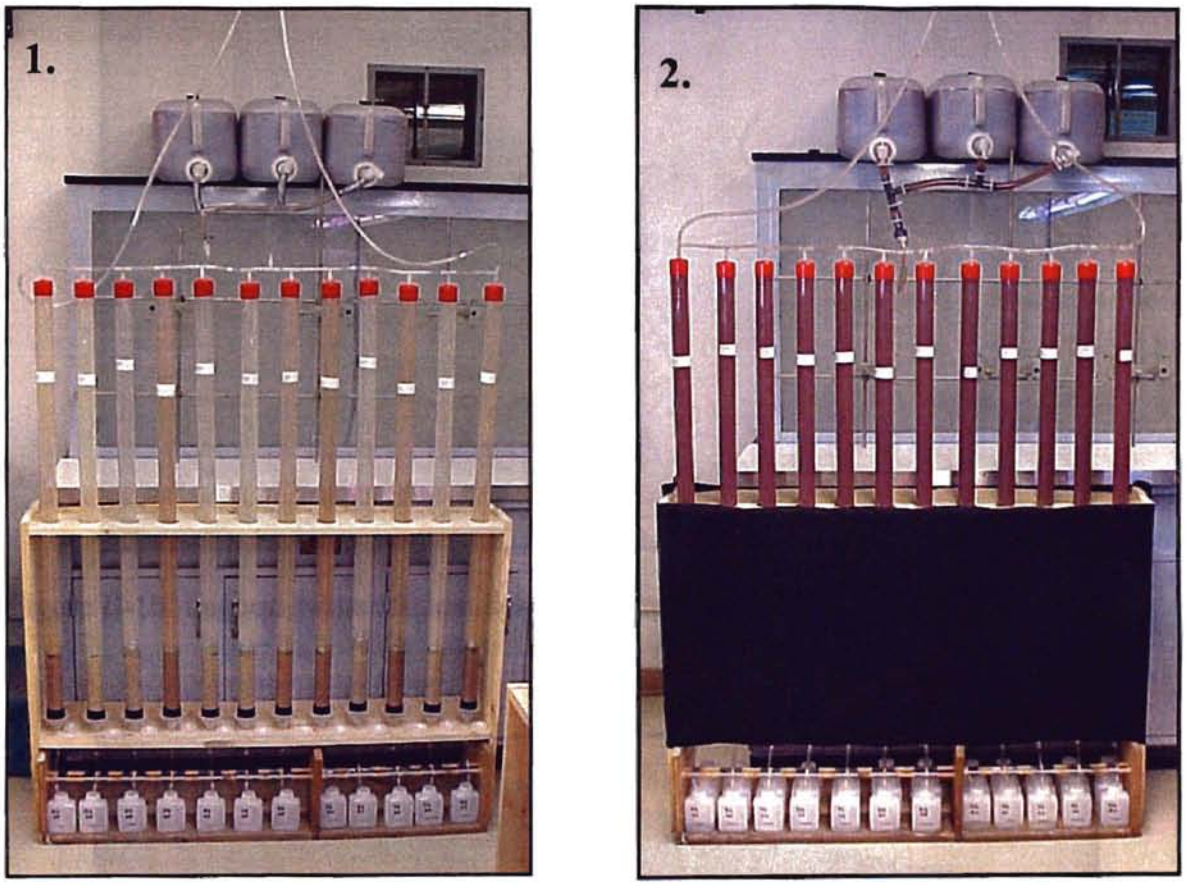


Figure II-1a. Soil columns (1) before the swine lagoon effluent leaching experiment began and (2) after a 1.5 m hydraulic head of effluent was applied.

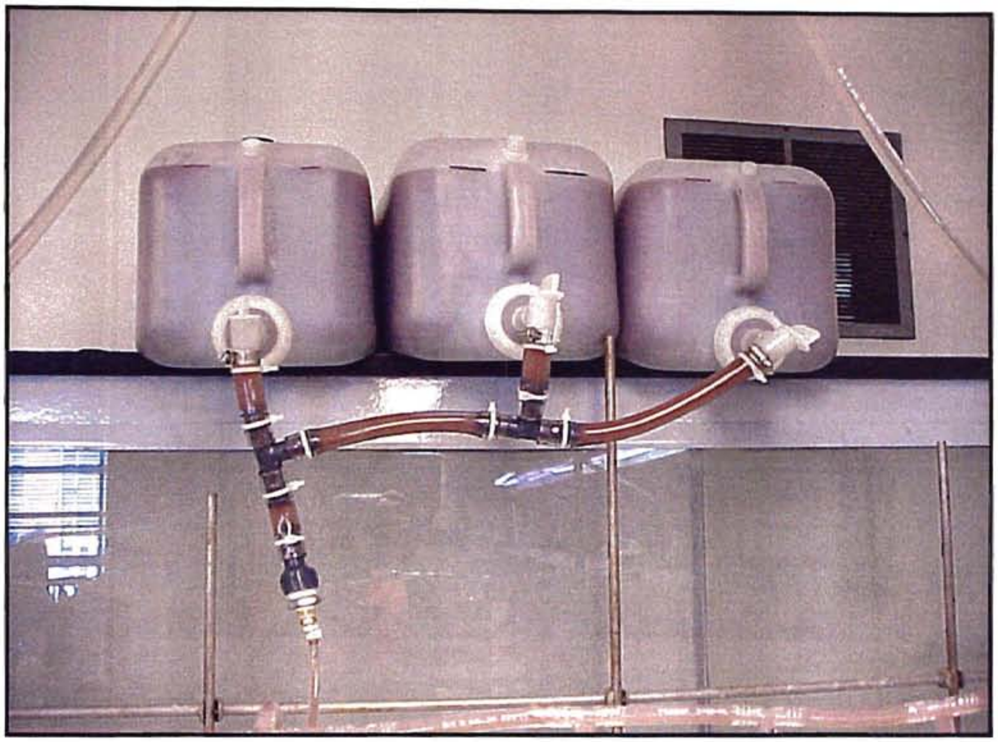


Figure II-1b. Lagoon swine effluent delivery system for column study.

Figure II-1c. Leachate collection bottles for coarse textured soils.

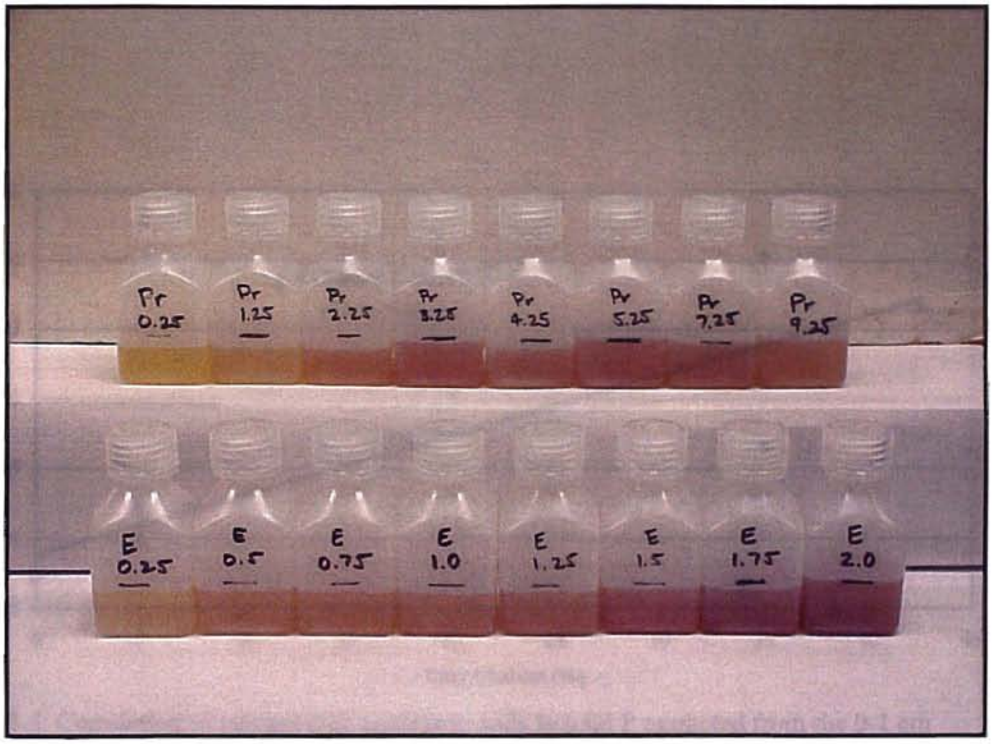


Figure II-1d. Color variation of leachate captured from Pratt sand and Eufaula sand. Notice the purple coloration due to the swine effluent as pore volume increments increase.

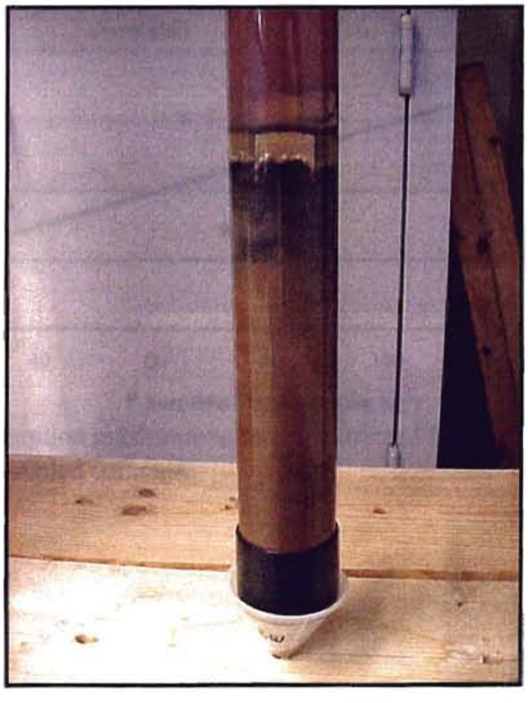


Figure II-1e. Organics accumulation near top portion of Yahola sandy loam column.

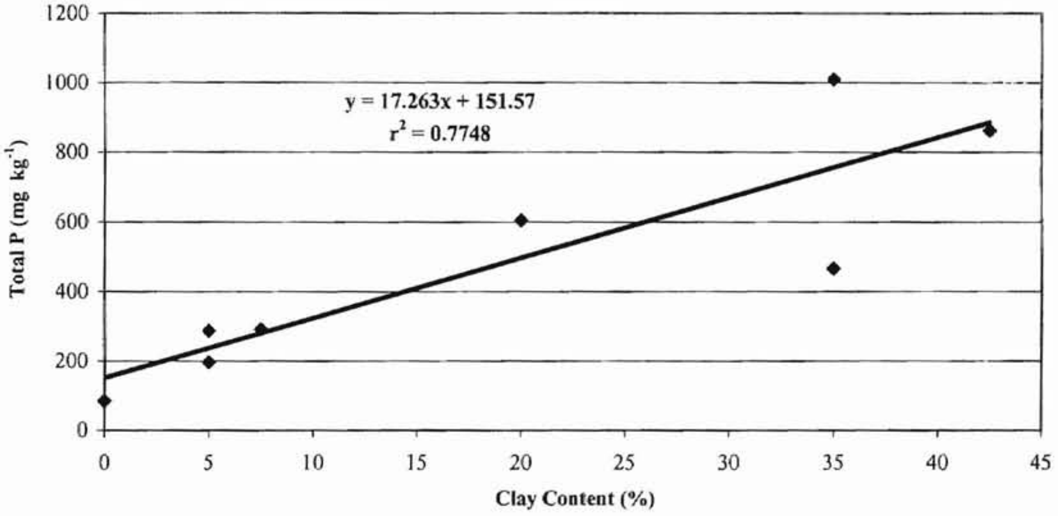


Figure II-2. Correlation of percent clay content in soils to total P extracted from the 0-1 cm section of destructively sampled columns.

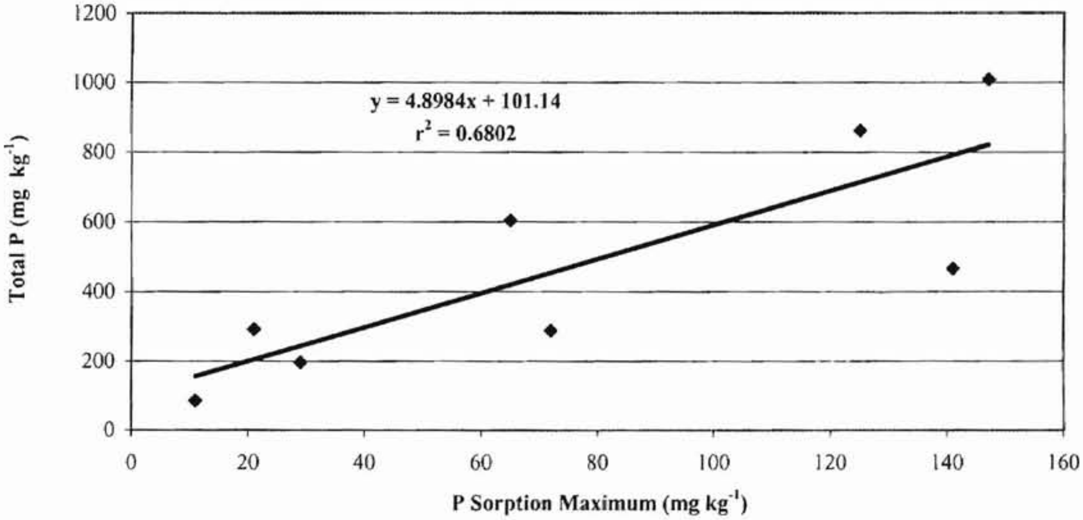


Figure II-3. Correlation of P sorption maximum to total P extracted from the 0-1cm section of destructively sampled columns.

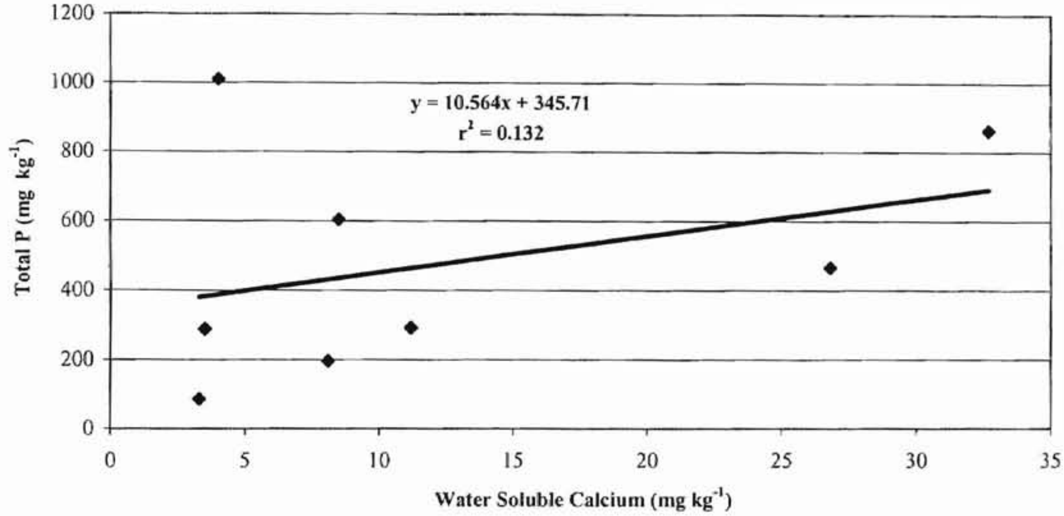


Figure II-4. Correlation of water soluble calcium extracted from soils to total P extracted from the 0-1cm section of destructively sampled columns.

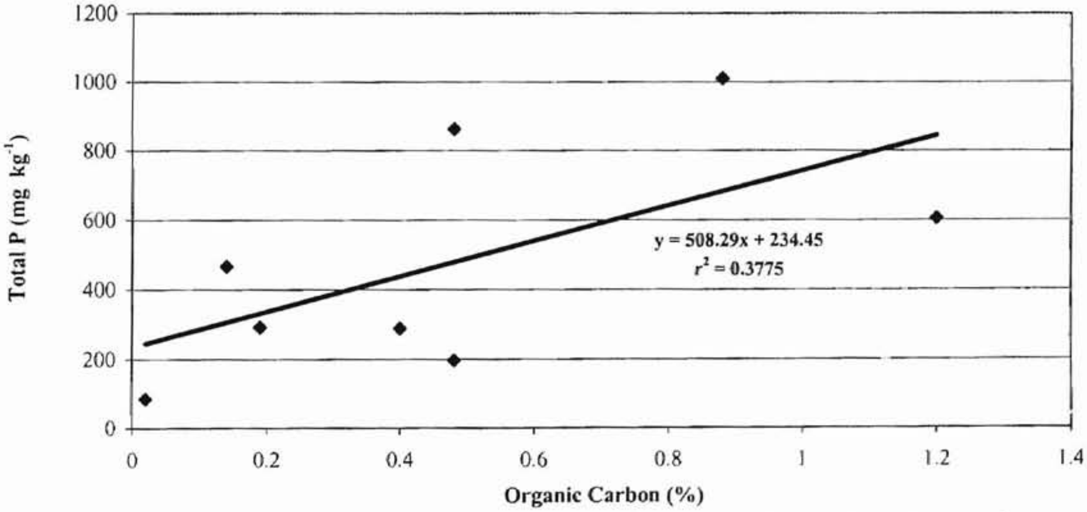


Figure II-5. Correlation of percent organic carbon to total P extracted from the 0-1cm section of destructively sampled columns.

VITA

Damon Gene Wright

Candidate for the Degree of

Master of Science

Thesis: PHOSPHORUS DISTRIBUTION AND MOBILITY THROUGH SOILS USING SWINE LAGOON WASTE

Major Field: Plant and Soil Sciences

Biographical:

Personal Data: Born in Oklahoma City, Oklahoma, on January 13, 1975

Education: Graduated from Yukon High School, Yukon, Oklahoma, in May 1993; Received Associates of Science degree from Redlands Community College, El Reno, Oklahoma, in May, 1995; Received Bachelor of Science degree in Plant and Soil Sciences from Oklahoma State University, Stillwater, Oklahoma, in May, 1998; Completed requirements for Masters of Science degree with a major in Plant and Soil Sciences at Oklahoma State University, Stillwater, Oklahoma, in May, 2002

Professional Experiences: Undergraduate Teaching and Research Assistant, Department of Plant and Soil Sciences, Oklahoma State University, August 1997 to May 1998; Graduate Teaching and Research Assistant, Department of Plant and Soil Sciences, Oklahoma State University, June 1998 to May 2001; Project Manager/Environmental Soil Scientist, Apex Environmental, Inc., August 2000 to May 2002

Professional Organizations: National Society of Consulting Soil Scientists, Professional Soil Scientist Association of Oklahoma, American Society of Agronomy, Soil Science Society of America, National Ground Water Association, Gamma Sigma Delta Honor Society of Agriculture