

ABSTRACT

Roberts, John Christopher. Impact of Manure and Soil Test Phosphorus on Phosphorus Runoff from Soils Subjected to Simulated Rainfall. (Under the direction of Daniel W. Israel.)

Runoff from agricultural fields amended with animal manure or fertilizer is a source of phosphorus (P) pollution to surface waters, which can have harmful effects such as eutrophication. The objectives of this study were to evaluate the impact of soil P status and the P composition of manure sources on P in runoff, characterize the effects of manure sources on mass loss of dissolved reactive P (DRP), total dissolved P (TDP), algal available P (AAP) and total P (TP) in runoff, and enhance the PLAT database with respect to soluble P attenuating factor (SPAF) and non-soluble P attenuating factor (NSPAF) values. Soil boxes set at 5% slopes received 7.5 cm hr⁻¹ of simulated rainfall. Study soils included a Kenansville loamy sand (*loamy siliceous subactive thermic Arenic Hapludults*, a Coastal Plain soil) and a Davidson silt loam (*kaolinitic thermic Rhodic Kandiudults*, a Piedmont soil). Soil test P concentrations ranged from 16 to 283 mg P kg⁻¹. Sources of P included broiler litter (BRL), breeder manure (BRD), breeder manure treated with three rates of alum (Al₂(SO₄)₃) BRD0-0 kg m⁻², BRDL-3.9 kg m⁻², and BRDH-7.8 kg m⁻² and DAP along with an unamended control. All manure sources were applied at 66 kg P ha⁻¹. Water extractable P (WEP) represented an average of 10 ± 6% total P in manure. Runoff samples were taken over a 30-min period. Piedmont soil contained greater amounts of clay, Al and Fe concentrations, and higher P sorption capacities that produced

significantly lower DRP, TDP, AAP, and TP losses than the Coastal Plain soil. Runoff P loss did not differ for low and high STP soils of same taxonomy with the exception of AAP mass losses for Coastal Plain soil samples. Water extractable P in manures accounted for all DRP lost in runoff with DRP correlating strongly with WEP concentration (0.9961). A weak relationship between DRP in runoff and WEP applied to soil boxes was observed ($R^2=0.6547$) and increased when a possible outlying manure treatment, BRL, was omitted from regression data (0.9927). Overall, manures containing the highest WEP concentrations supplied the largest losses of DRP in runoff. Manure treated with 3.9 and 7.8 kg m⁻² of Al₂(SO₄)₃ (alum) decreased DRP in runoff by 29%. Values calculated for PLAT SPAF and NSPAF coefficients were higher for Coastal Plain soil than Piedmont soil and overall higher than default values in PLAT. Management based on these results should help minimize harmful effects of P in runoff.

**IMPACT OF MANURE AND SOIL TEST PHOSPHORUS ON PHOSPHORUS
RUNOFF FROM SOILS SUBJECTED TO SIMULATED RAINFALL**

by
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BIOGRAPHY

John Roberts was born in Newport, North Carolina, on December 3, 1978. After graduating from West Carteret High School in 1997, John attended North Carolina State University in Raleigh, North Carolina where he received a B. S. degree in Natural Resources with a concentration in Soil Science. Following a successful stint on the N. C. State Soil Judging team, John decided to further develop his soils knowledge by returning to N. C. State in August, 2002, in an attempt to attain a M. S. in Soil Science. John's research dealt with soil P and its interaction and influences on P in runoff from soils amended with differing P sources.

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Chapter 1

Review of Literature

Enactment of the Federal Water Pollution Control Act of 1972 and its amendment to the Clean Water Act in 1972, raised public and regulatory awareness for pollution of national waters. The act granted federal regulatory agencies, such as the U. S. Environmental Protection Agency (USEPA), the authority to set standards in regards to discharges into waters of the United States. Pollution was considered a point or non-point source; point from a known or identifiable location and non-point from a broad unidentifiable location. While regulatory enforcement of point source pollution is more attainable, non-point source pollution is confounding. In a report to Congress on the nation's water quality, USEPA identified runoff from 360 million hectares of agricultural land to be the leading cause of impairment in rivers and lakes and increased nutrients the greatest damaging factor in lakes, reservoirs and ponds in the US. State and jurisdictional participants noted decreased fish communities and recreational usage and increased algal blooms, aquatic weeds, and water treatment costs in sampled waters (USEPA 2000).

Phosphorus effects in surface waters

The prevalent effect of P enrichment to surface waters is eutrophication, a process where the addition of P available to algae rapidly increases algae growth and reproduction, forming thick algal mats. After P supplies are exhausted, the thick mats die, descend and are decomposed by benthic organisms. Dissolved oxygen is decreased in the water with the increase in aerobic respiration from the organisms decomposing the algal biomatter. The correlation between agricultural P lost to

surface waters and eutrophication has been well established (Cassell et. al. 1998, Correll 1998, Sharpley et. al. 1994). Schindler reported P to be the nutrient most often limiting for algae growth in freshwater systems (1977) which is the first requirement for eutrophication to occur. Since agricultural runoff is considered a major contributor of nutrients to surface water systems and since P is the most limiting nutrient in freshwater systems, factors influencing P pollution of surface waters have been meticulously researched in the last twenty years.

Phosphorus forms and transport to surface waters

Main transport mechanisms for the loss of P from agricultural lands are surface runoff and subsurface leaching to groundwater (Sharpley et. al., 1994; Eghball 2003). Amounts of P lost by these two mechanisms have been related to soil P, amount and intensity of rainfall, and time and rate of fertilizer and manure application (Edwards and Daniel, 1993; Heckrath et. al., 1995; Hooda et. al., 1999; Pote et. al., 1996). Current terminology for P forms in runoff most often examined includes a dissolved P (DP) form, a bioavailable P (BAP) form, and total P (TP). Dissolved P forms are referred to as either dissolved reactive P (DRP), molybdate reactive P (MRP), or soluble P (SP) depending on the researchers preference. The dissolved P form, which will be referred to as dissolved reactive P (DRP) henceforth, is defined as the amount of orthophosphate remaining in a sample after filtering through a 0.45 μm membrane. This form is immediately available for uptake by plants and algae (Rekolainen et. al., 1997) which explains why it is the P form most analyzed for in runoff. Another P form found in runoff from agricultural land is particulate P (PP). Particulate P refers to P attached to hydrous oxides and clay

particles as well as P bound to organic materials. When combined, DRP and portions of PP readily available for plant and algae uptake represents algal-available P (AAP) also known as bioavailable P (BAP).

Dissolved reactive P in runoff can be sorbed or desorbed if interacting with suspended soil particles occurs (Sharpley et. al., 1994), therefore having the potential to either increase or decrease P concentrations in runoff. Continuous application of animal manures increases P loading in surface soils which would have a tendency to decrease sorption of DRP in runoff due to a decrease in sorption sites of suspended solids. Broadcast applications of manures and litters to the soil surface decreases soil loss as PP by reducing rainfall impact and thus detachment of surface soil. However, with the application of manures, total PP losses could increase due to the loss of manure as PP. Pasture systems reduce PP additions in runoff and as a result DRP is the dominant P form in runoff from these systems (Sharpley et. al., 2001). Particulate P is more dominant in runoff from cultivated lands. However, with reduced tillage management practices PP can be decreased (Coolman and Hoyt, 1993; Myers and Waggoner, 1996).

Phosphorus leaching

Increases in P leaching, as with increases in runoff P, have been related to soil receiving long term fertilizer and manure applications (Sims et. al., 1998). In unmanured soils, P is typically sorbed by P deficient surface and subsurface horizons. Exceptions to this are sandy and organic soils with low P fixing capacities (Heckrath et. al. 1995). However, with repeated applications of P sources and the infiltration of soluble P during rainfall or irrigation events, P enriched surface soils

have decreased capacities to attenuate the P in solution. In turn, soluble P continues its downward movement until it contacts soil with available sites for P sorption. Soils have shown little attenuation of some forms of P, particularly dissolved organic P (DOP). Chardon et al. found that DOP comprised 96% of the TP leached from a 100 cm column of a sandy loam soil amended with swine slurry (1997). Sharpley and Moyer (2000) found similar trends from column leaching studies where manures, litters and composts had been utilized. For poultry manure, roughly 63% of TP analyzed in leachate was DOP. Both findings demonstrate a high potential for the leaching of DOP.

Increases in soil P levels from inorganic fertilizers and manures raise the potential for P loss in runoff (Sharpley et. al. 2001; Sims et. al. 1998). The North Carolina broiler industry ranked fourth nationally in 2002 producing 735 million broilers worth 1.5 billion dollars (NCDA 2002). This large amount of poultry production creates a tremendous amount of manure. Almost all manure is applied to agricultural crops as fertilizer. However manures have been applied to supply agronomic rates of nitrogen (N) for crops. Manures typically have N:P ratios of 3:1 or lower due to the loss of N as ammonia via volatilization during the period from excretion to field application (Powers and Van Horn, 2001; Van-Horn et. al., 1996). Agricultural crops such as winter wheat and corn have N:P uptake ratios of around 4.5:1 and 5.9:1 (Eghball, 2002; Gilbertson, et. al., 1979). Therefore, if manure is applied to meet N requirements of crops, P will be applied above crop removal rates and cause an increase in soil P.

Soil phosphorus fractions

As with many soil nutrients, P is in a continuous cycle in relation to the soil and is in either an inorganic or organic form. Inorganic and organic P sources contribute to the soil solution P. Organic forms include P found in humus and organic materials. Inorganic P refers to P additions from sources such as P fertilizers, naturally occurring inorganic P minerals, and mineralization of organic P.

Inorganic P is added to the soil via P containing fertilizers and animal manures but can also naturally occur in soils as rock phosphates such as apatite. However, rock phosphates are slow dissolving and contribute only a small amount of plant available P. Soil pH determines the orthophosphate form. In acidic soils with pH levels ranging from pH 4 to 7.2, the dominant orthophosphate species is H_2PO_4^- while the HPO_4^{2-} species dominates alkaline soils with pH levels of pH 7.2 to 11 (Sparks, 1995). Generally plants have a preference for the H_2PO_4^- form (Tisdale et al., 1985). After application of P fertilizers, inorganic P is precipitated and/or sorbed by soil constituents such as Al, Ca, and Fe hydroxides. Calcium phosphate precipitates under basic conditions while Al and Fe phosphates precipitate under acidic conditions.

Organic P is present in soils in the tissues and residues of plants, inositol phosphates, phospholipids, nucleic acids, nucleotides, and sugar phosphates (Tisdale et al. 1985). Through mineralization organic P is released in an available form to plants. Organic P can comprise 30 to 65 % of the TP in soil (Harrison, 1987) with inositol phosphates representing 50% of the total organic P fraction, while phospholipids and nucleic acids representing about 1% (Anderson, 1967).

Solution P is referred to as the amount of P available to plants in the soil solution and is very low when compared to total P in a soil. The amount of P in solution at a given time is based on dissolution/precipitation of P minerals, hydrolysis of organic matter and sorption/desorption of P to Al/Fe oxides and soil surfaces (Matar et. al., 1992; Bhatti et. al., 1998; Hinsinger, 2001). Solution P is in equilibrium with P sorbed by Al/Fe and clay minerals. When solution P is increased with the application of commercial fertilizers or animal manures, sorption takes place to maintain the equilibrium. The same happens when solution P is decreased by the uptake of plants in that P is released into solution from easily dissolved P sources otherwise called labile P sources. Non-labile P sources are more stable forms of P and release very little P to replenish amounts lost from the soil solution.

Microorganisms play an important role for biochemical reactions and the release of organic P in a plant available form (Stewart and Tiessen, 1987). Microbes immobilize P by combining orthophosphates, adenosine diphosphate (ADP) and an energy source to create adenosine triphosphate (ATP). Cellular P may also be mineralized if microbial cells are lysed and the internal P is released into the soil solution as inorganic P. Extracellularly released enzymes, termed phosphatases, catalyze reactions to mineralize inorganic P from organic sources (Bhatti et. al., 1998; Hinsinger, 2001). Three types of phosphatase enzymes have been identified as phosphomonoesterases, capable of hydrolyzing P from nucleotides or phospholipids; phosphodiesterases, capable of hydrolyzing P from nucleic acids; and phytases which hydrolyze inositol P (Sylvia et. al., 1999). The C:P ratio in a soil determines whether mineralization or immobilization of organic P can occur. Ratios

less than 200 signify conditions where mineralization occurs while immobilization occurs with ratios above 300 (Havlin et. al. 2005).

Plant available forms

The importance of P in plant life includes the production of nucleic acids and phospholipids, while controlling enzyme reactions for the regulation of metabolic pathways. (Theodorou and Paxton, 1993; Schachtman et. al., 1998). On average, P has been found to represent 0.2 % of plant dry matter. Due to slow diffusion rates of plant available P in the soil, P deficient plants increase root growth to increase surface area and extend roots to areas of higher P concentrations (Schachtman et. al. 1998). The form of plant available P in the soil solution is dependent on soil pH. The two previously discussed forms of inorganic P found in the soil solution, H_2PO_4^- and HPO_4^{2-} , are in equilibrium at pH 7.2 (Sparks, 1995). Below this value H_2PO_4^- is more dominant and vice versa. Since most North Carolina agricultural soils are in the range of 5.5 - 6.5, plant uptake of orthophosphate in the form of H_2PO_4^- is more common.

Change in soil phosphorus with manure applications

The continuous application of fertilizers and animal manures is the leading cause of P enrichment in agricultural surface soils (Sharpley et. al., 1995). With repeated applications of manures, soil P pools can change over time. Motavalli and Miles (2002) found significant increases in inorganic P, organic P, and labile P in the upper 20 cm of soils amended with dairy manure compared to native soils over a 111 year period. Sharpley (1999) applied poultry litter at a rate of 140 kg P ha^{-1} for a three year period to a fine sandy loam (Typic Paleudult) with an initial Mehlich 3 (M3)

P concentration of 10 mg P kg⁻¹ soil. Mehlich 3 concentrations of P in the upper 5 cm of soil after the three year period had increased to 235 mg P kg⁻¹ soil. An increase in organic matter has been hypothesized to increase P solubility.

Iyamuremye and Dick (1996) theorized three mechanisms for desorption of P in soils where organic matter is continually applied. First, organic molecules are in competition with P for sorption sites. Secondly, complexes with surface bound P to Al/Fe oxides will form soluble organic-metal compounds and thus release P originally sorbed. Lastly, an increase in the surface negative charge from organic matter sorbed to soil particles will decrease the electrostatic attraction of P to the soil.

Phosphorus sorption

Soils have a natural chemical affinity to bind P. Sorption occurs more commonly with Fe/Al oxides in non-calcareous soils and CaCO₃ in calcareous soils as well as on the edges of kaolinite clay minerals. Additions of P to alkaline, limed acid soils and even heavily fertilized acidic soils can produce calcium phosphates that are then precipitated out of solution (Lookman et al., 1996; Pierzynski et. al., 1990). Acidic soils, such as those found in North Carolina, most often contain Fe/Al oxides which fix plant available P (Barow 1985) and precipitate highly insoluble Al/Fe phosphates as secondary minerals (Lindsay, 1979; Pierzynski et.al., 1990).

McDowell and Condron (2001) reported Al/Fe oxides have the greatest impact on P sorption and desorption in alfisols of the UK while sorption of P by Al/Fe oxides in Bt horizons of an ultisol has been reported (Zhou et. al., 1997). Phosphorus can also sorb with complexes of organic matter, although amounts are much smaller when

compared to other P sorbing pathways. This binding scheme is important in soils where little Al/Fe oxides are still present due to extended periods of weathering and leaching. Manure applications over long periods reduce P sorption to these complexes because potential sorption sites are saturated with P. Phosphorus pollution potentials via erosion and leaching are consequences of P saturated soils.

Phosphorus capacity and degree of phosphorus saturation

A soils' capacity to fix P is important in reducing P loss in runoff and leaching. As with any capacity, soil and its inherent chemical constituents, such as Al/Fe oxides, have only a finite amount of sorption sites. Once these sites have been occupied with P, the capacity for further P sorption decreases and thus added P is either leached through the soil profile or can be lost in runoff during rainfall events. Recent studies have outlined chemical analyses that may indicate when a soil has the greatest potential for the loss of P and at what capacity this loss is most likely to occur. In general a soils' existing capacity to sorb P has been referred to as the degree of P saturation (DPS). Three chemical extractants that have been used to quantify DPS: ammonium oxalate (OX), Mehlich 3 and Mehlich 1 (M1) (Breeuwsma et. al., 1995; Hooda et. al., 2000; Maguire et. al., 2001b; Maguire and Sims, 2002; D'Angelo, 2002). In each case, DPS is quantified by the ratio of extractable mmol P kg⁻¹ soil to the summation of the extractable mmol Al kg⁻¹ and mmol Fe kg⁻¹ and multiplied by some coefficient and is described by the following equation:

$$DPS_{(ox,M1,M3)} = [P_{(ox,M1,M3)} / a(Al_{(ox,M1,M3)} + Fe_{(ox,M1,M3)})]$$

The a coefficient is used to estimate the proportion of Al and Fe available for fast and slow P sorption. Soil physical and chemical properties can alter values of a.

Values for a range between 0.5 and 1 and are most often between 0.5 and 0.68. Van der Zee and van Riemsdijk calculated $a=0.61$ for acid, sandy, low organic matter topsoils in the Netherlands (1988) while Paulter and Sims (2000) calculated $a=0.68$ using 41 soils sampled in Delaware. Somewhat lower values have been determined for Spodosols, $a=0.55$ (Nair and Graetz, 2002). A coefficient value of 0.5 has reportedly been used most often (Beauchemin and Simard, 1999; Lookman et. al., 1995; Paulter and Sims, 2000; and Schoumans, 2000).

The DPS equation can be used to transform DPS to DPS%. Quantified values of DPS% have been related to a soil's potential for loss of P via leaching and runoff. In a study performed by Nair et. al. (2004), samples of Alfisols, Entisols, and Ultisols used as dairy manure sprayfields were analyzed for the capabilities of $DPS_{OX}\%$, $DPS_{M1}\%$, and $DPS_{M3}\%$ to estimate the release of water soluble P (WSP). Values for DPS% were plotted as a function of WSP to determine a 'change point', the DPS% value above which a rapid increase in WSP occurred. For $DPS_{OX}\%$ and $DPS_{M1}\%$ change points, values were determined to be 20 while the $DPS_{M3}\%$ expressed a value of 16. Similar values for Netherlands soils were found by Breeuwsma et. al. (1995) in that soil above 25% DPS_{OX} was considered to contribute to P leaching to ground waters. Determining DPS% change point values could provide an assessment of potential P loss from a soil. Incorporating commonly used soil test P analyses such as Mehlich 1 and Mehlich 3 extracts could provide a more rapid analysis of problematic soils since these procedures are already used by many soil testing facilities.

Aluminum sulfate as a manure amendment

Since soils have high affinities for the sorption of P, long term application of fertilizer and manure P has greatly increased P pools in soils especially in fields adjacent to high volume animal feeding operations (Sims, 1993). High soil test P (STP) levels may result in regulations that cause abandonment of current receiving fields or the implementation of best management practices (BMP). One such BMP is amending manure before application to receiving fields with a P sorbing treatment such as aluminum sulfate (alum). Aluminum sulfate is used in treatment of wastewater as a flocculent in order to concentrate nutrients and metals (Mackie and Kilgow, 1995; Tse and Yu, 1997).

Alum was first used as a manure amendment in a study conducted by Moore and Miller (1994). The reasoning behind using alum is that although Fe and Ca have been shown to bind and precipitate P, these metals can dissolve under somewhat normal soil conditions. Therefore, several experiments have examined the P attenuating capabilities of alum. Smith et. al. (2001) determined that when swine manure was treated with alum at a rate of 430 mg Al L^{-1} , DRP was reduced 84% in runoff from manured soils. Moore et. al. (2000) reduced DRP in runoff by 73% at rates of $1816 \text{ kg alum} \cdot \text{poultry house}^{-1}$ from pastures where treated and untreated manure was applied over a three year period. Although exact binding mechanisms and the P species have not been identified, around 40% of the P in alum amended manure is complexed with Al while the rest is most likely organic phosphate species such as phytic acid (Hunger, 2004).

Alum has an equally important benefit to animal producers as it decreases animal mortality and provides a healthier working environment for employees via reduction of nitrogen (N) loss through volatilization (Carlisle, 1984). When added to manure, alum reduces the pH and as a result decreases ammonium volatilization. Moore et.al. (2000) found decreases in NH₃ levels of 99% in alum treated poultry houses as compared to untreated houses in the first four weeks with overall reduction of NH₃ to be 75%. Lefcourt (2001) reported that treatment of dairy manure with alum reduced NH₃ emissions by 50% in a wind tunnel system used to simulate dairy barns. The effectiveness of alum in reducing DRP in manures along with NH₃ volatilization has proved to be a valuable aid to animal producers.

Nutrient Management in North Carolina

Guidelines for the land application of animal manures are managed in general by the Natural Resource Conservation Service (NRCS) 590 standard (NRCS 590). Previously focusing on the application of manure sources to meet crop N requirements, the 590 standard states that confined animal feeding operations (CAFO's) must develop nutrient management plans at least every five years with the assistance of a trained nutrient management specialist and have it certified by a state regulatory agency such as the Department of Environmental and Natural Resources (DENR). Operations are obligated to manage nutrients according to the plan. Threshold numbers for animal feeding operations to qualify as CAFO's and therefore regulated are listed below.

Animal	CAFO qualifying population
Cattle	1,000
Swine	2,500
Turkey	55,000
Poultry	
Liquid manure system	30,000
Laying hens with dry litter system	82,000
All others	125,000

USEPA. 2002. Concentrated Animal Feeding Operation Clean Water Act Requirements. Information Series Pamphlet.

Increased awareness of detrimental effects of P to the environment caused NRCS to revise the 590 standard and shift its focus to P. The new 590 standard requires each state to develop a method to evaluate P loss from agricultural fields and base P application on one of three strategies. The first is to determine P application based solely on STP values, ceasing to recommend P additions above STP levels where no increase in yield would be attainable. The second is the application of P on an environmental threshold limit in which the threshold limit would be related to STP values and P concentrations in runoff. Once a STP level is reached corresponding to an increased P concentration in runoff over a critical limit (e.g. > 1 mg L⁻¹), no further P applications is be allowed. The third and final strategy is a P index system dependent on site specific variables including soil properties, climate, topography and manure type.

The latter method is used by the state of North Carolina due to its wide variety of soils, physiographical regions, farming systems, crops, and weather. An interagency committee consisting of scientists and engineers in the Departments of Soil Science and Biological and Agricultural Engineering at North Carolina State University including the participation from the N. C. Department of Agriculture, N. C.

Department of Environment and Natural Resources, and the USDA-NRCS, developed the Phosphorus Loss Assessment Tool (PLAT). Generally, PLAT evaluates P loss from agricultural fields by dividing potential pathways into four groups; particulate P, soluble P, subsurface P, and source P.

Particulate P loss at the edge of a field is defined as P attached to soil particles and is estimated from M3 STP, clay content, and Fe-P attenuation. Losses are increased with increases in erosion rates, determined by the Revised Universal Soil Loss Equation (RUSLE, 1995). Soils were generally categorized into 4 textural classes and paired with a soil P index threshold level: clays-500, loams-200, sands-100, and organics-50. Particulate P loss is weighted more for soils with indexes above threshold values. However, BMP's such as buffers, retention basins, water control structure and the presence of receiving slopes reduce PLAT particulate P loss.

Soluble P is derived from P released from the soil particle transported by surface runoff. Soluble P loss increases as STP and runoff volume increase. Artificial/natural drainage, cropping systems, residue management and rainfall can influence runoff volumes. Soil management practices that increase infiltration decrease runoff. This decreases estimates of soil P loss by PLAT.

The movement of subsurface P via leaching can vary drastically across the state and is dictated in PLAT by soil particle size and STP. Well drained, fine particle size class soils are correlated with high P retention capacities while wetter, coarser particle size soils tend to show lower capacities. Surface soils containing amounts of P higher than threshold values for the four different particle size class

soils must be sampled at 75 cm below the surface. Subsurface samples are determined to be above critical limits for organics, sands, loams and clays when M3-P levels are greater than 50, 100, 200, and 500 mg P kg⁻¹.

Source P is defined as P lost at the edge of field generated from fertilizers and manures. This is estimated by chemical properties in P sources including soluble P, non-soluble P, soluble P attenuating factors, non-soluble attenuating factors and runoff fraction. Application rate and method (i.e. incorporated, injected, broadcasted) also effects the PLAT source P losses. These variables are then multiplied to calculate P losses. Specifically, the soluble P attenuating factor (SPAF) can be defined as the amount of soluble P that will be delivered to the stream edge. SPAF is calculated using the following equation:

$$\text{SPAF} = [\text{Sol. P loss} / (\text{Sol. P fraction} \times \text{RF} \times \text{AF} \times \text{AR})]$$

where sol. P loss is the amount of soluble P collected in runoff, sol. P fraction is the fraction of P in manure that is water soluble, RF is the rainfall factor, AF is the application factor, and AR is the application rate of manure. Non-soluble P factor (NSPAF) is defined as the amount of non-soluble P that will be delivered to the stream edge. NSPAF values are calculated using the following equation:

$$\text{NSPAF} = [\text{Non-sol. P loss} / (\text{Non-sol. P fraction} \times \text{RF} \times \text{BF} \times \text{AF} \times \text{AR})]$$

where non-sol. P loss is the amount of non-soluble P collected in runoff, non-sol. P fraction is the fraction of P in manure that is not water soluble, RF is the rainfall factor, BF is the buffer attenuation factor, AF is the application factor, and AR is the application rate of manure.

A single numerical value is given to each of the four PLAT P loss pathways. The summation of these values determines the overall PLAT rating value for a particular field. PLAT ratings and their subsequent values are listed below.

PLAT Rating	PLAT Value
Low	0 – 25
Medium	25 – 50
High	50 – 100
Very High	> 100

If a field is evaluated and the PLAT rating is determined to be medium or low, a producer may continue to apply P sources on a crop N utilization basis. If a rating of high is calculated, then a producer may only apply P sources at levels equal to amount of P removed in the harvested crop. When a rating of very high is determined, no more P may be applied to that field with the exception of starter fertilizer.

Rationale for study

North Carolina is unique in respect to soils found in Mountain, Piedmont and Coastal Plain regions. Soils have been derived from igneous, metamorphic and sedimentary rocks giving rise to over 480 soil series and 16 separate soil systems (Daniels et. al., 1999). Of the top ten broiler producing counties within North Carolina half are located in the Piedmont and half in the Coastal Plain.

Characteristic soil surface textures for these piedmont areas include silt loams and loams containing finer fractions than the sands, loamy sands and sandy loams typical of leading broiler producing counties in the Coastal Plain. There is limited data and literature on the loss of P in runoff from North Carolina’s agricultural fields

and even more lacking is P runoff data for specific soil series and surface textures that receive most of the manure applied from animal feeding operations.

Simulated rainfall over field plots (FP) and over soil boxes (SB) are two methods used to measure P in runoff. Rainfall simulators adapted for simple set up and transportation on and to agricultural fields are utilized to supply a specific rainfall rate and duration. Field plot experiments allow assessment of effects of various cropping and tillage systems, manure types and manure application methods on P runoff with minimal degradation of in situ soil properties. Soil box experiments use fabricated boxes with specific dimensions to evaluate P runoff characteristics of soils with contrasting properties such as; STP values, textures and mineralogy, along with various manure types. This method is not useful for evaluation of crop/soil management practices on P runoff.

Research Objectives

1. Evaluate the impact of soil P status and the P composition of manure sources on P in runoff.
2. Characterize effects of manure sources on quantity and forms of P collected in runoff.
3. Enhance the PLAT database with respect to soluble P attenuating factor (SPAF) and non-soluble P attenuating factor (NSPAF) values.

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Chapter 2

Evaluation of soil effect and P source effect on P in runoff

Introduction

The prevalent effect of P enrichment on surface waters is eutrophication. The correlation between agricultural P lost to surface waters and eutrophication has been well established (Cassell et. al. 1998, Correll 1998, Sharpley et. al. 1994). Schindler reported P to be the nutrient most often limiting for algae growth in freshwater systems (1977) which is the first requirement for eutrophication to occur. Since agricultural runoff is considered a major contributor of nutrients to surface water systems and since P is the most limiting nutrient in freshwater systems, factors influencing P pollution of surface waters have been meticulously researched in the last twenty years.

Main transport mechanisms for the loss of P from agricultural lands are surface runoff and subsurface leaching to groundwater (Sharpley et. al., 1994; Eghball 2003). Amounts of P lost by these two mechanisms have been related to soil P, amount and intensity of rainfall, and time and rate of fertilizer and manure application (Edwards and Daniel, 1993a; Heckrath et. al., 1995; Hooda et. al., 1999; Pote et. al., 1996). Current terminology for P forms in runoff most often examined includes a dissolved P (DP) form, a bioavailable P (BAP) form, and total P (TP).

A soils capacity to fix P is important in reducing P loss in runoff and leaching. Acidic soils, such as those found in North Carolina, most often contain Fe/Al oxides which have been found to fix plant available P (Barow 1985) and precipitate highly insoluble Al/Fe phosphates as secondary minerals (Lindsay, 1979; Pierzynski et.al.,

1990). As with any capacity, soil and its inherent chemical constituents, such as Al/Fe oxides, have only a finite amount of sorption sites. Once these sites have been occupied with P, the capacity for further P sorption decreases and thus added P is either leached through the soil profile or lost in runoff during rainfall events. Manure applications over long periods reduce P sorption to these complexes because they have significant concentrations of P, are applied in large amounts, and potential sorption sites are saturated with P. Phosphorus pollution potentials via runoff from applied P sources, erosion of soil particles and leaching of P are consequences of P saturated soils.

Decreasing losses of soluble P from agricultural fields receiving manure would greatly benefit producers while reducing detrimental environmental effects. One method of achieving this reduction is to decrease the amount of soluble P in manures before application to fields. Aluminum sulfate or alum has shown to reduce amounts of soluble P up to 73% and 84% in manures (Moore, et. al., 2000 and Smith, et. al., 2001).

Increased awareness of effects of P in the environment caused NRCS to revise the 590 standard and shift its focus to P. The new 590 standard requires each state to develop a method to evaluate P loss from agricultural fields.

Since North Carolina has a wide variety of soils, physiographical regions, farming systems, crops, and weather, a P index system dependent on site specific variables including soil properties, climate, topography and manure type has been developed. An interagency committee consisting of scientists and engineers in the Departments of Soil Science and Biological and Agricultural Engineering at North

Carolina State University including the participation from the N. C. Department of Agriculture, N. C. Department of Environment and Natural Resources, and the USDA-NRCS, developed the Phosphorus Loss Assessment Tool (PLAT).

Generally, PLAT evaluates P loss from agricultural fields by dividing potential pathways into four components; particulate P, soluble P, subsurface P, and source P.

A single numerical value is given to each of the four PLAT rating components. The summation of these values determines the overall PLAT rating value for a particular field. If a field is evaluated and the PLAT rating is determined to be medium or low, a producer may continue to apply P sources on a crop N utilization basis. If a rating of high is calculated than a producer may only apply P sources at levels equal to amount of P removed in the harvested crop. In the case that a rating of very high is determined, no more P may be applied to that field.

Of the top ten broiler producing counties within North Carolina half are located in the Piedmont and half in the Coastal Plain (CP). Characteristic soil surface textures for these Piedmont areas include silt loams and loams containing finer fractions than the sands, loamy sands and sandy loams typical of leading broiler producing counties in the CP. A limited amount of data and literature is currently available on the loss of P in runoff from North Carolina's agricultural fields and even more lacking is P runoff data for specific soil types and surface textures that receive most of the manure applied from animal feeding operations.

Simulated rainfall over field plots (FP) and over soil boxes (SB) are two methods used to measure P in runoff. Soil boxes used in this study were fabricated

with specific dimensions and used to evaluate P runoff characteristics of various soils with contrasting STP values and of different types of manures.

Research objectives

1. Evaluation of the impact of soil P status and the P composition of manure sources on P in runoff by collecting and analyzing runoff from soils varying in soil P values and physiographical origins.
2. Characterization of effects of manure sources on mass loss of dissolved reactive P (DRP), total dissolved P (TDP), algal available P (AAP) and total P (TP) in runoff
3. Enhance the PLAT database with respect to soluble P attenuating factor (SPAF) and non-soluble P attenuating factor (NSPAF) values

Materials and Methods

Study soils

A Davidson loam, *fine kaolinitic thermic Rhodic Kandiudults*, was collected from a small grain farm under no-till management located in Orange County, NC (Piedmont region). The sampling depth was 0-4 cm. Soil with high (HPP) and low (LPP) soil test P levels (STP) were taken from fields planted in soybeans. The high STP sample was collected from a field which had previously received manure from a dairy operation but had received no manure in the last twenty years. The low STP sample was collected from a field 200 m from the high and is currently managed the same as the high STP field.

A Kenansville loamy sand, *loamy siliceous subactive thermic Arenic Hapludults*, was collected from a small grain and swine farm located in Johnston County, NC, (Coastal Plain region) under conventional tillage. The sampling depth was 0-10 cm. The high (HPCP) STP sample was collected from a soybean field currently used as a spray field for a swine operation. The farmer indicated the field received applications of poultry manure from early 1960's through the early 1980's. The low (LPCP) STP sample was collected from a hedgerow adjacent to the spray field which has never received any fertilizer or organic amendments.

After collection, soils were air dried in a greenhouse. The Piedmont soil was passed through a 2.54 cm stainless steel screen to cull rocks present in collected samples. All four soil samples were hand mixed using shovels and stored in separate containers. Difference in sampling depths was based on current management practices. Plow layers for the minimum tillage operation for the

Davidson soil were shallower than the Kenansville soil that is under conventional tillage.

Treatments

Seven treatments were used to illustrate effects of P forms and losses from varying P sources. Five manure P sources were collected from three poultry operations. Broiler breeder litter (BRD) was collected from an operation located in Johnston County, NC. Broiler litter (BRL) was sampled from an operation located in Duplin County, NC. Broiler breeder litter amended with three rates of aluminum sulfate (alum), 0 (BRD0), 3.9 (BRDL), and 7.8 kg m⁻² (BRDH), was sampled from Piedmont Research Station in Salisbury, NC. Manure P sources were applied at a 66 kg TP ha⁻¹ rate and corresponded to total nitrogen (TN) rates of 576, 96, 147, 182, and 180 kg TN ha⁻¹ for BRL, BRD, BRD0, BRDL, and BRDH manures, respectively. Unamended soil and diammonium phosphate (DAP) were utilized as controls. The application rate of TP and TN for the DAP treatment was 100 kg TP ha⁻¹ and 91 kg TN ha⁻¹. Manures were stored frozen until application immediately before rainfall simulations. Total storage time for frozen manures was 5 months.

Soil runoff boxes

Soil runoff boxes were constructed of wood (pine) and sealed with latex paint and a marine adhesive. Depth and sides of boxes measured 7.5 cm except the front which measured 5 cm. This allowed for 2.5 cm of freeboard to prevent any loss of applied materials due to splash from raindrop impact. Dimensions of the boxes were 20 cm wide x 100 cm long giving a total area of 0.2 m². Nine 5 mm holes were drilled in the bottom of all boxes to allow drainage of excess water after pre-wetting.

Air dried soils were tamped to desired bulk densities in soil boxes in five separate layers of known and equal weight to avoid differing densities throughout the 5 cm of soil. A layer of cheesecloth was placed in the bottom of boxes to insure drainage holes were not clogged with soil during runoff experiments. Any foreign objects, including bits of gravel or organic materials, found in the soils were discarded.

Rainfall Simulator

A variable rate rainfall simulator placed underneath a pole barn delivered a rate of 7.5 cm hr⁻¹ to four soil runoff boxes located within the simulator using a size ½-HH-SS-30WSQ TeeJet® nozzle at 8.5 psi. Water was supplied from a city water spigot and stored in a 350 gallon tank during and between simulations. Tarps encircling the simulator were secured and used to minimize any effect of wind. To achieve similar rainfall rates for each of the four boxes a uniformity test was preformed. Pre-weighed plastic cups were placed on a 20 cm x 20 cm grid within the simulator and rainfall was administered for 16.5 min and weighed again to determine the weight of water. A coefficient of uniformity (C_u) of 0.814, was then calculated using the following equation:

$$C_u = 1 - \frac{\text{absolute deviations from the mean}}{\text{mean weight of water}}$$

Cups located on the outer border of the grid were eliminated from the coefficient of uniformity calculation due to the possibility of overestimation of rainfall resulting from splash off the tarps. This information allowed delineation of four areas of similar rainfall rates for each of the four boxes.

Runoff Collection

Before each rainfall event soil boxes were tamped with desired soil and placed in their predetermined area within the simulator. Soil boxes were placed on prefabricated 5% slope stands and leveled to restrict runoff during pre-wetting. Common household air filters were used to cover soil boxes to reduce disruption of the soil surfaces. Rainfall was simulated until all nine drainage holes of each box were visibly draining. Soil moisture was allowed to equilibrate 24-28 hrs.

Treatments of P sources were applied evenly by hand to surfaces of soil boxes and immediately followed by rainfall. Rainfall was simulated until 30 min of runoff was collected. Runoff was collected at 5 min. intervals and a volume for each collection was determined by weight. After collection, samples were immediately transported to the laboratory where subsamples were taken for later analysis and stored according to the chemical attribute to be measured.

Dissolved Reactive P and Total Dissolved P

All runoff samples were analyzed for dissolved reactive P (DRP) and total dissolved P (TDP) (Pote and Daniel, 2000a; Pote and Daniel, 2000b). Immediately after collection, representative unfiltered samples were taken and pre-filtered through medium fine porosity filter paper to reduce amounts of larger particles in solution. Pre-filtered runoff samples were then passed through 0.45 μm filter and stored at -20°C in 20 mL scintillation vials to inhibit microbial activity. Dissolved reactive P in the filtrates was determined by reaction with ascorbic acid and molybdate (Murphy and Riley, 1962) and by colorimetric detection with an auto-analyzer. Analysis of TDP required an additional step whereby a portion of the DRP

sample was digested by increased temperature (160°C) and analyzing colorimetrically with an auto-analyzer.

Algal Available P

All runoff samples were analyzed for algal available P (AAP). Representative unfiltered samples were immediately taken in the laboratory and transferred into 50 mL centrifuge tubes. Samples were stored at -20°C to inhibit microbial activity until analyzed.

Unfiltered samples were thawed and (20 mL) transferred to 250 mL centrifuge tubes along with 180 mL of 0.11 M NaOH (Sharpley et al., 1991). Samples were placed on an orbital shaker (250 revolutions min⁻¹) for 17 hr, centrifuged @ 10 000 rpm for 10 min and passed through 0.45 µm filter. Phosphate in filtrates was determined by reaction with ascorbic acid and molybdate (Murphy and Riley, 1962) and detected colorimetrically with an auto-analyzer.

Total P Digestion

All runoff samples were analyzed for total P (TP) by an ammonium persulfate digestion method (Pote and Daniel, 2000b). Samples were stored at 4° C between time of collection and time of analysis. Approximately 50 g of unfiltered samples were weighed in duplicate in 100 mL beakers. One mL of 11 N H₂SO₄ was added to each beaker along with 0.4 g of NH₄S₂O₈. Beakers were placed on a hot plate at 160° C and gently boiled to achieve a 10 mL sample. Deionized water was added to samples until original weight was achieved. Phosphate in the filtrates was determined by reaction with ascorbic acid and molybdate (Murphy and Riley, 1962) and by colorimetric detection with an auto-analyzer.

Dilute Salt Extractable P

Study soils were analyzed for dilute salt extractable P (Self-Davis et al. 2000). A 1 g sample was weighed into a 50 mL centrifuge tube and 25 mL of 0.01 CaCl₂ was added to each sample to achieve a final soil to solution ratio of 1:25. Samples were placed on a reciprocating shaker (250 revolutions min⁻¹) for 1 hr and centrifuged at 10 000 rpm for 15 min. After centrifugation, solutions were filtered using medium fine porosity filter paper. Phosphate in the filtrates was determined by reaction with ascorbic acid and molybdate (Murphy and Riley, 1962) and by colorimetric detection with an auto-analyzer.

Degree of P Saturation

A chemical extraction was conducted to determine the degree of P saturation (Schoumans, 2000). Samples (2.5 g) of air dried soil were weighed in triplicate for each of the 4 soil samples and placed in a 250 mL shaker tubes. Fifty mL of an extraction solution consisting of 0.11 M ammonium oxalate monohydrate and 0.13 M oxalic acid dihydrate with a constant pH of 3.0 was pipetted into each shaker tube. Samples were allowed to shake at a rate of 180 revolutions min⁻¹ for 2 hr in a darkened room. Afterwards samples were filtered thru 0.45 µm filter and refrigerated at 4°C until analyzed for P, Al and Fe by inductively coupled plasma emission detection.

Phosphorus Sorption Isotherm Determination

Phosphorus sorption isotherms were determined for each study soil sample (Graetz and Nair, 2000). One gram of sieved air dried soil was weighed in duplicate into 50 mL centrifuge tubes. Phosphorus concentrations ranging from 0 to 60 mg L⁻¹

were produced using K_2HPO_4 in 0.01 M $CaCl_2$. Samples were placed on a reciprocating shaker (250 revolutions min^{-1}) for 24 hr and allowed to settle for 1 hr. Extracts were filtered through 0.45 μm filters. Phosphate in the filtrates was determined by reaction with ascorbic acid and molybdate (Murphy and Riley, 1962) and by colorimetric detection with an auto-analyzer.

Water Extractable P in Manures

Water extractable P (WEP) was determined for all manures. Concentrations were expressed on a dry weight basis (Kleinman et al., 2002a). Poultry manures were dried at 70° C in a forced air oven until a constant weight was obtained. Percent moisture was determined and 10 g of “fresh” manure was weighed into 250 mL centrifuge tubes. Deionized water was added to attain a manure to solution ratio of 10:200. Samples were placed on a reciprocating shaker (250 revolutions min^{-1}) for 2 hr and then centrifuged at 10 000 rpm for 15 min. Extracts were filtered through 0.45 μm membranes and stored at -20°C until analyzed. Phosphate in the filtrates was determined by reaction with ascorbic acid and molybdate (Murphy and Riley, 1962) and by colorimetric detection with an auto-analyzer.

Soil bulk density and soil moisture for runoff boxes

Oven dry bulk density and soil moisture of packed soil boxes were determined. Before application of P sources and immediate simulation of rainfall, PVC caps with known volumes were used to sample soils from rear locations of soil boxes. The sample of soil was weighed before and after oven drying at 105°C for 24 hr to determine soil moisture % and soil bulk density.

Particulate loss determination

Raw unfiltered runoff samples were utilized to determine loss of sediment or combination of sediment and P source. A 250 mL sample was transferred into a 500 mL tared beaker and dried at 65°C. After a constant weight was observed, the final weight of the beaker was subtracted from the original weight to determine particulate loss.

Data organization and statistical analysis

The experimental design is an incomplete block design with all seven treatments randomly assigned within the four areas within the simulator. Two rainfall events were required for one replication of each combination of soil sample STP and seven P source treatments. Three replications of each soil by sample STP by P treatment combination was achieved by repeated rainfall simulations over time.

Analysis using SAS GLM (SAS, 1998) procedures was used to compare loss of DRP, TDP, AAP, and TP in runoff (dependant variables) on a mass loss basis with soils from different locations, Coastal Plain (CP) and Piedmont (P), and soil test P values, High (H) and Low (L), and for seven P source treatments (unamended, diammoniumphosphate, broiler manure, breeder manure, breeder manure no alum, breeder manure low rate alum, and breeder manure high rate alum). Total mass losses for runoff P forms for first and second reps were calculated by summing mass losses from each 5 min sample to obtain a single 30 min sample value. Due to lack of available freezer space for samples, the final replication consisted of a single 30 min runoff sample. Mean separation using LSD analysis in SAS was used to

separate DRP, TDP, AAP, and TP mass losses in runoff for manure treatments averaged over all soil samples.

Initial ANOVA on non-transformed data for all soil by P source treatments showed a large error of variance. A second analysis was carried out excluding the unamended and DAP P source treatments because of the large error of variance associated with these treatments. Due to the reduction of error, the unamended and DAP treatments were excluded from the data set for statistical analysis.

Dissolved reactive P, TDP, AAP, and TP losses from unamended treatments represented $2.0 \pm 2.6\%$, $2.0 \pm 2.5\%$, $8.7 \pm 5.0\%$, and $3.4 \pm 2.3\%$, respectively, of the DRP, TDP, AAP, and TP loss from treatments where P was applied (calculated from data in Table 10). Losses for all runoff P forms were not corrected from runoff P forms from unamended soil treatments. Also, in manure treatments, P in runoff released from the soil is unknown since manure completely covered the surface and reduced rainfall interaction with the soil surface. Thus, amounts of runoff P derived from soil to which manure is applied is likely to be less than that which occurs in unamended soil treatment.

Results

Soil P status

Mehlich 3 (M 3) STP ranged from 26 mg P kg⁻¹ to 200 mg P kg⁻¹ in soil samples (Mehlich, 1984). Phosphorus index values for study soils were determined by the North Carolina Department of Agriculture Soil Analysis Laboratory and ranged from 31 to 226. Soluble P extracted with 0.01M CaCl₂ ranged from 2 to 20 mg P kg⁻¹ in CP soil samples and 2 to 17 mg P kg⁻¹ in Piedmont soil samples (Table1).

Table 1. Values for water soluble P (extracted with 0.01M CaCl₂), Mehlich 3 soil test P, and Mehlich 3 P index for study soils.

Soil	CaCl ₂ -P	M 3 STP	M 3 P Index
	mg P kg ⁻¹		
HPCP	20	186	226
LPCP	2	26	31
HPP	17	200	150
LPP	2	62	48

Degree of P saturation values (DPS%) for the soils ranged from 16 to 144% (Table 2). Oxalate extractable Al and Fe in Piedmont soil samples ranged from 35.1 to 46.5 mmol Al kg⁻¹ and 26.5 to 60.9 mmol Fe kg⁻¹. Oxalate extractable Al and Fe found in CP soil samples were determined to be 8.8 mmol Al kg⁻¹ and 3.9 mmol Fe kg⁻¹ (Table 2).

Maximum P sorption capacities were determined for study soils using Langmuir P isotherms (Figure 1). Maximum sorption capacities ranged from 256 mg P kg⁻¹ for the LPCP soil to 1960 mg P kg⁻¹ for HPP soil (Table 3). The HPCP soil was saturated with P beyond its P sorption capacity therefore desorption of P prevented development of a P sorption isotherm. Capacities were determined by the linearized equation:

$$[c/(x/m)^{-1}] = (1/bK) + (c/b) \dots \dots \dots \text{Equation 1}$$

where x/m represents the mass of P sorbed per mass of soil including previously sorbed P estimated from P_{ox}, b represents the estimate of the maximum amount of P that can be sorbed, K is a constant related to the binding energy interaction, and c is the concentration of P in solution at equilibrium with the soil surface. Plotting the combined left side term of equation 1 versus c will result in a straight line with a slope of (1/b) and an intercept of (1/bK). Linear regression was used to obtain values for b and K.

Manure Source P

Dry matter percentages varied between manures and ranged from 53% to 78% (Table 4). Differences were due to storage treatment prior to sampling of the manures. Manures with the lowest % dry matter, BRD0, BRDL and BRDH, were

Table 2. Values for degree of P saturation % and ammonium oxalate extractable P, Al, and Fe for study soils.

Soil	DPS%	Degree of P Saturation		
		P _{ox}	Al _{ox}	Fe _{ox}
		mmol kg ⁻¹		
HPCP	144	9.2	8.8	3.9
LPCP	16	1.0	8.8	3.9
HPP	68	32.9	35.1	60.9
LPP	27	10.0	46.5	26.5

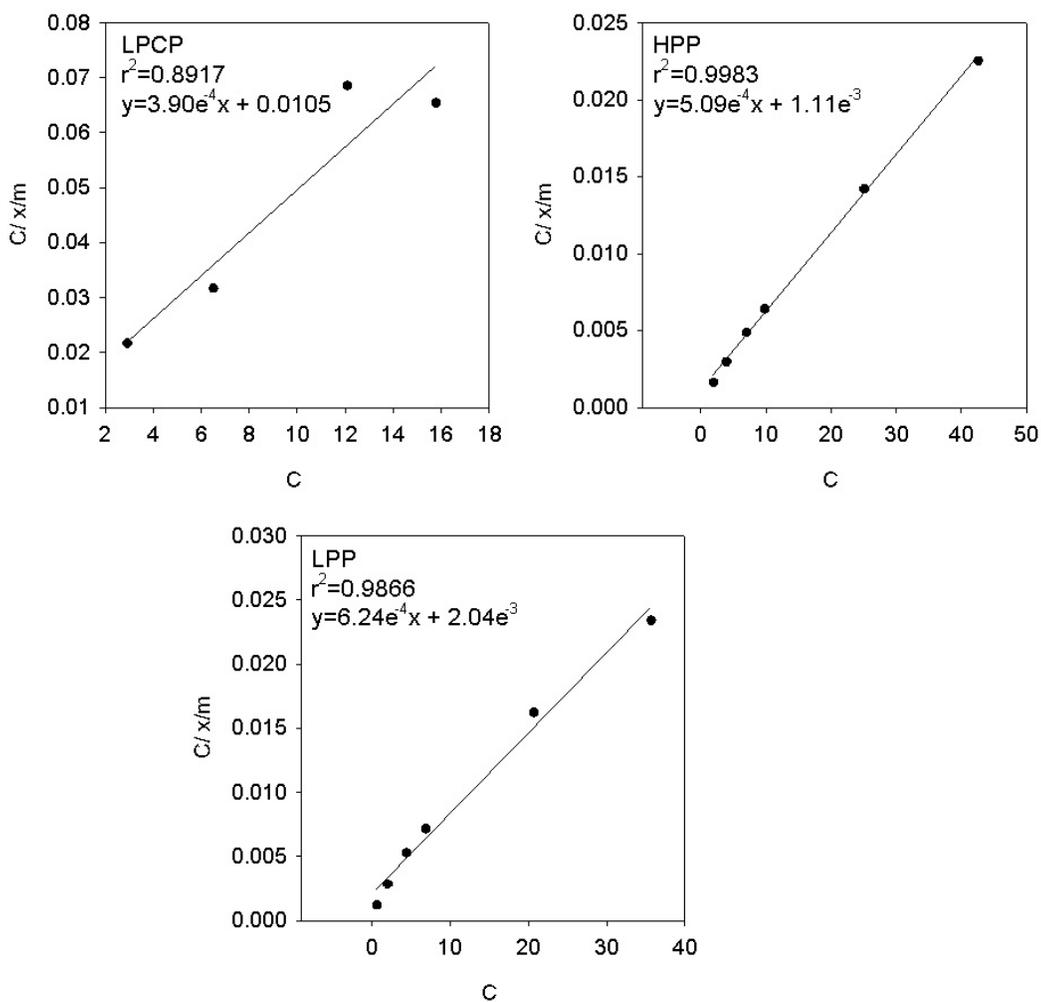


Figure 1. Linear regression of phosphorus isotherms for soil samples LPCP, HPP, and LPP where C is equal to the equilibrium concentration of phosphorus (mg P L⁻¹) and C/ x/m is equal to the equilibrium concentration of phosphorus over the amount of phosphorus sorbed per mass of soil (mg P kg⁻¹ soil).

Table 3. Maximum sorption capacities estimated using Langmuir P isotherms along with linear regression equations and correlation coefficients for study soils.

Soil	Langmuir parameters		r^2
	Maximum P sorption capacity mg P kg ⁻¹	Regression equation	
HPCP	---	---	---
LPCP	257	$y=3.89e^{-3}x + 0.0105$	0.8917
HPP	1960	$y=5.09e^{-4}x + 1.11e^{-3}$	0.9983
LPP	1600	$y=6.24e^{-4}x + 2.04e^{-3}$	0.9866

Table 4. Dry matter %, total N, total P, water extractable P, water extractable P as a percent of total P, and alum rates for manure P sources. Total P and water extractable P is expressed on dry basis.

Manure (Alum:mg m ⁻²)	Dry matter %	Total N	Total P mg kg ⁻¹	Water extractable P	Water extractable P %
BRL ¹	76	31900	5540	619	11
BRD ¹	78	16650	17300	668	4
BRD ⁰ ² (0.0)	64	37800	23200	3740	16
BRDL ² (3.9)	54	39200	20700	2220	11
BRDH ² (7.8)	60	39000	21700	2210	10

1. Stockpiled manure.

2. Sampled one day after chicken houses emptied

sampled one day after chicken houses were emptied while manures with highest % dry matter values, BRD and BRL, were sampled from stockpiled manure.

Manure sources used in this study were analyzed for TN, TP and WEP (Table 4). Total N in manures were determined by the North Carolina Department of Agriculture (NCDA) and ranged from 16650 to 39200 mg TN kg⁻¹. Phosphorus in manures ranged from 5540 to 23200 mg P kg⁻¹ for TP, as determined by the NCDA, and 619 to 3740 mg P kg⁻¹ for WEP. Percent WEP as a function of TP was determined for each manure P source with limits from 4 to 16% (Table 4). Water extractable P was reduced by 32 and 38% with low and high rates of alum when compared to the untreated manure (Table 4).

Runoff Parameters

Soil bulk densities, sampled before application of P sources and averaged for each soil, were 1.12 and 1.49 g cm⁻³ for Piedmont and CP soil (Table 5). Soil moisture values determined for samples taken just before application of P sources, were 19% for CP soil and 32% Piedmont soil. Piedmont soil retained more soil water than CP soil due to increased micropore volume versus the greater amount of macropore volume for CP soil which allows for little water retention.

Average total runoff for all soils and treatments was 7.0 L with a standard deviation of 0.9 L representing 84% of total rainfall applied. More specifically, total runoff volumes for CP and Piedmont soils ranged from 6.7 to 7.2 L, representing 83% and 85% of total rainfall applied (Table 6). There were no significant differences in runoff volumes within soil treatments or within P source treatments (Tables 7 - 8). Particulate loss ranged from 26 to 40 g m⁻² for unamended and DAP

Table 5. Bulk densities, D_b , and soil moisture % averaged across all samples for Coastal Plain and Piedmont soil.

Soil	Packed D_b g cm ⁻³	Standard deviation	Soil moisture %	Standard deviation
Coastal Plain	1.49	0.05	19	1
Piedmont	1.12	0.02	32	3

Table 6. Total runoff with standard deviations, total rainfall applied per box and runoff as a percent of total rainfall applied per box.

Soil	Total runoff L	Standard deviation	Total rainfall applied L	Runoff/applied %
HPCP	6.7	1.2	8.1	83
LPCP	6.8	1.2	8.1	84
HPP	7.2	0.9	8.5	85
LPP	7.2	1.0	8.5	85

Table 7. Mean total runoff for P source averaged over all soils. Runoff volumes followed by the same letter do not significantly differ at the 0.05 level.

P source	Mean total runoff L	N
Unamended	6.8a	12
DAP	6.9a	12
BRL	6.6a	12
BRD	7.0a	12
BRD0	7.2a	12
BRDL	7.2a	12
BRDH	7.1a	12

Table 8. Mean total runoff for soils averaged over all P sources. Runoff volumes followed by the same letter do not significantly differ at the 0.05 level.

Soil	Mean total runoff L	N
HPCP	6.7a	21
LPCP	6.8a	21
HPP	7.2a	21
LPP	7.3a	21

treatments. Particulate loss from manure P treatments significantly increased when compared to the unamended and DAP treatments (Table 9). Particulate losses from manure treatments ranged from 59 to 72 mg m⁻², which represented an average mass loss of 13% of total applied manure.

Phosphorus forms and losses in runoff

Dissolved reactive P represented 95% of TDP indicating almost all runoff P filtered through a 0.45 µm filter is the inorganic form and only a minute portion is in an organic form. Piedmont soil revealed lower losses of DRP and TDP from DAP treatments than CP soil, while supplying greater losses of DRP and TDP from unamended treatments than CP soil (Table 10). Study soils also showed decreased amounts of both DRP and TDP for unamended and DAP treatments for soil samples with low STP levels when compared to samples with high STP levels (Table 10). Means for treatments averaged over all soil samples ranged from 9.8 to 3940 mg P m⁻² for DRP and 11.0 to 3670 mg P m⁻² for TDP (Table 11). For both P forms BRD0 exhibited the greatest loss for manure P treatments in runoff with the lowest amount of P coming from unamended treatment. Over all DAP treatments had greatest amounts of losses of DRP and TDP.

Analysis of variance indicated significant main effects for soil type and manure treatment for both DRP and TDP (Table 12). This signifies that losses of DRP and TDP significantly differed between soils. Losses of DRP and TDP for Piedmont soil averaged over all manure treatments were 471 and 492 mg P m⁻², while DRP and TDP losses observed from the CP soil were 680 and 729 mg P m⁻² (Table 13). Significant differences in DRP and TDP losses for manure treatments

Table 9. Mean total particulate loss for P treatments averaged over all soil samples. Particulate losses followed by the same letter do not significantly differ at the 0.05 level.

P source	Particulate loss mg m ⁻²
Unamended	26a
DAP	40a
BRL	60b
BRD	71b
BRD0	61b
BRDL	72b
BRDH	59b

Table 10. Mean mass loss values for all P forms analyzed for in runoff for all soil sample and P treatments.

P source	HPCP				LPCP				HPP				LPP			
	DRP	TDP	AAP	TP												
	mg m ⁻²				mg m ⁻²				mg m ⁻²				mg m ⁻²			
Unamended	7.0	8.3	53.1	72.4	1.4	2.7	23.9	31.4	26.8	28.3	59.7	121	3.9	4.6	41.3	77.5
DAP	4810	4600	4560	9520	4500	3740	3800	6738	3570	3440	3430	7931	2890	2900	2950	7220
BRL	368	453	385	1450	493	563	544	1330	365	437	316	1390	338	425	333	1230
BRD	437	469	364	2510	416	479	376	2310	339	361	257	1150	292	317	257	1710
BRD0	1060	1090	1140	3380	992	1070	1070	4780	696	710	539	2350	723	764	515	2810
BRDL	702	724	945	4200	825	869	564	4960	432	445	418	2200	508	536	551	2040
BRDH	807	841	1110	3440	699	735	620	3160	486	502	396	2310	532	427	656	2100

Table 11. Mass loss values for runoff P forms for all P treatments averaged over all soil samples. Loss amounts within each separate runoff P form followed by matching letters were not significantly different at the 0.05 level.

P source	DRP	TDP	AAP	TP
	mg m^{-2}			
Unamended	9.8	11.0	44.5	75.7
DAP	3940	3670	3690	7850
BRL	391a	470a	394a	1350a
BRD	371a	406a	314b	1920a
BRD0	868b	909b	817c	3330b
BRD0	617c	644c	620d	3350b
BRDH	631c	626c	694cd	2750b

Table 12. Selected statistical output from SAS GLM software analysis for mass of dissolved reactive P, total dissolved P, algal available P, and total P collected in runoff over a 30 min collection period. Unamended and DAP treatments were excluded from data below.

DRP mass loss			
Source	Degrees of freedom	F value	Pr > F
Rep	2	0.59	0.5577
Soil location	1	24.71	<0.0001
Soil P level	1	0.09	0.7674
Soil location*Soil P level	1	0.00	0.9533
Rep*soil location*soil P level	6	0.49	0.8135
Treatment	4	18.82	<0.0001
Soil location*treatment	4	1.32	0.2832
Soil P level*treatment	4	0.40	0.8102
Soil location*soil P level*treatment	4	0.42	0.7940
TDP mass loss			
Source	Degrees of freedom	F value	Pr > F
Rep	2	0.69	0.5067
Soil location	1	29.66	<0.0001
Soil P level	1	0.12	0.7320
Soil location*Soil P level	1	0.08	0.7789
Rep*soil location*soil P level	6	0.31	0.9245
Treatment	4	16.06	<0.0001
Soil location*treatment	4	1.57	0.2054
Soil P level*treatment	4	0.63	0.6434
Soil location*soil P level*treatment	4	0.16	0.9563
AAP mass loss			
Source	Degrees of freedom	F value	Pr > F
Rep	2	3.16	0.0559
Soil location	1	32.39	<0.0001
Soil P level	1	0.55	0.4656
Soil location*Soil P level	1	5.13	0.0304
Rep*soil location*soil P level	6	0.72	0.6390
Treatment	4	13.71	<0.0001
Soil location*treatment	4	2.72	0.0467
Soil P level*treatment	4	0.60	0.6624
Soil location*soil P level*treatment	4	2.84	0.0400
TP mass loss			
Source	Degrees of freedom	F value	Pr > F
Rep	2	0.83	0.4446
Soil location	1	42.79	<0.0001
Soil P level	1	1.18	0.2856
Soil location*Soil P level	1	0.33	0.5719
Rep*soil location*soil P level	6	0.72	0.6361
Treatment	4	17.87	<0.0001
Soil location*treatment	4	4.25	0.0072
Soil P level*treatment	4	1.23	0.3179
Soil location*soil P level*treatment	4	0.74	0.5692

Table 13. Mass loss values for all runoff P forms for soil location averaged by soil test P level and over all manure treatments and excluding unamended and DAP treatments.

Soil Location	DRP	TDP	AAP	TP
	mg m^{-2}			
Piedmont	471	492	424	3151
Coastal Plain (CP)	680	729	711	1928

averaged over all soil samples were also observed (Table 11). Manure treatment BRD0 contributed the greatest loss of DRP and TDP, 868 and 909 mg P m⁻², with treatment BRD contributing the least loss, 371 and 406. Significant differences in DRP and TDP losses were not seen with samples of the same soil type differing in STP levels (Table 12).

Alum treated manures BRDL and BRDH exhibited a significant decrease in amounts of runoff DRP and TDP compared to unamended control manure (BRD0). Mean P loss for soil samples averaged over all treatments ranged from 755 to 1170 mg P m⁻² for DRP and 768 to 1170 mg P m⁻² for TDP (Table 10). Losses of DRP and TDP in runoff were greater from CP soil samples than for Piedmont soil samples.

Dissolved reactive P represented 94% of AAP indicating almost all runoff P is in a form immediately available for uptake by algae. Mean losses of AAP in runoff ranged from 23.9 to 4560 mg P m⁻² in Piedmont and CP soil samples (Table 10). Piedmont soil showed lower losses from DAP treatments than CP soil, while supplying greater losses of AAP from unamended treatments than CP soil (Table 10). Study soils also showed decreased amounts of AAP for unamended and DAP treatments for soil samples with low STP levels when compared to samples with high STP levels (Table 10).

Analysis of variance indicated significant main effects for soil type and manure treatment indicating that losses of AAP significantly differed between Piedmont and CP soils and between the five manure treatments (Table 12). Mass losses for Piedmont and CP soil averaged over all manure treatments were 424 and

711 mg P m⁻² (Table 13). Manure treatments BRD0 and BRDH contributed greatest AAP losses, 816 and 694 mg P m⁻², while the BRD manure treatment contributed the least amount AAP, 314 mg P m⁻² (Table 11). Treatment BRDL showed significant reduction of AAP when compared to the high and control rates of alum treated manures.

Significant interactions between soil type, soil P level, and manure treatment were also observed (Table 12). Interactions of soil type by soil P level originate from AAP losses averaged over all manure treatments being greater for HPCP than LPCP (Table 10). The soil type by treatment interaction was observed from greater AAP losses from manure treatments BRDL and BRDH from HPCP soil samples when compared to LPCP soil samples (Table 10). The interaction of soil type by soil P level by manure treatment can be seen in combination of the two interactions above.

Losses of TP ranged from 31.4 to 9520 mg P m⁻² in runoff from Piedmont and CP soil samples (Table 10). Dissolved reactive P, TDP, and AAP averaged 34% of TP in runoff samples. Piedmont soil averaged lower TP losses from DAP treatments than CP soil while supplying greater losses of TP from unamended soil treatments than CP soil (Table 10). Average total loss of DAP from Piedmont and CP soil represented 76% and 81% of total P applied. Study soils also showed decreased amounts of TP for unamended and DAP treatments for soil samples with low STP levels when compared to samples with high STP levels.

Analysis of variance indicated significant main effects for soil type and manure treatment indicating that losses of TP significantly differed between

Piedmont and CP soils and between the five manure treatments (Table 12). Mass losses for Piedmont and CP soil averaged over all manure treatments were 1928 and 3151 mg P m⁻² (Table 13). Fresh manure treatments, BRD0, BRDL and BRDH, contributed greatest TP losses, 3330, 3350, and 2750 mg P m⁻², and were not significantly different. Stock piled manure treatments, BRL and BRD, contributed the least amount TP, 1350 and 1919 mg P m⁻², and were also found not to differ significantly. The soil type by manure treatments effects on TP in runoff was significant (Table 12). This interaction resulted from large differences in TP in runoff from manure treatments BRD0, BRDL, and BRDH from Piedmont and CP soil samples (Table 10).

Discussion

Evaluation of soil effect on P in runoff

Soil test P values and DPS% were positively correlated with P extracted from study soils with 0.01M CaCl₂, $r^2=0.8619$ and $r^2=0.9296$, with the strongest correlation with M 3 P index, $r^2=0.9634$ (Figures 2-4). Due to limited numbers of soil samples and a clustering effect of data for soil samples, correlations found in this study were higher than those calculated for Mehlich 1 STP versus water soluble P ($r^2=0.71$) and DPS% versus water soluble P ($r^2=0.70$) determined from 122 Delaware soils and for ranges for STP versus water soluble P ($r^2=0.53$ to 0.88) for over fertilized European soils (Barberis et. al., 1996; Pautler and Sims, 2000). This relationship shows that an increase in STP, DPS%, or P index, would increase the amount of P that could be released in runoff during rainfall. These correlations also illustrate that with an increase in soil P, water soluble P also increases. Mass losses of DRP from

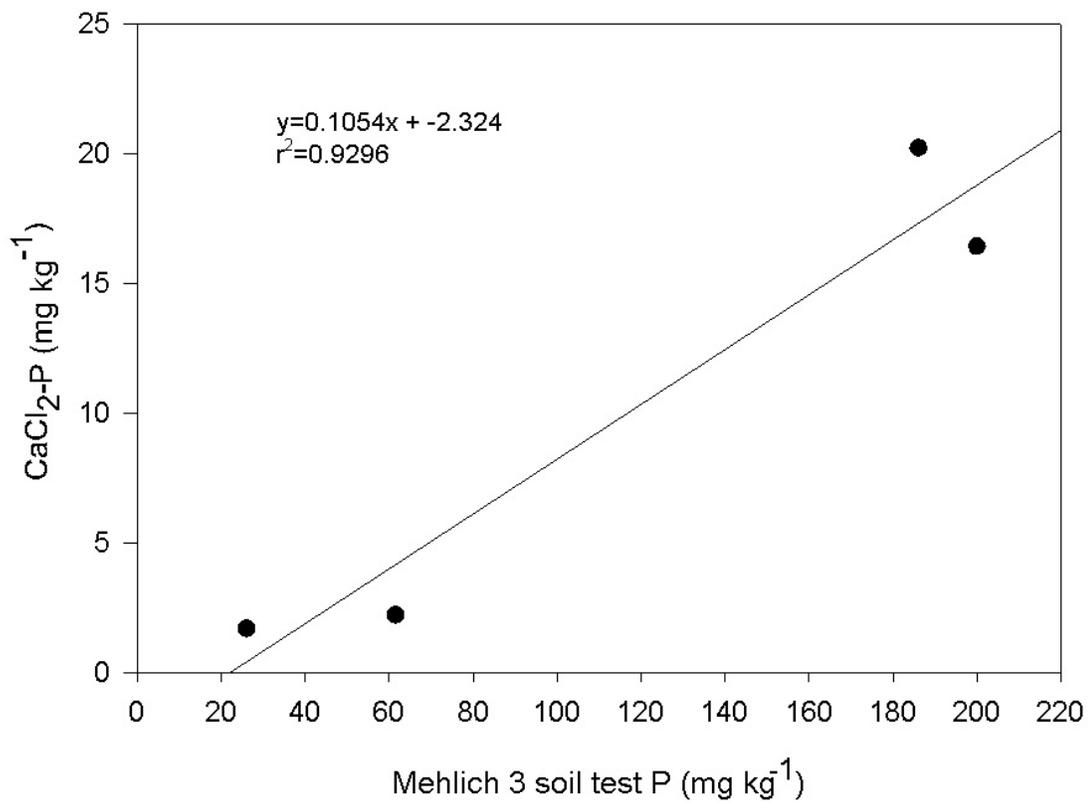


Figure 2. Relationship between water soluble P, extracted with 0.01M CaCl₂, and Mehlich 3 soil test P for Coastal Plain and Piedmont soil samples with high and low soil test P values.

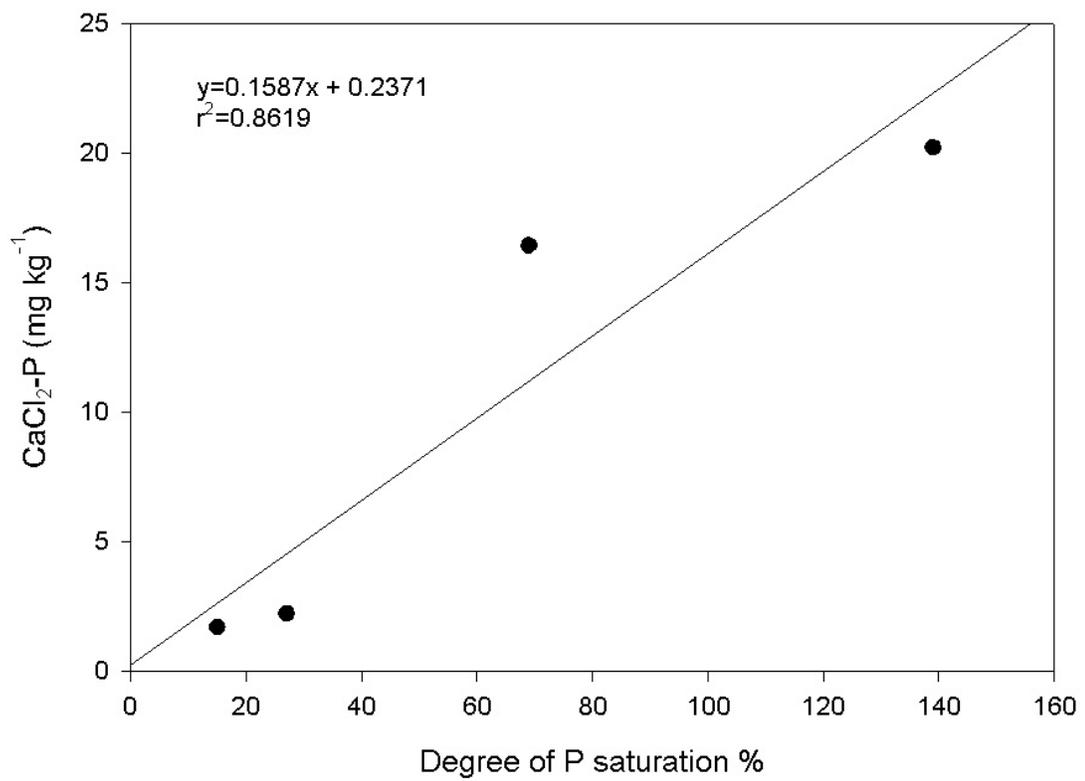


Figure 3. Relationship between water soluble P, extracted with 0.01M CaCl₂, and degree of P saturation % for Coastal Plain and Piedmont soil samples with high and low soil test P values.

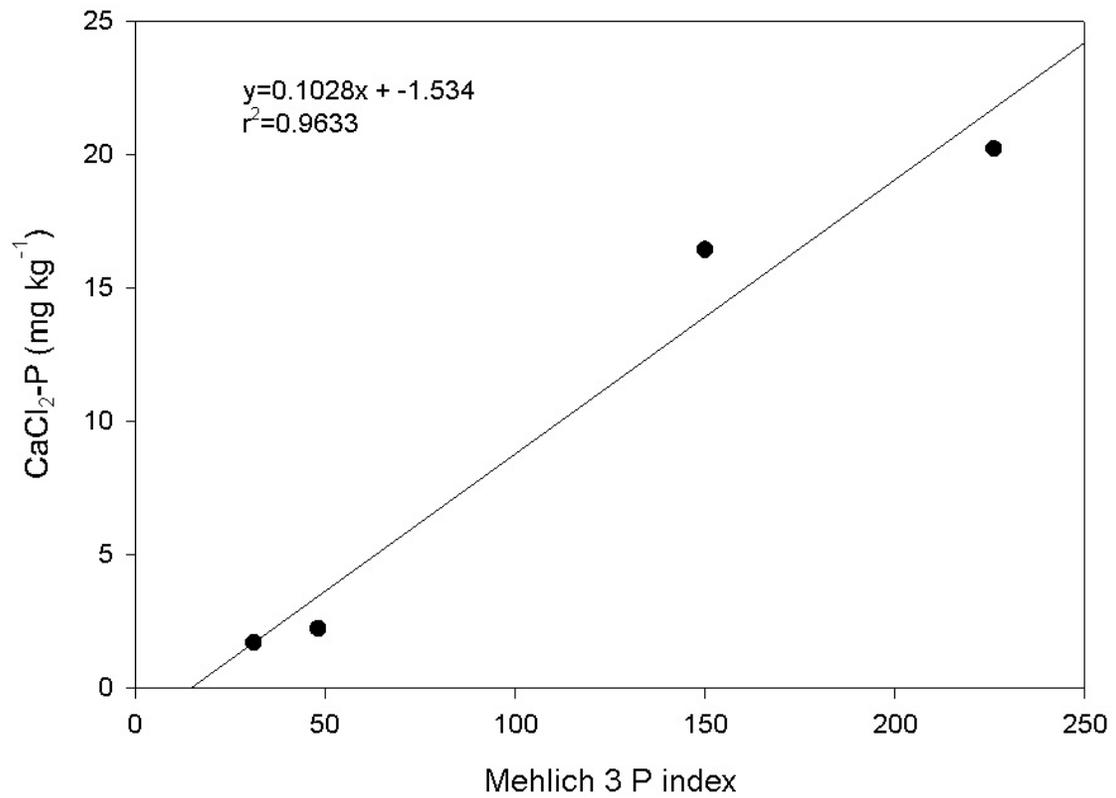


Figure 4. Relationship between water soluble P, extracted with 0.01M CaCl₂, and Mehlich 3 P index for Coastal Plain and Piedmont soil samples with high and low soil test P values.

unamended soil control treatments exemplify these relationships (Figure 5). High STP soils had almost 7 times higher DRP in runoff than their low STP soil counterparts demonstrating the greater potential for soil with high STP values to release P.

Increases in runoff P losses with increasing soil P have been reported in numerous studies (Sharpley, 1995; Pote et. al. 1999). Phosphorus losses from soils were averaged over all manure treatments to determine if STP had an effect on the amount of P lost in runoff. It was hypothesized that soils lower in STP would have a greater affinity to sorb released P from manure treatments than soils higher in STP, thus reducing P loading in runoff samples. In this study no significant main effect of STP level was observed for any runoff P form (Table 12). These results contrast with those reported by Kleinman et. al. (2002b) where P in runoff from a subset of three clayey soils with low (mean 17 mg P kg⁻¹ M 3-P) and high (mean 407 mg P kg⁻¹ M 3-P) STP concentrations treated with surface applied poultry, dairy, and swine manures was measured. In particular, DRP and TP losses in runoff from low STP samples treated with poultry manure were significantly lower than for high STP samples for two soil types. Runoff P from the third soil type did not differ significantly between low and high STP when rainfall was simulated 72 hr after application (Kleinman et. al., 2002b). In a study evaluating rainfall frequency with P in runoff from soils amended with poultry litter, Sharpley (1997) observed that an increase in time period between litter application and rainfall event resulted in decreased amounts of P in runoff which was attributed to greater P sorption to the soils with time (Sharpley, 1997). This could explain the lack of significant differences

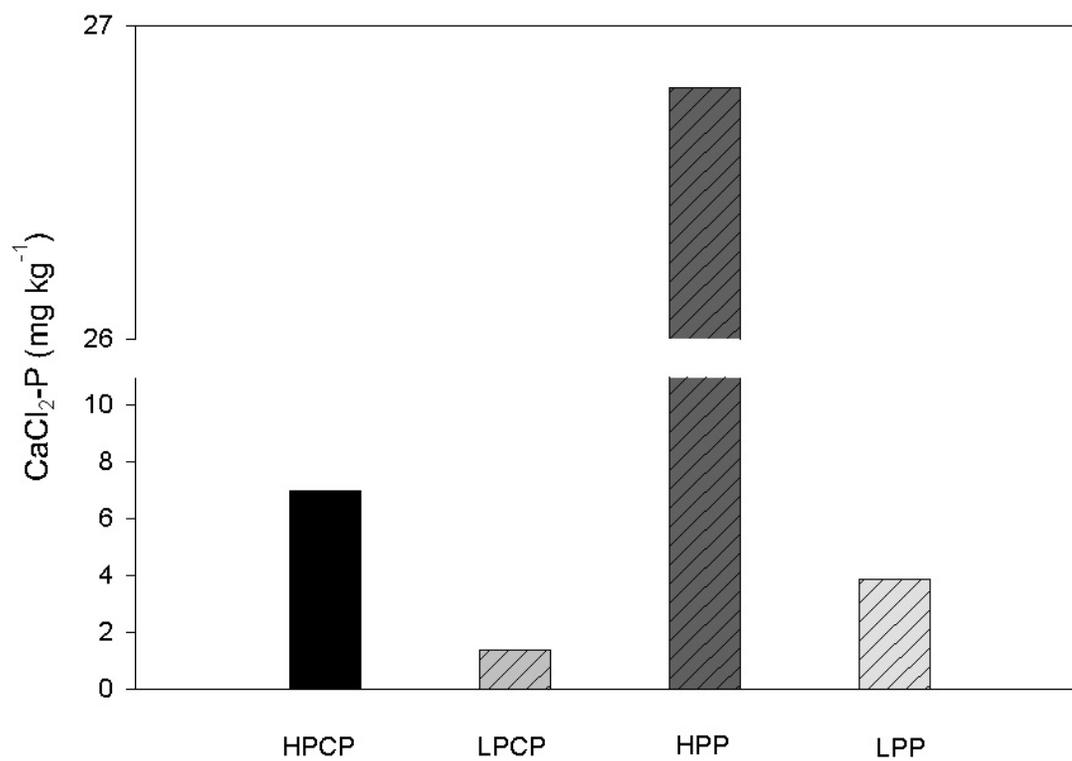


Figure 5. Mass losses of water soluble P for high P and low P Coastal Plain and Piedmont soils measured in runoff over a 30 min period.

in low and high STP soil samples from the CP and Piedmont in this study since rainfall was applied immediately after manure application, thus decreasing time for soil samples to interact with applied P sources.

Overall, significant differences for P losses were associated with soil types and their intrinsic properties such as clay % and P sorption capacities. Piedmont soil differed drastically from CP soil in maximum P sorption capacity estimated from Langmuir isotherms and oxalate extractable Al and Fe amounts. Piedmont soil samples averaged almost 5 times higher concentrations of oxalate extractable Al and Fe than CP soil and exhibited maximum P sorption capacities 7 times greater than LPCP (Table 2 and 3). Limiting amounts of critical P binding agents such as clay %, Al and Fe explains the difference in the P sorption capacity for the LPCP sample, as well as the Piedmont soil's ability to reduce P losses. Differences in P loss from Piedmont and CP soils were evaluated by relating mass losses of runoff P forms to M³ STP concentrations, clay %, P sorption capacity, remaining P sorption capacity and total Al and Fe concentration (Figure 6-10). Relationships for runoff P form versus STP concentrations were poorly correlated (Figure 6). Also, STP values give no bearing for the amount of P a soil can sorb and could only be used to relate P loss from soils with similar physical and chemical properties (i.e. clay %, P sorption capacities, Al and Fe concentrations). The strong correlation observed between runoff P form and % clay, P sorption, remaining P sorption, and Al and Fe concentrations indicate that P sorption and its inherent constituents are the major factors for lowering P loss in runoff. This was also observed in a study where a fine textured soil reduced DRP in runoff when compared to a much coarser textured soil

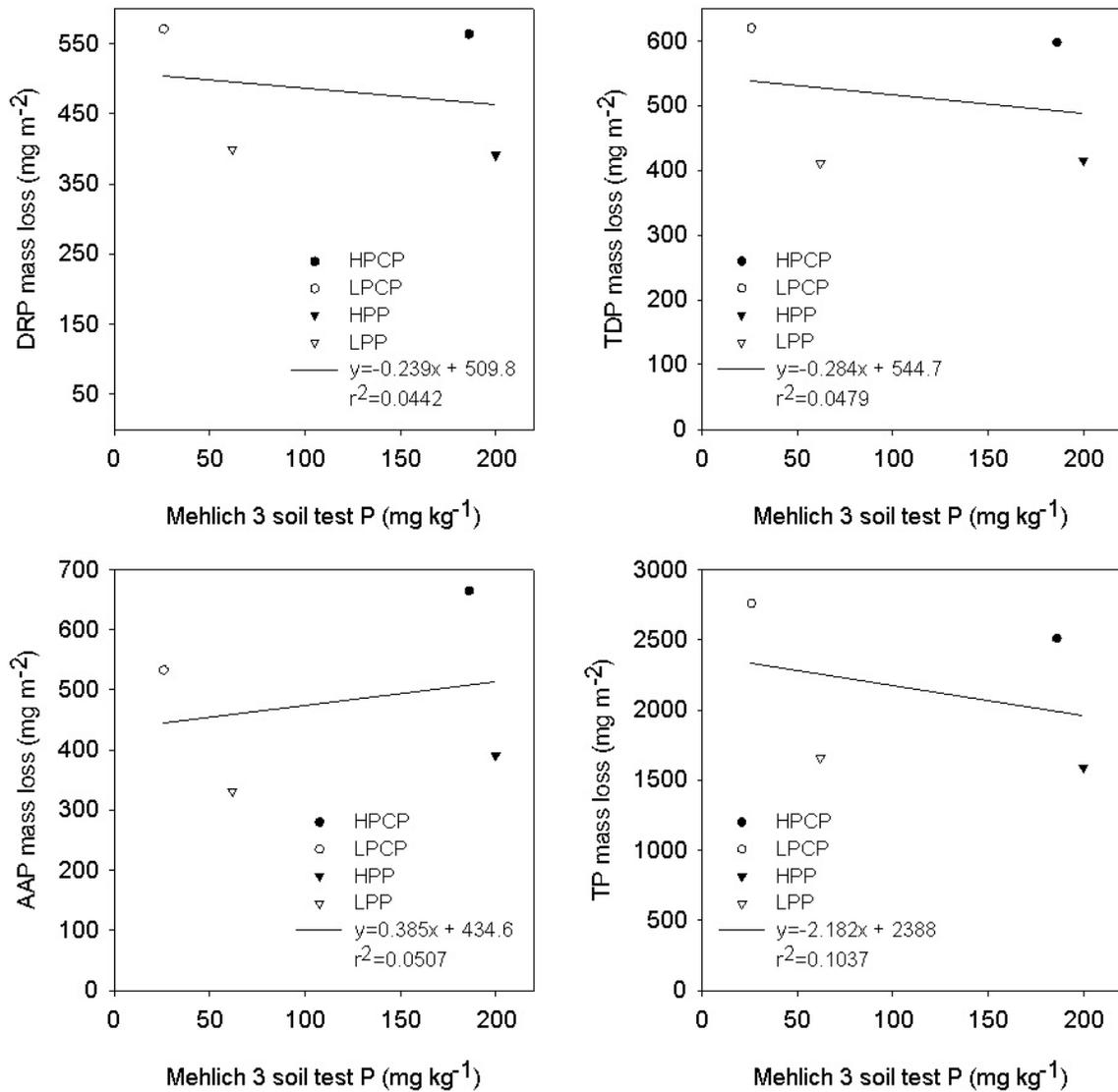


Figure 6. Relationship between dissolved reactive P, total dissolved reactive, algal available P, total P in runoff and Mehlich 3 soil test P for Coastal Plain and Piedmont soils excluding the P source treatment DAP.

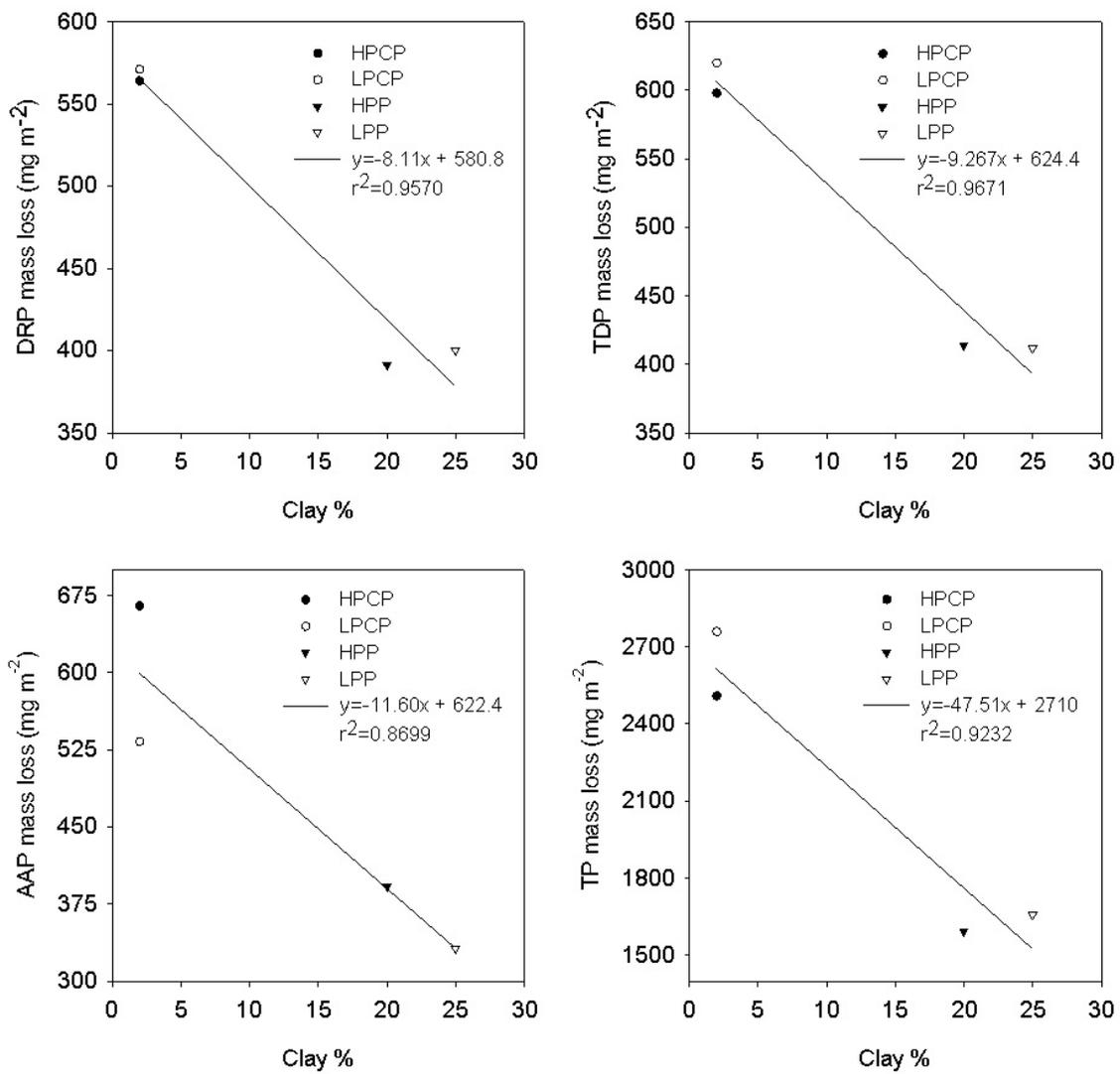


Figure 7. Relationship between dissolved reactive P, total dissolved P, algal available P, total P and clay % for Coastal Plain and Piedmont soils excluding the P source treatment DAP.

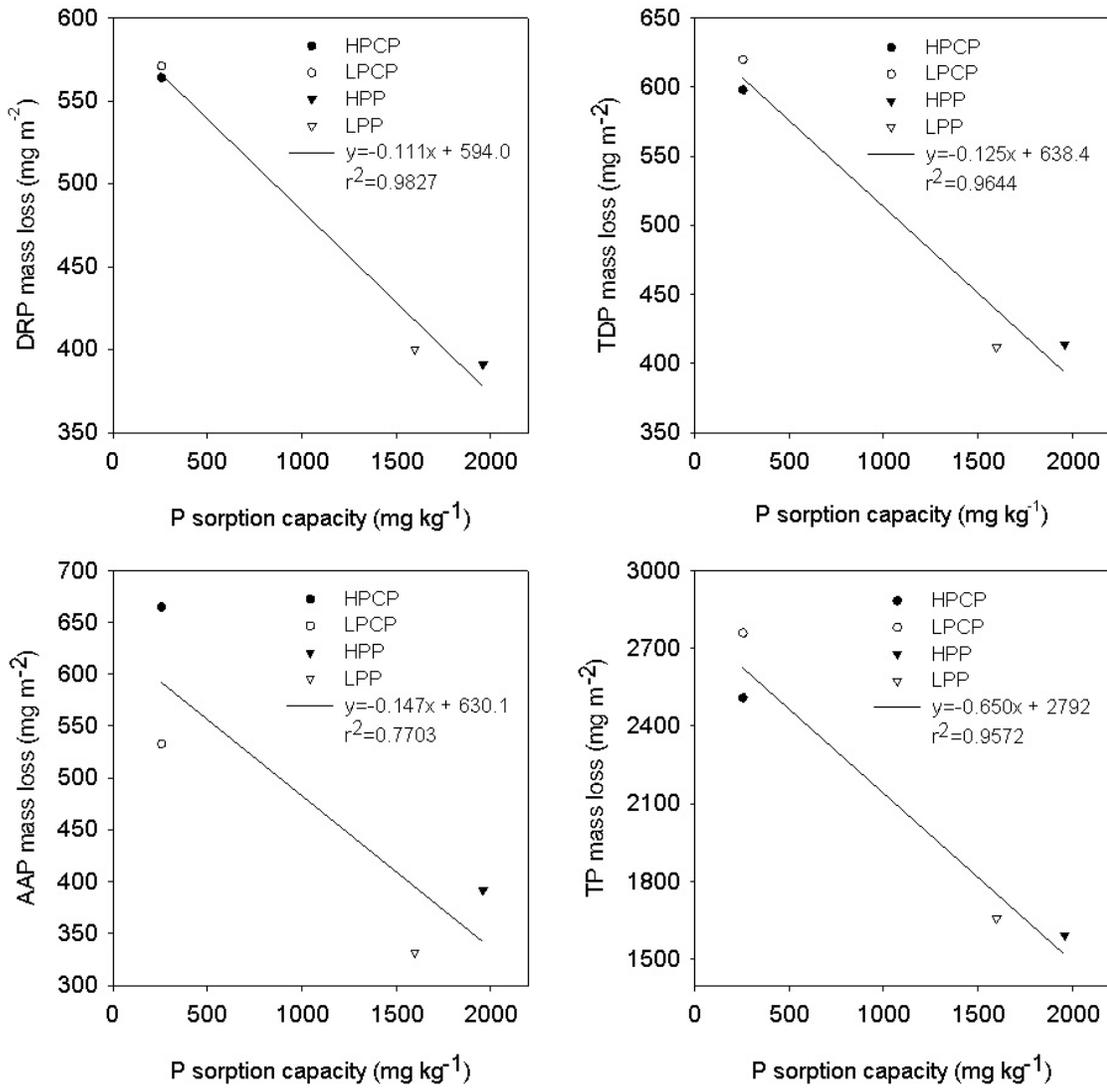


Figure 8. Relationship between dissolved reactive P, total dissolved P, algal available P, total P, and the P sorption capacities for Coastal Plain and Piedmont soils excluding P source treatment DAP.

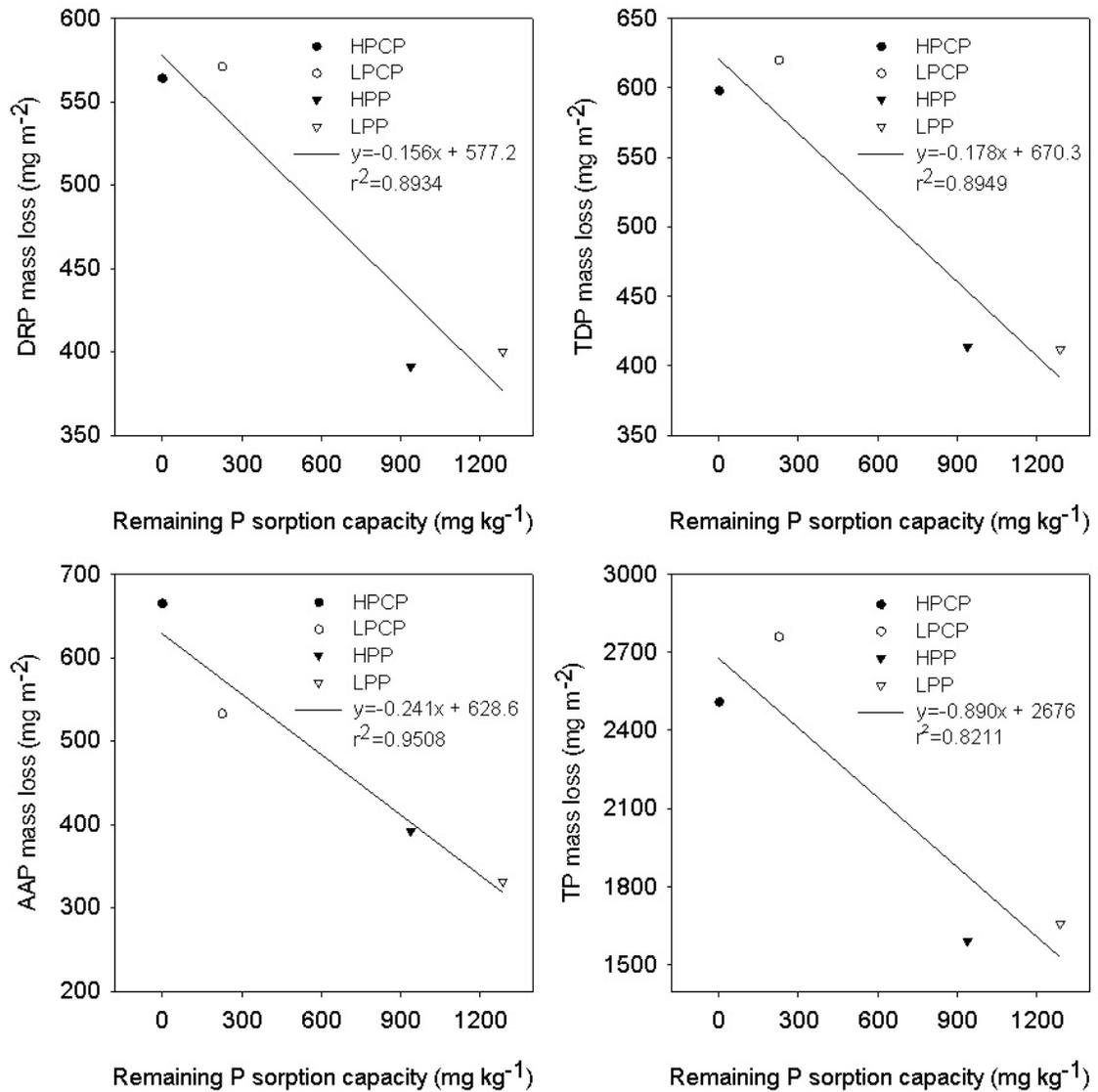


Figure 9. Relationship between dissolved reactive P, total dissolved P, algal available P, total P and the remaining P sorption capacities for Coastal Plain and Piedmont soils excluding the P source treatment DAP.

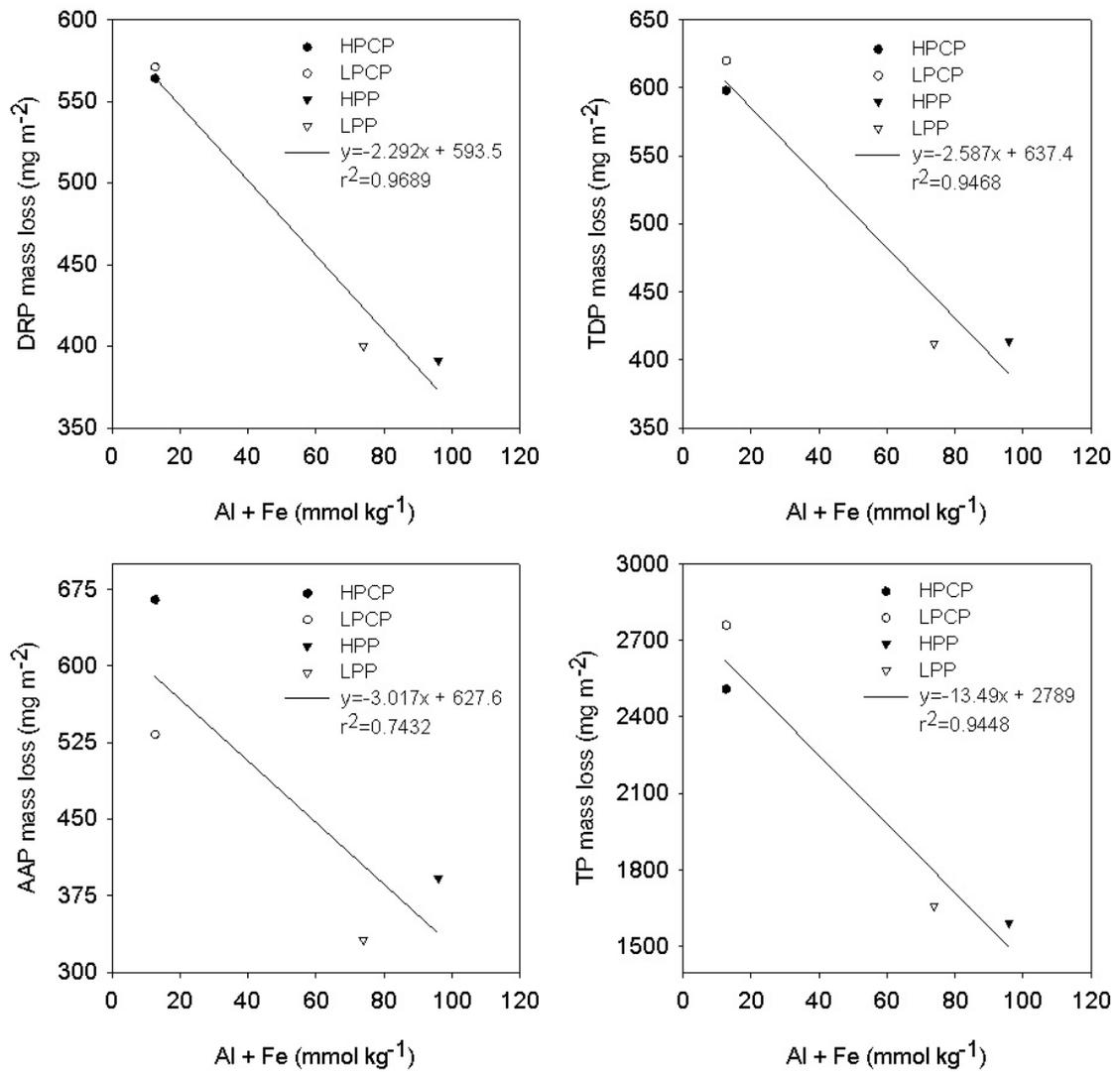


Figure 10. Relationship between dissolved reactive P, total dissolved P, algal available P, total P, and the summation of ammonium oxalate extractable Al and Fe concentrations for Coastal Plain and Piedmont soils excluding the P source treatment DAP.

after both were amended with equal amounts of biosolids and poultry litter (Penn and Sims, 2002). Differences in P sorption resulted from the increased amounts of oxalate extractable Fe and Al found and thus a greater P sorption capacity for soil with greater clay content.

Evaluation of P source effect on P in runoff

P losses in runoff were averaged over all soil samples to determine if P sources had an effect on P lost in runoff. A significant treatment effect ($p < 0.0001$) was observed for all P runoff forms, thus indicating that source P had a direct effect on amounts of P in runoff samples (Table 12). Manure treatments differing significantly in mean mass loss of P are illustrated in Table 11 for all P runoff forms. For DRP and TDP, treatment BRD0 released higher P runoff amounts than other manure sources. Manure treatments BRDL and BRDH, referring to low and high alum treatments, did not differ significantly while BRL had significantly higher amounts of DRP and TDP in runoff samples than BRD. Amounts of DRP were reduced by 29% by the low rate alum amended manure (BRDL) and further reduction was not seen by the high rate alum amended manure (BRDH) when compared to the unamended alum manure (BRD0). Manure P sources were further analyzed to determine specific characteristics likely to effect P amounts in runoff. Water extractable P was the manure attribute most closely associated with DRP and TDP losses from manure P sources. Dissolved reactive P losses from manure treatments in runoff comprised 52% (BRL, BRD0, BRDL, BRDH) and 98% (BRL) of the total WEP applied to each box (Figure 11), demonstrating that over half of the soluble P in manure can be lost in a single rainfall event.

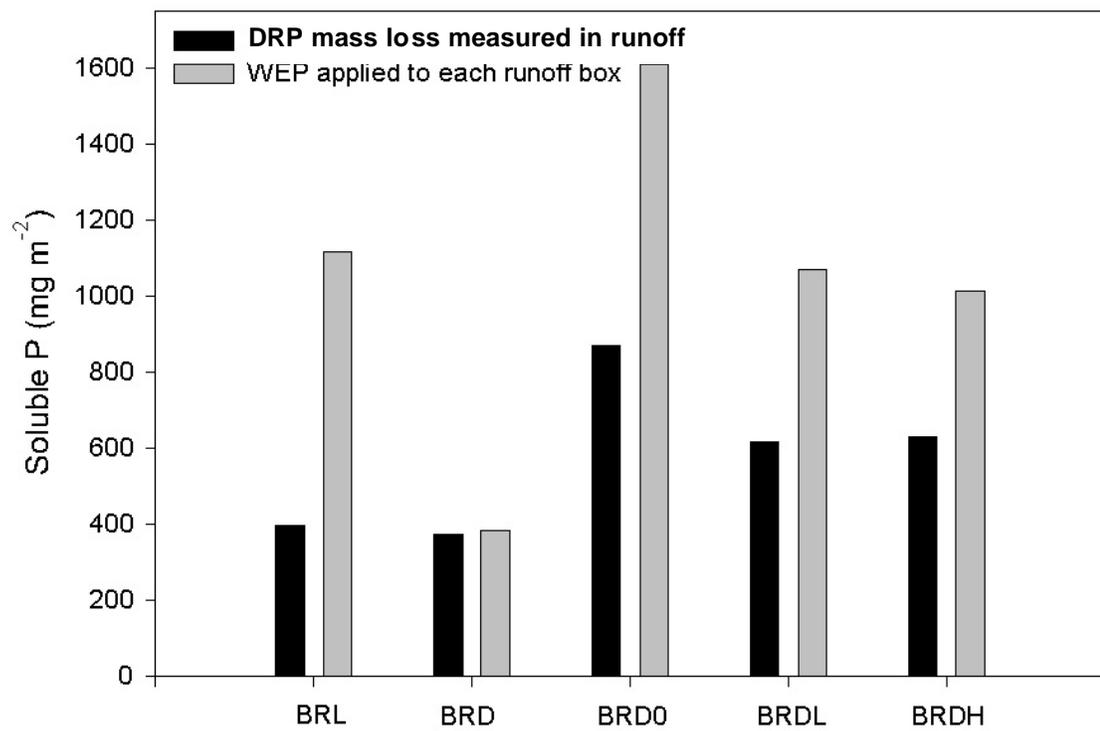


Figure 11. Mass loss of DRP averaged over all soils measured in runoff grouped with total WEP applied to each soil runoff box for each manure P source.

Losses of DRP and TDP were strongly correlated with WEP concentrations in manure P sources (Figure 12). Similar relationships were observed in this study with DRP, TDP and the rate of WEP applied per unit area. These relationships demonstrate weaker correlations than seen with WEP amounts in manures (Figure 13). However, a possible outlier for such data is identified as P source BRL. This particular manure has a much lower TP value than the other manure P sources. In order to achieve the 66 kg ha⁻¹ rate at which all manures were applied, 3 times as much BRL manure was applied to each soil box when compared to other manure applications. This resulted in greater application depth of BRL manure which would reduce interaction between rainfall and all manure material applied to soil boxes. Reduction in rainfall/manure interaction probably accounts for lower DRP and TDP losses in runoff samples from soils where BRL was applied. After omitting BRL manure treatments correlation coefficients for regression of DRP and TDP on WEP applied per unit area increased dramatically (Figure 14). Results for runoff P amount and concentration of WEP in manure correspond to those in a study by Kleinman et. al. (2002b) where WEP concentrations of dairy, poultry and swine manures were related to DRP in runoff when broadcast at the same rate ($r^2=0.86$). In a similar study Penn and Sims (2002) demonstrated a strong relationship between soluble P in biosolids and poultry litter, and DRP concentrations in runoff.

PLAT SPAF and NSPAF

Soluble P attenuating factor coefficients were somewhat higher for CP soils than Piedmont soils (Table 14). This is due to the greater sorption of runoff P by Piedmont soils. Overall, non-soluble P attenuating factor coefficients were greater

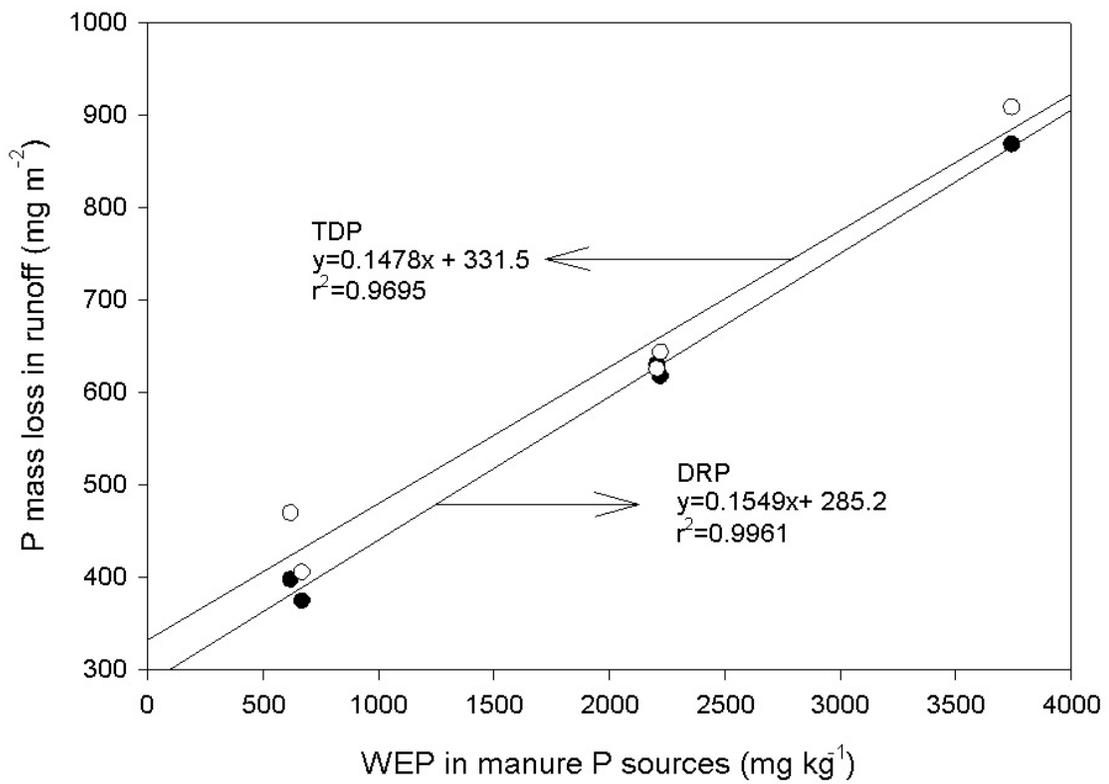


Figure 12. Dissolved reactive P and total dissolved P mass losses as a function of water extractable P in manure P sources BRL, BRD, BRD0, BRDL and BRDH.

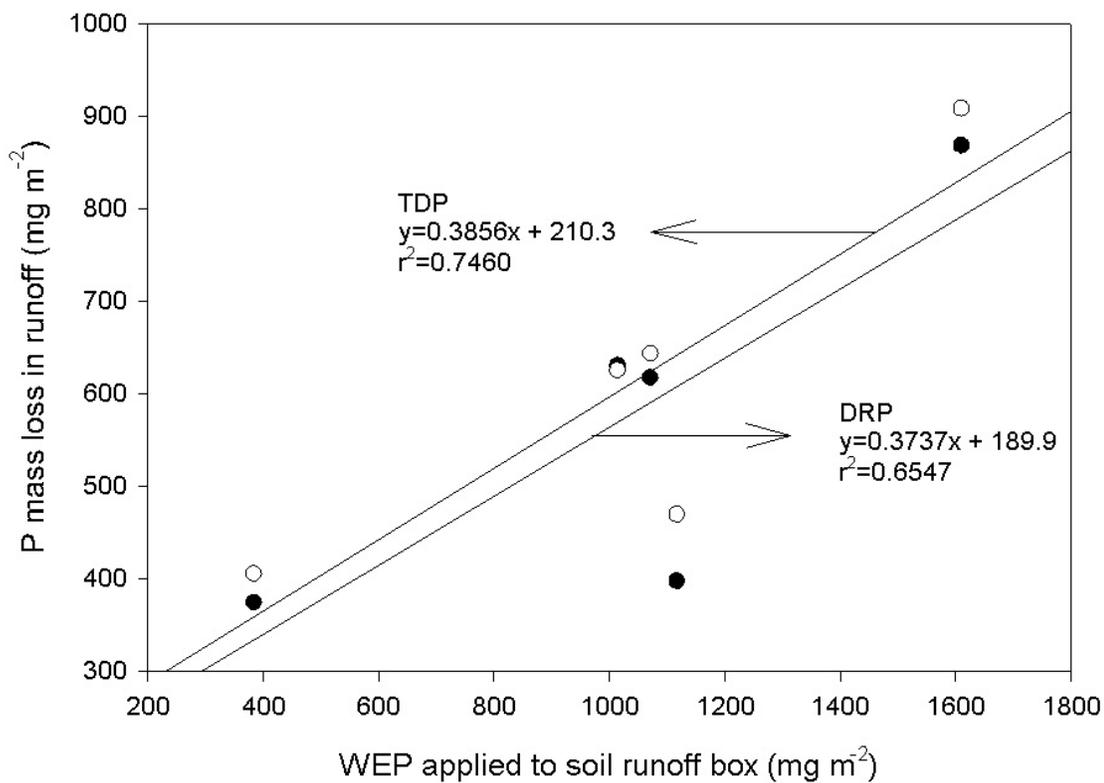


Figure 13. Dissolved reactive P and total dissolved P mass losses as a function of water extractable P applied per soil runoff box for P sources BRL, BRD, BRD0, BRDL and BRDH.

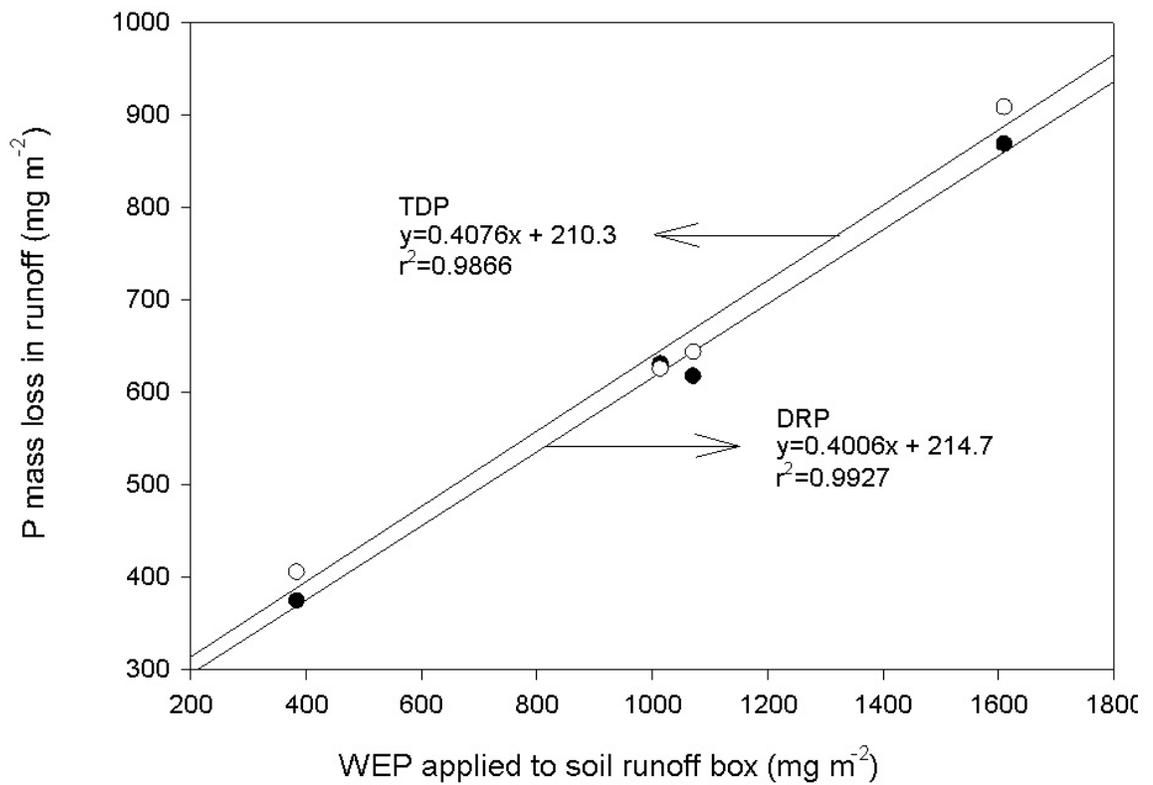


Figure 14. Dissolved reactive P and total dissolved P mass losses as a function of water extractable P applied per soil runoff box for P sources BRD, BRD0, BRDL, BRDH and excluding BRL.

Table 14. Soluble P attenuating and non-soluble attenuating factors calculated for Coastal Plain and Piedmont study soils.

Manure	SPAF			NSPAF		
	Coastal Plain	Piedmont	PLAT	Coastal Plain	Piedmont	PLAT
BRL	0.7	0.6	0.4	0.2	0.2	0.1
BRD	1.9	1.4	0.4	0.4	0.2	0.1
BRD0	1.2	0.8	0.4	0.7	0.4	0.1
BRDL	1.3	0.8	0.4	0.8	0.3	0.1
BRDH	1.4	0.9	0.4	0.5	0.3	0.1

for CP than Piedmont soils although similar numbers were seen for stockpiled manures (BRL and BRD). NSPAF coefficients for fresh manures were also higher than for stockpiled manures applied to CP and Piedmont soils. Values of SPAF for Piedmont and CP soils were on average 1.5 to 4 times greater than default values in PLAT and values determined for broiler litter in studies completed by Westerman et. al. (1983) and Edwards and Daniel (1993b). Values for NSPAF for Piedmont and Coastal Plain soils were determined to be 3 to 7 times higher than default values for NSPAF coefficients in PLAT and Westerman et. al. (1983). However, coefficients for Piedmont soils were similar to those determined in the study by Westerman et. al. (1983) in which coefficients of 0.2 were calculated.

Coefficients of SPAF and NSPAF for CP and Piedmont soil were on average greater than default values in PLAT. An explanation for this occurrence relates to the extreme runoff rates achieved in this study. PLAT estimates P loss on a lb P/acre/year, where total annual runoff from agricultural fields is estimated as 5 to 7.5 cm. In this study, runoff comprised on average 85% of total rainfall applied which corresponded to a runoff loss of approximately 3.2 cm. Also, the somewhat high soil moisture % for CP and Piedmont soils at time of runoff collection would have limited infiltration of rainfall which could have increased the amount of P forms in runoff, specifically water soluble inorganic P forms.

Conclusion

This study was used to test specific soil and manure characteristics and their impacts on P runoff under worse case scenario conditions. Parameters such as high total runoff/total rainfall applied ratios and soil moisture % increased P runoff

compared to natural runoff settings. Statistical analysis revealed P in runoff was not effected by low STP values when compared to samples with high STP levels, which was most likely due to immediate rainfall simulation after application of P treatments. However, despite the worse case scenario situation, soil type had a greater effect on P in runoff than manure treatments demonstrating soil properties (e.g. clay %, P sorption remaining P sorption capacity and Al and Fe concentrations) had a more significant impact on P in runoff than P sources. In order to maximize P sorbing potential in soils to reduce P in runoff, P sources should be incorporated to increase interaction between P sorbing properties of soils and the P source. Mehlich 3 STP concentration was poorly correlated with the amount of P in runoff emphasizing that when comparing a small number soil types with large differences in P sorbing characteristics, STP indexes have little value for predicting P in runoff.

Water extractable P in manures accounted for all DRP lost in runoff and was strongly correlated with WEP concentrations in manures. A similar trend was noted with DRP mass losses and the total amount of WEP applied to each box. Overall, manures contributing the larger losses of DRP in runoff also contained the largest concentrations of WEP. These relationships could be used to develop rankings of manures for estimated losses of P to nearby surface waters. If possible it may be advantageous for operators and regulators of animal feeding operations to analyze for WEP and include results in determining application rates of manures. Manures treated with aluminum sulfate (alum) had decreased runoff P when compared to untreated manure as a result of reduced WEP concentrations. Presently PLAT software does not account for reductions of P accomplished by treating manures

with P sorbing amendments. Default values for soluble P fractions of manures listed in the manure P characteristics table in PLAT should be decreased to account for reduction of soluble P in treated manures.

Soluble P attenuating factor and non-soluble attenuating factor coefficients were higher for CP soil than Piedmont soil. However, values for SPAF and NSPAF coefficients for both soils were greater than default values in PLAT.

This study demonstrated that soil box systems can effectively determine effects of soil properties and P source characteristics on P losses in runoff. Soil box systems can be evaluated in a relatively short amount of time and with less labor than larger field plot rainfall simulators. However, caution should be taken in extrapolating data from such small scale experiments. Amounts of P lost in soil box systems do not simulate the same losses found at the edge of fields. Soil runoff box systems can help determine relationships between soil properties and P sources.

In future soil box studies, researchers should pay close attention to parameters such as soil depth and soil moisture at the time of rainfall application in order to simulate runoff losses similar to field conditions. These parameters affect the infiltration rate and depth and therefore determine amount of total runoff and P amount therein. Shallow soil depths allow for less total infiltration while soil moisture % above field capacity reduces infiltration rates and produces runoff amounts that are unrealistically high when compared to the field situation.

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