PHOSPHORUS LOSS IN RUNOFF FROM GRASSLANDS RELATED TO SOIL TEST PHOSPHORUS AND POULTRY LITTER APPLICATION

By

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(Under the direction of David Radcliffe)

ABSTRACT

Phosphorus in runoff from agricultural fields to surface waters has been identified as an important contributor to eutrophication. The objective of this research were to determine the relationship between total P (TP), dissolved reactive P (DRP), and bioavailable P (BAP) in runoff from a benchmark soil (Cecil series) and Mehlich III, DI water, and Fe_2O_3 extractable soil P, and the effects of application rate and initial runoff timing on the long-term loss of P in runoff from surface applied poultry litter There was significant correlation between all forms of P in runoff and all soil P test methods used. The highest correlation between STP method and DRP in runoff occurred with water extractable soil P (R^2 = 0.68) Poultry (Gallus gallus domesticus) litter application rates of 2, 7, and 13 Mg ha^{-1} were used along with rainfall scenarios included; 1) sufficient rainfall to produce 30 min. of runoff immediately after litter application (R1); 2) no rainfall for 30 days after manure application, then sufficient rainfall to produce 30 min. of runoff (R2); and 3) small rainfall events every 7 d (5 min. at 75 mm hr^{-1}) for 30 days and then sufficient rainfall to produce 30 min. of runoff (R3). P loss was the greatest from the high application rate (13 Mg ha^{-1}) and from the immediate runoff (R1) plots. Non-linear regression procedures were used to develop equations relating to P loss over time to litter application. The resulting equations produced fairly accurate prediction of P concentration (0.68 to 0.91 R^2) under the conditions in this study. These equations were effective at predicting DRP concentration in runoff (r = 0.8). The prediction seemed to be most accurate for runoff events that occurred shortly after litter application. However, predictions became less accurate for events that occurred long after application due to fluctuations in observed DRP concentration. These fluctuations may be explained by changes in the pool of soluble P. Fluctuations in the soluble P pool way be related to variable source areas within the field and lysis of microbial cells caused by rapid wetting during large rainfall events.

INDEX WORDS: Phosphorus, Poultry litter, Runoff, Modeling, Simulated rainfall

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TABLE OF CONTENTS

CHAPTER 1

INTRODUCTION

Over the past decade, controlling non-point source pollution has come to the forefront in efforts to improve water quality in the United Stares and elsewhere. Non-point source pollution is typically divided into two categories, agricultural and urban. Agricultural non-point source pollution is one of the main contributors of nutrients to 50-60% of the eutrophied lakes and streams in the United States (USEPA, 1996). The principal components of agricultural non-point source pollution are sediment, bacteria, nitrogen (N), and phosphorus (P). Of these, P is the element most commonly associated with eutrophication in freshwater systems because these systems are usually P limited (Heckey and Kilham, 1988).

Traditionally, soil chemists considered P to be relatively immobile in soil because of its tendency to adsorb on the surface of Fe and Al oxides and hydroxides and to form insoluble compounds with these oxides and hydroxides over time. Because soils typically contain large amounts of Fe and Al, the concept of the soil as an “infinite sink” for P has been widely accepted. However, recent studies show that there is a limit to this “sink” and that, over time, P levels can build to the point where a soil’s capacity to “fix” P is exceeded. When this happens, P added in fertilizer or animal manure can be lost to the surrounding environment.

For economic reasons, it is unlikely that a soil’s sorption capacity will be exceeded when the P source is chemical fertilizer. However, in areas of concentrated livestock production, land application of large volumes of manure, with all its imported nutrients, is commonplace. Typically, animal manures have approximately equal amounts of available N and P contents, but crops generally require much more N than P. Application rates designed to meet a crop's N needs may, therefore, result in over-application of P.

The State of Georgia is one of the top poultry (*Gallus gallus domesticus*) producing regions of the United States with 1.2 billion broilers raised in 1998 (National Agricultural Statistics Service, 1999). Each of these broilers produces approximately 1.0 kg of litter (manure and bedding), each year resulting in an annual litter production of 1.2 million Mg. Most of this litter is concentrated in North and Central Georgia where intensive broiler production has become an important part of the region's economy.

Besides broiler production, agriculture in Central and North Georgia is generally limited to the production of beef cattle and its associated pastures and hayfields. Many farms in this region combine cattle and broiler production. Because broiler litter is readily available and inexpensive, it is commonly applied to pastures and hayfields as fertilizer. Soil testing data reveal that a significant number of the samples analyzed by the Georgia Cooperative Extension Soil Testing Lab have plant available P levels that are classified as very high (D.E.. Kissel, personal communication, 2000).

There is evidence that soils with high soil test P (STP) levels can contribute significant amounts of P to runoff waters either as dissolved reactive phosphate (DRP) or as particulate bound P (PP) (Pote et al., 1999; Sharpley, 1995). Because the relationship between STP and runoff P is soil and site specific, it is difficult to assign universally acceptable STP values above which P loss is unacceptable. Additionally, there are currently no federal or Georgia state standards for P levels in surface waters. However, the US EPA has proposed ambient water quality criteria for Ecoregion IV of 0.037 mg P L^{-1} for rivers and streams and 0.02 mg p L^{-1} for lakes

The relationship between STP and P in runoff is further complicated because manure applied to pastures and hay fields is not incorporated. Manure on the soil surface is a significant reservoir that can contribute P to runoff waters, but traditional soil test procedures may not measure this P (Pierson et al., 2001).

LITERATURE REVIEW

Phosphorus in Soils

Phosphorus is an important element in both plant and animal biology. It is essential to energy production through photosynthesis (ATP, ADP), genetic coding (DNA), and numerous structural elements such as the phospholipids that make up animal cell walls. Phosphorus levels in surface soils range from 100 to 3000 mg P kg^{-1}, but typical soil solution concentrations are < 0.01 mg P L^{-1} (Sharpley, 2000). Soil P exists in both organic (P_o) and inorganic (P_i) forms.

Inorganic P in soil generally occurs as either calcium phosphate, or iron and aluminum phosphate minerals. Lindsay and Moreno (1960) observed that the concentration of P in solution was controlled by soil pH and the solubility of the P minerals present. At the low pH levels commonly found in the weathered soils of the Southeastern United Sates, solution P concentration will be controlled by the solubility of Fe and Al minerals. However, Hsu (1965) concluded that, because P adsorbed on the surface of reactive Al and Fe oxides and hydroxides is more available than P in variscite or strengite, Al and Fe oxides and hydroxides are the soil components that control the concentration of P in soil solution.

Hsu (1965) further observed that P added to the soil system is subject to a two-step "fixation" process. The first step is a fast reaction where P is adsorbed to the surface of Fe and Al oxides and hydroxides in a matter of hours. The availability of this adsorbed P is related to the extent of surface saturation. When the oxide surface is nearly saturated, adsorbed P may be easily desorbed; in contrast, when the oxide surface is unsaturated, desorption may be very difficult. Taylor and Ellis (1978) suggest that P forms bidentate complexes at low concentrations of ortho-P ($H_2PO^-{}_4$), and monodentate complexes with $H_2PO^-{}_4$ at high concentrations. The second step in P fixation is a much slower reaction in which the adsorbed P becomes coated with Fe or Al oxides made available by the decomposition of soil minerals. The P is then "occluded" and is very resistant to dissolution and unavailable to plants (Hsu, 1965).

On average, organic P comprises about 50% of the P in soils, but can vary between 15 and 80%. Most of the organic P forms present in soils have not been characterized. The compounds known to make up soil organic P include inositol phosphates (10 to 50%), phospholipids (1 to 5%), and nucleic acids (0.2 to 2.5%). The remaining (approximately 50% of organic P) is likely to be in the form of stable esters that originate from microorganisms. Inositol phosphates are esters with one to six $H_2PO_4^-$ groups. Phytic acid, the hexaphosphate ester, is the most common form of inositol in soil systems and may make up 50% of soil organic P (Morgan, 1997). Although organic P may represent a significant pool of soil P, typically mineralization rates are low with only about 0.4 to 4.4% mineralized annually (Stewart and Sharpley, 1987). Because phytic acid makes up a large portion of soil organic P, the P mineralization rate is a function of phytase activity in the soil (Sharpley, 1985a). Through ^{31}P nuclear magnetic resonance spectroscopy, Rubaek et al. (1999) identified six categories of soil organic P: phosphonates, ortho-phosphates, monoester-P, teichoic acid-P, diester-P, and pyro-phosphate. Additionally, they found that most of the easily mineralized organic P was associated with clay-sized particles.

Phosphorus in Runoff

Forms of P in runoff can include DP (dissolved P) in both organic and inorganic forms as well as P associated with mineral or organic particles transported in the runoff. Because erosion rates from hayfields and pastures are low, DP is usually the dominant form of phosphorus in hayfield and pasture runoff (Sharpley et al., 1992). Dissolved P in runoff results from the interaction of rainwater with a thin layer of the soil's surface and any manure present at the surface. The effective depth of this interaction is dependent on soil aggregation, percent ground cover, slope, and rainfall intensity (Sharpley, 1985b). In two studies on effective depth of interaction, this depth ranged from 0.2 cm to more than 3.7 cm depending not only on the factors mentioned above but also on the experimental method used (Sharpley, 1981, 1985b). Regardless of the method, the soil layer that can supply P to runoff is very thin.

Numerous investigators have studied P runoff from agricultural fields under varying conditions related to manure or fertilizer application rate, storm interval, STP levels, and manure drying time. Application of animal manures can increase the concentration of P in runoff. In a study by Heathman et al. (1995), total P concentrations were 4.5 mg L^{-1} higher in runoff from bermudagrass plots receiving of poultry litter at a rate of 11 Mg ha^{-1} than from unfertilized control plots (Heathman et al., 1995). Studies have shown a positive relationship exists between the rate of manure application and the concentration TP, dissolved reactive P (DRP) and PP in runoff (McLeod and Hegg, 1984; Mueler et al., 1984; Edwards and Daniel, 1994; Vervoort et al., 1998; Wood et al., 1999).

The majority of the published research shows that most of the P lost to runoff is lost in the first runoff event from fields where manure has been surface applied (Edwards and Daniel, 1994; Sauer et al., 1999; Sharpley, 1997). However, the relationship between P loss in runoff from fields treated with animal manure, and storm interval and intensity is not clearly understood. Several researchers have shown after an initial spike, P levels in runoff remain above background levels for up to 18 months. Sharpley (1997) incubated 10 Oklahoman soils with poultry litter for 1 to 35 days prior to application of simulated rainfall, and found that although the P level in runoff declined through 10 consecutive rainfall events, it was still significantly above the P level in runoff from an unfertilized control plot. Similarly, in a study of surface applied poultry litter with and without tillage, Heathman et al. (1995) observed that both total P and soluble P concentrations in runoff were higher from plots receiving poultry litter than from an unfertilized control plot even after 10 rainfall events. In a multiple year study, Pierson et al., (2001) found that a single application of poultry litter to a small watershed in Georgia produced increased P in runoff for up to 18 months.

In contrast, some researchers report P levels in runoff returning to background levels after just a few rainfall events. For example, Sauer et al. (1999) report that soluble reactive P concentration in runoff from poultry litter amended plots dropped from 13.5 mg L^{-1} in the first runoff event to 1.2 mg L^{-1} in the second runoff event. Similarly, Edwards and Daniel (1994) found that levels of ortho-P and TP in runoff from plots receiving poultry litter did not significantly differ from control plots after two rainfall events.

These contradictory results may be due to differences in soil P sorption capacities and the fertility histories of the studied sites. If the results of these studies are examined more closely, it appears that there are differences in the levels of P coming off of the unfertilized controls. For example, soluble P in runoff from controls reported by Heathman et al. (1995) were an order of magnitude lower than those reported by Sauer et al. (1999). The plots used by both Sauer et al. (1999), and Edwards and Daniel (1994), had Mehlich III extractable P levels from 100 to 223 mg kg^{-1} (Sauer et al., 1999), whereas Sharpley (1997) and Heathman et al. (1995) reported Mehlich III extractable P levels of only 7 to 39 mg kg^{-1}. The higher STP levels may produce higher "background" levels of P in runoff, which in turn may make it impossible to detect the long-term effects of manure application.

Chemical extractants that simulate the availability of soil P have been used for decades to predict crop yield response to added P in fertilizer. These extractants are typically dilute acid, acetate, or bicarbonate solutions that remove Al or Ca from soil solution by precipitation of Al or Ca. As the activity of Al or Ca in solution decreases, native Al-P or Ca-P minerals dissolve to resupply solution Al or Ca, resulting in an increase in solution P, which gives a measure of the soil's ability to supply plant available P. According to Gartley and Sims (1994), the most common "routine" soil tests currently in use in the United States include Bray and Kurtz (0.025 *N* HCl + 0.03 *N* NH_4F), Mehlich I (0.025 *N* H_2SO_4 + 0.05*N* HCl), Mehlich III (0.02 *N* CH_3COOH + 0.025 *N*

NH_4NO_3 + 0.015 *N* NH_4F + 0.013 *N* HNO_3 + 0.01 *M* EDTA), Morgan (0.72 *N* NaOAc + 0.52 *N* CH_3COOH , pH 4.8), Modified Morgan (0.62 *N* NaOAc + 1.52 *N* CH_3COOH, pH 4.8), and Olsen-P (0.5 *N* $NaHCO_3$). The traditional interpretations of these tests as predictors of plant available P are based on extensive research, but there are less data to support interpretations of potential environmental impacts of soils that test high in P. Additionally, most soil samples submitted for agronomic testing are collected from 0 to 20 cm depth, but research has shown that as a result of long-tern manure application, P accumulates at the soil's surface. Phosphorus levels in the upper 2 cm of manure treated no-till fields may be three times higher than at 8 cm (Guertal et al., 1991). However, recent research supports the use of these "agronomic" tests for environmental purposes. Pote et al. (1999) report that P in runoff was highly correlated with STP in the upper 3 cm of pasture plots. Similarly, in a review of several P runoff studies, Sharpley et al. (1995) report that Bray-1 P extracted from the upper 0 to 5 cm of several soils accounted for 58 to 98% of the variation in DRP concentration in runoff. However, Sibbesen and Sharpley (1997) conclude that although STP and DP in runoff are related, variability in runoff volume and erosion because of climatic, topographic, and land management factors have a greater impact on actual P losses from agricultural lands than STP.

While the debate over environmental interpretations of traditionally agronomic soil tests continues, some researchers have developed alternative methods that may be more suited to environmental P assessments. One alternative for assessing P in soil is water extractable P. Because distilled water most closely resembles actual runoff, it may be that most appropriate soil extractant for predicting DP in runoff. Another alternative is to attempt to quantify the P in runoff that is directly available to aquatic plants, termed bioavailable P (BAP). Two methods of evaluating BAP have been described in the literature. One method uses 0.1 *M* NaOH to estimate the amount of P in runoff available to algae (Sharpley,1991). When NaOH is added to unfiltered runoff samples, some of the PP is solubilized. Phosphorus that is solubilized by 0.1 *M* NaOH, plus the DRP originally in the runoff constitutes bioavailable P. A second method to quantify BAP is to employ iron-oxide impregnated paper (Sharpley, 1993). The coated paper acts as a P sink that adsorbs both DP and some of the P that is adsorbed to soil particles. Iron oxide coated paper can also be used to estimate BAP in soil samples.

Because most P from unincorporated manure applications is lost in the first runoff event, several researchers investigated the effect of time between manure application and a runoff producing rainfall on P loss. Conceptually, the P in animal manure should become less susceptible to loss in surface runoff over time due to the adsorption and fixation process described previously. “Drying times” (the interval between application of manure and the first runoff event) of 1 hour to 3 days were studied by Westerman and Overcash (1980). They reported a 90 percent reduction in P concentration in runoff from fescue plots treated with liquid poultry manure after 3 days of drying. Similarly, soluble P concentrations in runoff from soils mixed with poultry litter were reduced from 0.74 to 0.45 mg L^{-1} when incubation time was increased from 1 to 35 days (Sharpley, 1997). Both of these studies conclude that because drying time reduces P losses, producers should plan manure applications at times when runoff producing rainfall events were less likely. It should be noted however, that these two studies did not address the issue of surface-applied dry manure because Westerman and Overcash spread liquid manure and Sharpley incorporated poultry litter in the soil. These distinctions may have contributed to their conclusions, because in both situations the manure comes into direct contact with the soil and this direct contact may have facilitated the adsorption of soluble P by the soil. In contrast, when dry poultry litter is surface applied to pastures and hayfields, a surface layer of thatch is likely to prevent direct contact between the litter and the soil and reduce the possibility that P in the manure will be adsorbed by the soil. Using fescue plots treated with dry poultry litter, Edwards et al. (1994a) observed that intervals of 4, 7, and 14 days between manure application and rainfall had no effect on either concentration or mass of ortho-P or TP in runoff. In a related study, drying time had no effect on the transport of manure particles from poultry litter treated sites (Edwards et al., 1994b).

Phosphorus Index

From the preceding discussion it is clear that there is uncertainty over the relative importance of the various sources of P to the actual levels of P in runoff. Additionally, the impact of these sources of P on sensitive water bodies will be affected by parameters that control runoff volume. To address these issues and develop a method for assessing the overall risk of P loss from individual sites, the P Index was developed (Lemunyon and Gilbert, 1993). The original P Index was an 8 x 5 matrix of landform site characteristics including erosion, runoff class, STP, fertilizer application rate, and manure application rate. Each of these factors was assigned a rating from 0 to 8 depending on the risk level of that factor, 0 being no risk and 8 being very high risk. Once each characteristic is assigned a value that value is multiplied by a "weighting factor", the weighted values are added together to arrive at the P Index rating. One problem with this approach has been the assignment of the weighting factors. These factors were based on the "professional judgement" of the scientists involved, not on scientific data. Studies are currently under way to determine these weighting factors based on scientific data. The advantage of the P Index over simply relying on STP as an indicator of P loss risk is the inclusion of transport factors. The P Index, in effect, allows the identification of sites where P loss is likely, and where remediation or alternative management practices can best be put to use.

Since the original P Index was developed, several researchers and state agencies have developed variations of the P Index. Most of these variations are very similar to Lemunyon and Gilbert's original index with a few notable exceptions. A recent modification of the P Index is the addition of "critical source areas" (CSA), which are specific identifiable areas within a watershed that pose the greatest risk of P loss (Gburek et al., 2000). This approach puts more emphasis on areas closest to water bodies based on storm return period and its impact on runoff from near stream areas. Storm return period refers to the probability that a storm of some given intensity will occur during that period of time. The area around a stream that contributes runoff is based on storm size and return period. Areas that have a high return period, high soil P levels, and greatest runoff will contribute the most to P runoff and therefore receive the highest P Index rating.

Another approach to the P Index has been taken in Georgia, where the risk of P loss from a site has been divided into ratings related to the risk of soluble, particulate, and leaching losses of P (Radcliffe, personal communication, 2000). Including both source and transport factors, each of these ratings is an independent indicator of the risk of P loss. This approach evaluates the risk of P loss based on the factors that actually contribute to P loss from a specific site and was taken in Georgia because there are distinct regions in Georgia (i.e. Piedmont vs. Coastal Plain), where the risk of P loss is dominated by only one or two of these P loss pathways. For example, in the Piedmont surface runoff and STP may control P loss, but in the coastal Plain STP and sub-surface flow may be the main P loss pathways.

Computer Models

Phosphorus indices are valuable tools for identifying sites with high risk of P loss, but they are not well suited to predicting actual P loss. Since it is impractical to perform actual experiments on individual sites or watersheds to determine actual P loss, indirect methods of estimating actual P loss, such as computer models, must be used. Modeling can be a cost-effective approach if the chosen equations and parameters adequately reflect the actual chemical and physical process occurring.

Computer models have been developed to simulate the fate and transport of non-point source pollutants including pesticides, metals, and nutrients. Several Computer models, including CREAMS (Knisel, 1980), ANSWERS (Beasley and Huggins, 1982), SWAT (Arnold et al., 1998), AGNPS (Young et al., 1989), GLEAMS (Leonard, 1987), and EPIC (Sharpley and Williams, 1990) have been developed to predict the impact of agricultural practices on water quality. Several researchers have used EPIC to model P loss from pastures and hayfields receiving poultry litter (Edwards et al., 1994; Pierson et al., 2001; and Yoon et al., 1993).

Phosphorus Runoff in EPIC

The modeling of P loss in EPIC is based on splitting P in runoff into soluble P and particulate P based on the concept of partitioning pesticides into solution and sediment phases as described by Wauchope and Leonard (1980). Most of the P lost in runoff is assumed to be associated with the sediment phase, and sediment transport of P is simulated with a loading function developed by McElroy et al. (1976) and modified for application to single events by Williams and Hann (1978). The function is

$$YP = 0.001\ (Y)\ (c_p)\ (ER)$$

where YP is the of PP lost in runoff (kg ha^{-1}), c_p is the concentration of P in the surface soil layer (g Mg^{-1}), Y is the sediment yield (kg ha^{-1}), and ER is the enrichment ratio. The thickness of thr surface soil layer is user defined. The enrichment ratio is the concentration of P in the sediment divided by the P concentration in the soil.

Because most of the P lost in runoff is assumed to be associated with the sediment phase, soluble P loss is described by the following simple equation

$$YSP = 0.01\ (c_{LPl})\ (Q)\ /\ k_d$$

where YSP is soluble P (kg ha^{-1}) lost in runoff, Q is runoff volume, c_{LPl} is the available P (g Mg^{-1}) in horizon l , and k_d is the concentration of P in sediment divided by the P concentration of the runoff water. The default value of k_d used in EPIC is 175. This value was chosen as a best fit estimate by Williams (1995).

Recently, a modification has been developed for EPIC's soluble P loss equation because the original EPIC underestimated soluble P losses when surface applied manure was present (Williams, Personal communication, 1999). The modified soluble P equation uses a nonlinear function of organic P to improve the soluble P loss estimate

$$YSP = ((AP)(Q)(RTO))/0.1(Wt)(k_d) \text{ and}$$

$$RTO = 10(Wp/Wt)$$

where AP is the labile P content of the upper 10 cm of the soil, Q is runoff volume, k_d is the P concentration of the sediment divided by the P concentration of the water, Wt is soil weight, and Wp is the organic P content of the soil (mg kg^{-1}).

The unmodified EPIC P model produced fairly accurate results under a variety of conditions. The unmodified version of EPIC accurately predicted soluble P loss, but overpredicted sediment P loss from conventionally tilled corn plots in Alabama (Yoon et al., 1993). The overprediction of sediment P was due to extremely high sediment loss predictions. Edwards et al. (1994) used EPIC to predict P losses in runoff from pastures in Arkansas. They observed that EPIC was a reasonably good predictor of runoff volume, TP, and DRP on both an annual and an event basis. However, they also observed that predicted P losses were less accurate when poultry litter (manure and bedding) was surface applied to a plot. More recent work (Pierson et al., 2001) has shown that both the original EPIC P model and the modified EPIC P model underestimated DRP losses from fields treated with poultry litter on an event basis, but on an annual basis the modified EPIC model was fairly accurate. Because of the widespread surface application of poultry litter, future research should include modification of the EPIC P model or development of a new sub-model to improve the prediction of P loss from surface applied poultry litter.

The overall objectives of this research are three fold. First, to determine the relationship between several measures of STP and P lost in runoff from grasslands where the Cecil soil series is present. Second, to determine the impact of surface applied unincorporated poultry litter on P loss from grasslands and to develop equations to predict this loss. Third, to evaluate the equations relating STP and litter application to P in runoff by comparing predicted DRP loss to DRP loss observed at field sites in

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REFERENCES

Arnold, J.G., B.A. Engel, and R. Surinivasan. 1993. A continuous time, grid cell watershed model. Proceedings of the Application of Advanced Information Technologies: Effective Management of Natural Resources Conference. ASAE, St. Joseph, MI.

Beasley, D.B., and L.F. Huggins. 1982. ANSWERS: Areal non-point source watershed environmental response simulation. User's manual. USEPA Rep. 905/9-82-001. USEPA, Chicago, IL.

Edwards, D.R., and T.C. Daniel. 1994. Quality of runoff from fescuegrass plots treated with poultry litter and inorganic fertilizer. J. Environ. Qual. 23:579-584.

Edwards, D.R., and T.C. Daniel, P.A. Moore, Jr., and P.F. Vendrell. 1994a. Drying interval effects on quality of runoff from fescue plots treated with poultry litter. Trans. ASAE. 37:831-843.

Edwards, D.R., and T.C. Daniel, P.A. Moore, Jr., and A.N. Sharpley. 1994b. Solids transport and erodibility of poultry litter surface-applied to fescue. Trans. ASAE. 37:771-776.

Gartley, K.L., and J.T. Sims. 1994. Phosphorus soil testing: Environmental uses and implications. Commun. Soil Sci. Plant Anal. 25:1565-1582.

Gburek, W.J., A.N. Sharpley, L. Heathwaite, and G.J Folmar. 2000. Phosphorus management at the watershed scale: A modification of the phosphorus index. J. Environ. Qual. 29:130-144.

Guertal, E.A., D.J. Eckert, S.J. Traina, and T.J. Logan. 1991. Differential phosphorus retention in soil profiles under no-till crop production. Soil Sci. Soc. Am. J. 55:410-413.

Heathman, G.C., A.N. Sharpley, S.J. Smith, and J.S. Robinson. 1995. Land application of poultry litter and water quality in Oklahoma, U.S.A. Fert. Res. 40:165-173.

Heckey, R.E., and P. Kilham. 1988. Nutrient limitation of phytoplankton in freshwater and marine ecosystems: A review of recent evidence of the effects of enrichment. Limnol. Oceanogr. 33:796-822.

Hsu, P.H. 1965. Fixation of phosphorus by aluminum and iron in acidic soils. Soil Sci. 99:398-402.

Knisel, W.G. (ed.). 1980. CREAMS: A field scale model for chemical runoff and erosion from agricultural management systems. Conserv. Res. Rep. 20. USDA-ARS, Washington, DC.

Lemunyon, J.L. and R.G. Gilbert. 1993. The concept and need for a phosphorus loss assessment tool. J. Prod. Agric. 6:483-486.

Leonard, R.A., W.G. Knisel, and D.A. Still. 1987.GLEAMS: Groundwater Loading Effects of Agricultural Management Systems. Trans. ASAE. 30: 1403-1418. Lindsay, W.L., and E.C. Moreno. 1960. Phosphate phase equilibria in soils. Soil Sci. Soc.Am. Proc. 24:177-182.

Lindsay, W.L. and E.C. Moreno. 1960. Phosphate phase equilibria in soils. 1960. Soil sci. Soc. Am. Proc. 24:177-182.
McElroy, A.D., S.Y. Chiu, J.W. Nebgen. 1976. Loading functions for assessment of water pollution from nonpoint sources. Environ. Prot. Tech. Ser., EPA 600/2-76-151, 445pp.
McLeod, R.V. and R.O. Hegg. 1984. Pasture runoff water quality from application of inorganic and organic nitrogen sources. J. Environ. Qual. 13:122-126.
Morgan, M.A. 1997. The behavior of soil and fertilizer phosphorus. p. 137-149. *In* H. Tunney, O.T. Carton, P.C. Brookes, and A.E. Johnston (eds.) Phosphorus loss from soil to water. CAB International, New York.
Meuler, D.H., R.C. Wendt, and T.C. Daniel. 1984. Phosphorus losses as affected by tillage and manure application. Soil Sci. Soc. Am. J. 48:901-905.
National Agricultural Statistics Service. 1999. USDA-NASS Agricultural Statistics 1999. U.S. Department of Agriculture Washington, D.C.
Pierson, S.T., M.L. Cabrera, G.K. Evanylo, H.A. Kuyrkendall, C.S. Hoveland, M.A. McCann, and L.T. West. 2001. Phosphorus and ammonium concentrations in surface runoff from grasslands fertilized with broiler litter. J. Environ. Qual. 30:1784-1789
Pote, D.H., T.C. Daniel, D.J. Nichols, A.N. Sharpley, P. A, Moore, Jr., D.M. Miller, and D.R. Edwards. 1999. Relationship between phosphorus levels in three ultisols and phosphorus concentrations in runoff. J. Environ. Qual. 28:170-175.
Rubaek, G.H., G. Goggenberger, W. Zech, and B.T. Christensen. 1999. Organic phosphorus in soil size separates characterized by phosphorus-31 nuclear magnetic resonance and resin extraction. Soil Sci. Soc. Am. J. 63:1123-1132.
Sauer T.J., T.C. Daniel, P.A. Moore, Jr., K.P. Coffey, D.J. Nichols, and C.P. West. 1999. Poultry litter and grazing animal effects on runoff water quality. J. Environ. Qual. 28:860-865.
Sharpley, A.N. 1981. The depth of rainfall-runoff-soil interaction as determined by ^{32}P. Water Res. Res. 17:969-974.
Sharpley, A.N. 1985a. Phosphorus cycling in unfertilized and fertilized agricultural soils. Soil Sci. Soc. Am. J. 49:905-911.
Sharpley, A.N. 1985b. Depth of surface soil-runoff interaction as affected by rainfall, soil slope, and management. Soil Sci. Soc. Am. J. 49:1010-1015.
Sharpley, A.N. 1991. The measurement of bioavailable phosphorus in agricultural runoff. J. Environ. Qual. 20:235-238.
Sharpley, A.N. 1993. An inovative approach to estimate bioavailable phosphorus in agricultural runoff using iron oxide-impregnated paper. J. Environ. Qual. 22:597-601.
Sharpley, A.N. 1995. Dependence of runoff phosphorus on extractable soil phosphorus. J. Environ. Qual. 24:920-926.
Sharpley, A.N. 1997. Rainfall frequency and nitrogen and phosphorus runoff from soil amended with poultry litter. J. Environ. Qual. 26:1127-1132.
Sharpley, A.N. 2000. Phosphorus availability. p 18-33. *In* M.E. Sumner (ed.) Handbook of soil science. CRC Press, Boca Raton, FL.
Sharpley A.N., and J.R. Williams, (eds).1990. EPIC: Erosion/Productivity Impact Calculator; 1. Model Documentation. USDARS Tech. Bull. 1768.

Sharpley, A.N., S.J. Smith, O.R. Jones, W.A. Berg, and G.A. Coleman. 1992. The transport of bioavailable phosphorus in agricultural runoff. J. Environ. Qual. 21:30-35.

Sharpley, A.N., T.C. Daniel, J.T. Sims, and D.H. Pote. 1996. Determining environmentally sound soil phosphorus levels. J. Soil Water Cons. 51:160-166.

Sibbesen, E., and A.N. Sharpley. 1997. Setting and justifying upper critical limits for phosphorus in soils. p. 151-176. *In* H. Tunney, O.T. Carton, P.C. Brooks, and A.E. Johnston (eds.) Phosphorus loss from soil to water. CAB Int. New York.

Stewart, .J.W.B, and A.N. Sharpley. 1987. Control on dynamics of fertilizer phosphorus and sulfur. Soil Sci. Soc. Am. Special Publication Series. 19:101-121. Soil Sci. Soc. Am., Madison, WI.

Taylor, R.W., and B.G. Ellis. 1978. A mechanism of phosphate adsorption on soil and anion exchange resin surfaces. Soil Sci. Soc. Am. J. 43:432-436.

USEPA.1996. Environmental indicators of water quality in the United States. USEPA Rep. 841-R-96-002.

Vervoort, R.W., D.E. Radcliffe, M.L. Cabrera, and M. Latimore, Jr. 1998. Field scale nitrogen and phosphorus losses from hayfields receiving fresh and composted broiler litter. J. Environ. Qual. 27:1246-1254.

Westerman, P.W. and M.R. Overcash. 1980. Short-term attenuation of runoff pollution potential for land applied swine and poultry manure. p. 289-292. *In* R.J. Smith (ed) Livestock waste: A renewable resource. Proc. 4th International Symp. On Livestock Wastes. Amarillo, Tx, 15-17 Apr. ASAE, St. Joseph, MI.

Wauchope, R.D. and R.A. Leonard. 1980. Maximum pesticide concentrations in agricultural runoff – a semi-empirical prediction formula. J. Environ. Qual. 9:335-672.

Williams, J.R. 1995. The EPIC model. pp. 909-1000. *In* V.P. Singh (ed.) Computer models of watershed hydrology. Water Res. Publications, Littleton, CO.

Williams, J.R., and R.W. Hann. 1978. Optimal operation of large agricultural watersheds with water quality constraints. Texas Water Resources Institute, Texas A&M Univ., Tech. Rep. No. 96, 152pp.

Wood, B.H., C.W. Wood, K.H. Yoo, K.S. Yoon, and D.P. Delaney. 1999. Seasonal surface runoff losses of nutrients and metals from soils fertilized with broiler litter and commercial fertilizer. J. Enviorn. Qual. 28:1210-1218.

Yoon, K.S., K.H.Yoo, C.W. Wood, and B.M. Hall. 1993. Epic simulation of water quality impact by land application of poultry litter. International Winter Meeting, Paper No. 93-2531/93-2550. ASAE, St Joseph, MI.

Young, R.A., C.A. Onstad, D.D. Bosch, and W.P. Anderson. 1989. AGNPS, a nonpoint source pollution model for evaluating agricultural watersheds. J. Soil Water Conserv. 44:168-173.

CHAPTER 2

RELATIONSHIP BETWEEN SOIL TEST PHOSPHORUS AND PHOSPHORUS IN RUNOFF FROM PASTURES AND HAYFIELDS IN THE GEORGIA PIEDMONT [1]

[1] P. D. Schroeder, D. E. Radcliffe, M. L. Cabrera and C. D. Belew. Submitter to J. Environ. Qual. 3/15/02

ABSTRACT

Phosphorus in runoff from agricultural fields to surface waters has been identified as an important contributor to eutrophication. The objective of this research was to determine the relationship between total P (TP), dissolved reactive P (DRP), and bioavailable P (BAP) in runoff from a benchmark soil (Cecil series) and Mehlich III, DI water, and Fe_2O_3 extractable soil P, and to determine if other, readily available, soil properties could contribute to P loss prediction. Simulated rainfall was applied (75 mm hr^{-1}) to 54 1x1 m plots with different STP levels. Runoff was collected in toto for 30 min. and analyzed for TP and DRP. Soil samples were collected from 02, 0-5, and 0-10 cm depths. There was significant correlation between all forms of P in runoff and all soil P test methods used. STP correlated best with runoff P when extracted with Fe oxide-coated paper or distilled water. The strongest correlation between STP method and DRP in runoff occurred with water extractable soil P (R^2 = 0.68). Normalizing DRP by runoff volume resulted in improved correlation with DI water extractable P. Although STP levels were different among sampling depths, comparison of the slopes and intercepts of the regression lines revealed there was no difference in the relationship between STP and P in runoff due to soil sampling depth. For all forms of P in runoff and all STP methods, R^2 increased with the inclusion of extractable Fe and landscape position in the regression equation. Overall, the results of this study indicate that, no matter what STP method is used, what soil sampling depth chosen, or form of runoff P measured, P loss can be predicted with some accuracy. Furthermore, the addition of a few easily obtainable soil and site characteristics, such as pH, extractable Fe, and landscape position, can significantly improve the prediction of P loss compared to STP alone.

INTRODUCTION

Over the past decade, controlling non-point source pollution has come to the forefront in efforts to improve water quality in the United States and elsewhere. The principal components of agricultural non-point source pollution are sediment, bacteria, nitrogen (N), and phosphorus (P). Of these, P is the element most commonly associated with eutrophication in freshwater systems because these systems are usually P limited (Correll, 1998).

Traditionally, soil chemists considered P to be relatively immobile in soil because the rates of fertilizer P application are commonly low compared to the soil sink for P. However, recent studies show that there is a limit to this "sink" and that, over time, P levels can build to the point where a soil's capacity to "sorb" P is exceeded. When this happens, P added in fertilizer or animal manure can be lost to the surrounding environment. In areas of concentrated livestock production, land application of large volumes of manure, enriched with imported nutrients, is commonplace.

The state of Georgia is one of the top poultry (*Gallus gallus domesticus*) producing regions of the United States, with 1.2 billion broilers raised in 1998 (National Agricultural Statistics Service, 1999). Besides broiler production, agriculture in central and north Georgia is generally limited to the production of beef cattle and its associated pastures and hayfields. Many farms in this region combine cattle and broiler production. Soil testing data reveal that a significant number of the samples analyzed by the Georgia Cooperative Extension Soil Testing Lab have extremely high levels of plant available P (D.E.. Kissel, personal communication, 2000).

There is evidence that soils with high soil test P (STP) levels can contribute significant amounts of P to runoff waters either as dissolved reactive P (DRP) or as particulate bound P (PP) (Pote et al., 1999b; Sharpley, 1995). Because the relationship between STP and runoff P is soil and site specific, it is difficult to assign universally acceptable STP values above which P loss is unacceptable. Additionally, there are currently no federal standards for P levels in surface waters, although ambient water quality criteria have been recommended recently by the USEPA.

Forms of P in runoff can include DRP in both organic and inorganic forms as well as PP associated with mineral or organic particles transported in the runoff. Because erosion rates from hayfields and pastures are low, DRP is usually the dominant form of phosphorus in hayfield and pasture runoff (Sharpley et al., 1992). Dissolved P in runoff results from the interaction of rainwater with a thin layer of the soil's surface. The effective depth of this interaction is dependent on soil aggregation, percent ground cover, slope, and rainfall intensity (Sharpley, 1985). In two studies on effective depth of interaction, this depth ranged from 0.2 cm to more than 3.7 cm depending not only on the factors mentioned above but also on the experimental method used (Sharpley et al., 1981, Sharpley, 1985).

Chemical extractants that simulate the availability of soil P have been used for decades to predict crop yield response to added P in fertilizer. According to Gartley and Sims (1994), the most common "routine" soil tests currently in use in the United States include Bray and Kurtz (0.025 N HCl + 0.03 N NH_4F), Mehlich I (0.025 N H_2SO_4 + 0.05N HCl), Mehlich III (0.02 N CH_3COOH + 0.025 N NH_4NO_3 + 0.015 N NH_4F + 0.013 N HNO_3 + 0.01 M EDTA), Morgan (0.72 N NaOAc + 0.52 N CH_3COOH , pH 4.8), Modified Morgan (0.62 *N* NaOAc + 1.52 *N* CH_3COOH, pH 4.8), and Olsen-P (0.5 *N* $NaHCO_3$). The traditional interpretations of these tests as predictors of plant available P are based on extensive research, but there are less data to support interpretations of potential environmental impacts of soils that test high in P. Additionally, most soil samples submitted for agronomic testing are collected from a depth of 0 to 15 cm, but research has shown that as a result of long-tern manure application, P accumulates at the soil's surface. Phosphorus levels in the upper 2 cm of no-till fields may be three times higher than at 8 cm (Guertal et al., 1991). However, a review of literature published on the subject reveals that there is a strong relationship between STP and P in runoff (Table 1). Although these studies covered a wide range of soil types, STP methods, and soil sampling depths, the reported regression slopes are similar and for the most part coefficients of determination were > 0.60. The biggest differences among the reported regression relationships were in the value of the intercept. The fact that some of the intercepts are positive and some are negative suggests that the STP method in question either extracted more P than runoff water (positive intercept) or less P than runoff water (negative intercept). This effect is likely due to soil specific factors that regulate the interaction between runoff water and the soil's surface. Another significant implication of these studies is that these data seem to support the prevailing wisdom that sampling depth has a direct impact on the relationship between STP and P in runoff since there seems to be a positive relationship between sampling depth and R^2 (Table 1.1). However there is a considerable range in the R^2 values reported and if the values associated with tilled or packed soils are disregarded the evidence that sampling depth is important is not as strong. Additionally, none of the researchers cited in Table 1 studied the effect of soil sampling depth on the relationship between STP and P in runoff.

While the debate over environmental interpretations of traditionally agronomic soil tests continues, some researchers have developed alternative methods that may be more suited to environmental P assessments. One alternative for assessing P in soil is water extractable P. Because distilled water most closely resembles actual runoff, distilled water may be the most appropriate soil extractant for predicting DP in runoff.

Another alternative is to attempt to quantify the P in runoff that is directly available to aquatic plants, termed bioavailable P (BAP). Bioavailable P may be quantified by extraction with iron-oxide impregnated paper (Sharpley, 1993). The coated paper acts as a P sink that adsorbs both DP and some of the P that is adsorbed to soil particles. Iron oxide coated paper can also be used to estimate BAP in soil samples.

The objective of this research was threefold. First; to determine the relationship between the concentration of DRP, bioavailable P (BAP), and total P (TP) in runoff from hayfields and pastures, and three different measures of soil P for a benchmark soil in the Piedmont region of Georgia, second; to study the effect of soil sampling depth of this relationship and, third; to determine if other, readily available, chemical and physical properties could contribute to P loss prediction.

MATERIALS and METHODS

Field Plots

Experiments were conducted on soils of felsic igneous and metamorphic parent materials from the Piedmont region of Georgia. The benchmark soils chosen for this study are fine, kaolinitic, thermic Typic Kanhapludults (Cecil, Madison, and Pacolet soil series). The well-drained Cecil soil comprises 14.7% of all soils mapped in the Piedmont. Madison and Pacolet soil series were included in this study because their surface horizons are very similar to that of Cecil soil and they are often mapped together.

Six sites in the Piedmont region of Northeastern Georgia were selected. Three criteria were utilized in site selection: 1), Cecil and associated (Madison and Pacolet) soils present; 2), Mehlich III Soil Test P (STP) level covering a range from low (<10 mg kg^{-1}) to very high (>37.5 mg kg^{-1}); and 3), no manure application in the previous 12 months in order to eliminate fresh manure as a source of P that might not be reflected in STP. Soil series was verified from borings prior to field experimentation. The owners and/or operators of the fields verified manure application history verbally.

Following site selection, plot areas were mowed to a uniform (10 cm) height one week prior to the initial rainfall simulation. Plots were pre-soaked 24 hours prior to the first simulated rainfall event via a drip-irrigation system with approximately 400 L of water to standardize antecedent soil moisture levels and reduce time to runoff. Paired 1 x 1-m plots were installed at three different side-slope positions within each site (top, middle, and bottom). Plot borders consisting of approximately 0.3-cm thick sheet metal 15-cm tall were pressed into the ground to a depth of at least 7 cm to isolate runoff. Aluminum flumes were installed at the down-slope edge of each plot to divert surface runoff to a collection point.

A total of 54 rainfall simulations (3 rainfall events, 3 paired plots, and 6 sites) were conducted in this study. The rainfall scheme consisted of three rainfall simulations at 48-hour intervals at each of the six sites. Cassel and Nielson (1986) reported that a 48-hour delay between rainfall events was sufficient time for soil to return to field capacity.

Simulated rainfall was applied to each pair of plots with a JOERNS INC. (West Lafayette, Indiana) rainfall simulator. Local well water was used as the water source for the simulator and rainfall was applied at a rate of 75 mm hr^{-1} (range =65 to 85 mm hr^{-1} STDEV = 3.2 mm hr^{-1}) to allow comparison of runoff between the six sites.

Sampling Methods

Collection of runoff began after significant runoff commenced and continued for 30 minutes. Runoff was collected *in toto* and a 500-mL composite sample was taken and immediately placed on ice. Total runoff volume was recorded and a source water sample taken. In the lab, 125 mL of each runoff sample was filtered (0.45-µm pore diameter) to remove particulate matter. All samples were acidified with concentrated HCl and stored at -20°C until analyzed.

After the 3rd rainfall event, soil samples were collected to depths of 0-2 cm, 0-5 cm, and 0-10 cm (composites of ten random samples) from within each plot. Soil samples were air dried, ground, and sieved to 2 mm to remove large rock fragments and most of the grass thatch material, then oven-dried for 48 hours at 35°C (A.N. Sharpley, 2000, personal communication).

Chemical Analysis

Soil pH was determined in a 1:2 soil/water mixture using a glass electrode, and particle size distribution was determined by the pipette method (Kilmer and Alexander, 1949). Extractable P in each soil sample was determined using three different methods: Mehlich III (Mehlich, 1984), Mehlich I (Nelson et al., 1953), Fe_2O_3 Paper circles (Myers et al., 1997), and DI water (Pote et al., 1996). Mehlich III extractable Fe was determined by atomic adsorption spectrophotometry. Total carbon content was determined by combustion in a LECO (St. Joseph, Michigan) CNS analyzer.

Total P for all unfiltered runoff samples was determined colorimetrically (Murphy and Riley, 1962) following Kjeldahl digestion. Dissolved Reactive P (DRP) for all filtered runoff samples was also determined colorimetrically. Bioavailable P (BAP) was determined for the unfiltered field runoff samples by the Fe oxide-coated paper circle method (Myers et al., 1997) and measured colorimetrically. The Fe oxide-coated paper circle method is a slight modification of the filter strip paper method documented by Sharpley (1993).

Data Analysis

For each soil test method, the mean, range, and coefficient of variation (CV) was calculated by site and depth for all samples and analyzed. For each P runoff analysis method, the mean, range, and CV was also calculated by site and analyzed. Flow weighted TP, DRP, and BAP concentration was calculated for each site and rainfall event by multiplying the P in runoff from each plot by runoff volume from that plot and then adding them together and dividing by the total runoff volume from the two plots. Using the flow weighted P values from each rainfall simulation, overall flow weighted TP, DRP, and BAP, were calculated by adding these values together for the three simulated rainfall events at 48-hr intervals, and dividing by the total runoff volume collected for the three events.

The concentration of each form of P in runoff was regressed against each measure of STP. Stepwise multiple regression, with a 0.25 significance level for entry, was also employed to determine if including extractable Fe, pH, texture, total carbon content, or landscape position would improve the ability to predict P levels in runoff. Analysis of variance was applied to both runoff P levels and STP levels. Statistical analyses were performed using the Statistical Analysis Software (SAS Institute, 1987).

The principle of conditional error, (Bose,1949; Milliken and Johnson, 1984) was used to evaluate regression equations relating STP to P in runoff. This is a technique for obtaining the sum of squares due to deviations from a hypothesis for linear models. The null hypothesis first tested was that one equation could be used to describe TP, DRP and BAP in runoff vs STP. This procedure provided an estimate of the residual sum of squares for the null hypothesis. Secondly, the alternative hypothesis that a separate equation was needed for each relationship was tested. The addition of the residual sum of squares for each separate equation provided an estimate of the residual sum of squares for the alternative hypothesis. The difference between the residual sum of squares of the null and alternative hypotheses provided an estimate of the residual sum of squares due to deviations from the null hypothesis. This residual sum of squares was then used in an *F* test against the residual sum of squares of the alternative hypothesis to determine if the relationships between TP, DRP and BAP were significantly different.

RESULTS AND DISCUSSION

Soil Phosphorus

The range, mean, and standard deviation of STP extracted from all plots for each STP method and sampling depth are shown in Table 1.2. DI water extracted the smallest amounts of P followed by Fe_2O_3 Paper, and Mehlich III. Although DI water extracted P and Fe_2O_3 extracted P were strongly correlated (Figure 1.1), they were significantly different ($p < 0.01$). Additionally, there was a strong relationship between Mehlich III extractable P and both DI water extractable P and Fe_2O_3 Paper extractable P (Figure 1.2). The strong relationships between the different STP methods suggests that they are extracting P from the same pool and should exhibit a similar relationship with P in runoff.

Sampling depth significantly affected STP regardless of the extraction method used. All sampling depth/extractant combinations produced different P levels except for the Mehlich III P 0-2 cm and 0-5 cm sampling depths, and the 0-5 and 0-10 cm sampling depths (Table 1.2). For both DI water and Fe_2O_3 all the sampling depths produced different extractable P levels. Sharpley et al. (1994) reported dramatic accumulations of P in the surface 5 cm of soil when manure is not incorporated. Although the field sites chosen for this study had all received surface applied poultry manure in the past, none of the sites had received manure over the preceding 12 months. This, combined with the fact that soil samples were collected after three rainfall simulations, helps explain why we did not find the dramatic accumulation of P at the surface described by Sharpley et al. (1994). In general however, STP values did decrease as depth increased (Table 1.2).

P in Runoff vs. STP

Source water was analyzed for TP and ranged from 0.012 mg L^{-1} to 0.094 mg L^{-1}, with a mean of 0.04 mg L^{-1}. To eliminate this variability from data analyses, initial source water P concentrations were subtracted from TP, DRP, and BAP runoff concentrations.

Total P in runoff ranged from 0.42 to 1.25 mg L^{-1}, DRP ranged from 0.15 to 0.80 mg L^{-1}, and BAP ranged from 0.13 to 0.78 mg L^{-1} (Figure 1.3). BAP and DRP levels were similar at each site but DRP values showed greater variability with the CV of DRP ranging from 33 to 114%, whereas the CV of BAP ranged only from 17 to 57%. The high degree of variability in the DRP may be attributed to the variability of runoff volume. The relationship between DRP and runoff volume will be discussed in detail later.

For most sites in this study BAP exceeded DRP, but in several cases the opposite was true (Figure 1.3). Since BAP is measured by adsorption of P from unfiltered runoff onto Fe_2O_3 coated paper one would expect BAP to always exceed DRP, a measure of P that passes a 0.45 μm filter. However, P in runoff may be in organic forms that cannot be extracted by the Fe_2O_3 paper but that may be molybdate reactive and small enough to pass through a 0.45 μm filter.

The relationship between each STP method and overall flow weighted TP, DRP, and BAP in runoff from field simulations are summarized in Table 1.3. Strong relationships were found between all forms of P in runoff and all soil P test methods. The strongest relationships ($R^2 = 0.69$) between STP and P in runoff were obtained with Mehlich III, and BAP soil test on 5-cm samples and TP in runoff, and between Mehlich III, DI water and BAP soil test on 5-cm samples and BAP in runoff. The highest correlation between STP method and DRP in runoff occurred with water extractable P and the 0-5 cm sampling depth ($R^2 = 0.68$). Since water is the solvent for P in runoff, a strong correlation was expected between distilled water extractable P and DRP in runoff. Similar to Pote et al. (1996, 1999a,b), we found that the strongest correlations to DRP and BAP in runoff were obtained with distilled water or the Fe-oxide paper method (BAP). It is important to note, however, that DRP and Mehlich III, at 0-5 cm depth, were also strongly correlated ($R^2 = 0.64$). Regardless of the STP method or runoff P form, the strongest correlation between the two was observed for soil samples collected to 5 cm.

Since Mehlich I is the STP method employed by the University of Georgis Soil Testing Lab, we regressed P extracted from 0-10 cm samples by Mehlich I against both TP and DRP in runoff (Fig. 1.4). The relationship between Mehlich I and P in runoff is similar to that with Mehlich III. In fact the slopes are not significantly different ($p < 0.05$) for the relationships between Mehlich I and III with either TP or DRP (0.004 vs. 0.003, and 0.003 vs. 0.002 for TP and DRP, respectively). However, the intercepts were different with Mehlich III having a smaller intercept with both TP and DRP (0.44 vs. 0.55, and 0.15 vs. 0.24 for TP and DRP, respectively. The fact that the Melich I intercepts were higher than the Mehlich III indicates that Mehlich I less effectively extracted P that could be lost from the soil to runoff than did Mehlich III

The relationships between STP in the 0-2 cm samples and P in runoff reported here were not as strong as those reported by Pote et al. (1996). There are two likely reasons why we did not observe the same level of correlation as reported by Pote et al. (1996, 1999b). The first lies in the much lower STP levels present at the field sites chosen for this study and the natural variability in soil chemical and physical properties associated with such sites. Plots used in studies by Pote et al. (1996) were constructed at a single site in 1990 on well-established fescue pastures and STP levels had been controlled and manipulated by the additions of various fertilizer amendments over several years. Additionally, limiting our sites to actual agricultural fields where poultry litter had not been applied in the preceding 12 months made it difficult to find sites with very high

STP (>400 mg P kg^{-1}). As a result, we did not have the opportunity to study the relationship between STP and P in runoff for sites with very high STP values.

Secondly, because the soil samples in this study were collected after three rainfall events some amount of the readily available P in these samples was “extracted” by the rainfall and therefore not available for extraction in the lab. We did not collect samples before the initial rainfall event because removing 30 cores from an area of only one square meter would have caused significant surface disturbance. As a result the relationship between 0-2 cm STP and P in runoff may have been degraded. This is supported by the results of analyzing the three rainfall events individually (results not shown). For both TP and BAP in runoff, the coefficient of determination with STP increased from the first rainfall event to the third. In effect, the rainfall event closest to the time of soil sampling produced a slightly better fit. Also of interest in the analysis of individual rainfall events was that the relationship between DRP and STP was constant across rainfall events with only the intercept of the regression equation changing as DRP concentration decreased from the first to third rainfall event (results not shown).

However, by calculating the mass of DRP lost from the three rainfall events we were able to estimate the impact of the three rainfalls on the DI water extractable P. Assuming that runoff water interacted with the upper 1 cm of soil ($\rho_b \approx 1.5$ g cm^3) we calculated the total mass of DRP lost from the soil over the three rainfall events (through both runoff and infiltration). This mass of P (mg kg^{-1}) was added to the DI water extractable soil P and the regression equation was recalculated. The regression equation for the “adjusted’ DI water extracted soil P was not significantly different from the original DI water extractable soil P equation (Fig. 1.5) suggesting that taking soil samples after three rainfalls had no significant impact on the relationship between STP and P in runoff.

Effect of Runoff Volume

The total runoff from each plot ranged from 0.09 mm to 37.5 mm, with a mean of 16.1 mm. There was appreciable runoff variability from the plots (CV = 55.1%). The runoff variability was most likely related to differences in permeability and antecedent moisture conditions, despite prewetting. Likewise, a portion of the variability in runoff volume was likely due to leakage around the plot borders and runoff collection flumes although we worked carefully to seal the borders and flumes.

In general, from plots with similar STP levels, DRP concentrations were higher when runoff volumes were higher. Similar results were reported by Pote et al. (1999b), who suggested that this might be due to differences in infiltration rates, where high infiltration rates reduce runoff and increase transport of DRP into the soil where it is adsorbed. Further analysis of our data revealed that the relationship between DRP and runoff volume was most pronounced for the third rainfall event. This indicates that there were some adsorption/desorption processes taking place at the soil surface over the three events. We suggest that during the first two rainfall events some of the P desorbed from the upper portion of the plots, entered runoff, and was re-adsorbed before it reached the runoff collection flume. This had the effect of reducing the pool of desorbable P with progressive rainfall events, and in fact from the first to third rainfall event DRP as a percent of TP decreased from 65 to 42 %. So, by the third event most of the potential P adsorption sites close to the runoff flume were satisfied. Consequently as runoff volume increased DRP concentration increased because desorbed P could not be readsorbed. This effect would be enhanced in plots with high runoff volume because runoff from these plots would be more channelized (which was actually observed in some plots), thus increasing the extent of P adsorption site saturation. As a result, the runoff from the high runoff volume sites was able to maintain a higher DRP concentration.

In an attempt to reduce the effect of runoff volume variability on the relationship between P in runoff and STP, we normalized P concentration in runoff by dividing by the amount of runoff (expressed as a depth). When DRP was normalized, its relationships with Mehlich III extractable P and DI water extractable P improved. The strongest relationship between normalized DRP and STP was with DI water extractable P (Figure 1.6). The effect was most pronounced with the 0-10 cm soil sample where R^2 increased from 0.58 (Table 3) to 0.81 (Figure 6). For both TP and BAP normalization did not improve the relationship with STP when all data points were included. However, when one data point was removed the relationship R^2 increased from 0.58 to 0.81 for TP and Mehlich III P and from 0.59 to 0.74 for BAP and Mehlich III P. The point removed was one that had the lowest runoff volume (0.09 mm).

Soil Sampling Depth

The issue of the effect of soil sampling depth on the relationship between STP and P in runoff has not been directly addressed in the published literature. Most of the published studies (Table 1.1) involved soil samples collected from a single soil depth (0-2, 0-5, or 0-7.5 cm). None of the published studies employed more than one depth. The issue of sampling depth is important because there is a potential conflict between environmental sampling and agronomic sampling. We collected soil samples from three depths (0-2, 0-5, and 0-10 cm) so that we could compare our results with other studies and to determine if 0-10 cm samples (the depth recommended in Georgia for agronomic sampling of pastures and hayfields) could be used for environmental purposes. Although STP levels were numerically different among the three soil sampling depths (Table 1.2), comparison of the slopes and intercepts of the regression lines for each combination of STP and depth vs P in runoff revealed that for all sampling depths there were no statistical differences between the slopes and intercepts ($p < 0.05$). These results indicate that the depth over which soil samples are taken may not have a significant effect on the relationship between STP and P in runoff. This is an important finding in light of producer concerns over the possibility of their being required to take a soil sample for environmental analysis in addition to their agronomic samples.

Multiple Regression Analysis

By relating STP to P in runoff we were able to account for between 50 and 80% of the variability in P loss. However, since there were soil properties other than STP that could influence P loss it seemed reasonable that by including some of these properties we could develop better P loss prediction equations. We regressed P in runoff against STP, pH, texture, Mehlich III extractable Fe, total C, slope, and landscape position. For all forms of P and all STP methods R^2 increased with the inclusion of Mehlich III extractable Fe and landscape position in the regression equation (Table 1.4). For several runoff P vs STP combinations, inclusion of Mehlich III extractable Fe, landscape position, and pH provided the best equation. It is interesting to note that the coefficients on the Fe and pH factors were both positive indicating that an increase in extractable Fe or pH produced an increase in P loss. At first the positive relationship between Mehlich III extractable Fe and P loss seemed counter intuitive since one would expect a soil with more Fe to bind more P, however it is likely that this extractable Fe is related to some soluble Fe-P forms that were lost in runoff. Conversely, the coefficient of the landscape position terms were all negative indicating that P loss would be lower from lower landscape positions.

CONCLUSIONS

There were significant correlations between all forms of P in runoff and all soil P test methods used. However, STP correlated best with field runoff P when extracted with Fe_2O_3 Paper or DI water. The strongest relationship ($R^2 = 0.69$) between STP and P in field runoff was obtained with BAP soil test on 5-cm samples and BAP in runoff. The highest correlation between STP method and DRP in runoff occurred with DI water extractable P and the 0-5 cm sampling depth ($R^2 = 0.68$). These results are similar to those reported by Pote et al. (1996, 1999a,b). Normalizing DRP to reduce variability associated with differences in runoff volume resulted in improved correlation between DRP and DI water extractable P. The effect was most pronounced with the 0-10 cm soil sample where R^2 increased from 0.58 to 0.816.

Although STP levels were different among the three soil sampling depths, comparison of the slopes and intercepts of the regression lines for each combination of STP and depth vs P in runoff revealed that for all sampling depths there were no statistical differences between the slopes and intercepts of the regression lines. This indicates that sampling depth may not have a large impact on the relationship between STP and P in runoff. Also there was no difference between the DRP vs STP and the BAP vs STP regression lines indicating that one equation could be used to predict either form of P in runoff.

By relating STP to P in runoff we were able to account for between 50 and 80% of the variability in P loss. However, for all forms of P in runoff and all STP methods, R^2 increased with the inclusion of extractable Fe and landscape position in the regression equation. The coefficients on the Fe and pH factors were both positive indicating that an increase in extractable Fe or pH produced an increase in P loss. Conversely, the coefficient of the landscape position terms were all negative indicating that P loss was lower from the lower landscape positions.

Additionally, the results of this study indicate that, for Piedmont soils in hay or pasture, no matter what STP method is used or form of runoff P measured, P loss can be predicted with reasonable accuracy. Furthermore, the addition of a few easily obtainable soil and site characteristics can significantly improve the prediction of P loss compared to STP alone. Also, the fact that extractable Fe, pH, and landscape position had an effect helps explain why the relationship between STP and runoff P varies among some soils.

REFERENCES

Bose, R.C. 1949. Least squares aspects of analysis of variance. Institute of Statistics, Mimeo Series 9. The University of North Carolina, Chapel Hill, N.C.

Cassel, D.K., and D.R. Nielsen. 1986. Field capacity and available water capacity. *In* A. Klute (ed.) Methods of soil analysis, Part 1. Physical and mineralogical methods. Agronomy Monograph no 9 (2nd ed.).

Correll, D.L. 1998. The role of phosphorus in the eutrophication of receiving waters: A review. J.Environ. Qual. 28:261-266.

Cox F.R., and S.E. Hendricks. 2000. Soil test phosphorus and clay content effects on runoff water quality. J. Environ. Qual. 29:1582-1586.

Daniel, T.C, D.R. Edwards, and A.N. Sharpley. 1993. Effect of extractable soil surface phosphorus on runoff water quality. TRANS. ASAE. 36:1079-1085.

Gartley, K.L., and J.T. Sims. 1994. Phosphorus soil testing: Environmental uses and implications. Commun. Soil Sci. Plant Anal. 25:1565-1582.

Guertal, E.A., D.J. Eckert, S.J. Traina, and T.J. Logan. 1991. Differential phosphorus retention in soil profiles under no-till crop production. Soil Sci. Soc. Am. J. 55:410-413.

Kilmer, V.J., and L.T. Alexander. 1949. Methods of making mechanical analysis of soil. Soil Sci. Soc. A. J. 68: 15-24.

McDowell, R.W., and A.N. Sharpley. 2001. Approximating phosphorus release from soils to surface runoff and subsurface drainage. J.Environ. Qual. 30:508-520.

Mehlich, A. 1984. Mehlich III soil test extractant: A modification of Mehlich II extractant. Commun. Soil Sci. Plant Anal. 15:1409-1416.

Milliken, G.A., and D.E. Johnson. 1984. Analysis of messy data. Vol I: Designed experiments. Lifetime Learning Publications Belmont, California.

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.

Myers, R.G., G.M. Pierzynksi, and S.J. Thien. 1997. Iron oxide sink method for extracting soil phosphorus: paper preparation and use. Soil Sci. Soc. Am. J. 61:1400-1407.

National Agricultural Statistics Service. 1999. USDA-NASS Agricultural Statistics 1999. U.S. Department of Agriculture Washington, D.C.

Nelson, W. A. Mehlixh, and E. Winters. 1953. The development, evaluation, and use of soil tests for phosphorus availability. Agronomy 4:153-188.

Pote, D.H., T.C. Daniel, D.J. Nichols, P.A. More, D.M. Miller, and D.R. Edwards. 1999a. seasonal and soil-drying effects on runoff phosphorus relationships to soil phosphorus. Soil Sci. Soc. Am J. 63:1006-1012.

Pote, D. H., T.C. Daniel, D.J. Nichols, A.N. Sharpley, P.A. Moore, Jr., D.M. Miller, and D.R. Edwards. 1999b. Relationship between phosphorus levels in three Ultisols and phosphorus concentrations in runoff. J. Environ. Qual. 28:170-175.

Pote, D. H., T.C. Daniel, A.N. Sharpley, P.A. Moore Jr., D.R. Edwards, and D.J. Nichols. 1996. Relating extractable soil phosphorus to phosphorus losses in runoff. Soil Sci. Soc. Am. J. 60:855-859.

SAS Institute. 1987. Users guide. SAS Inst., Cary, NC.

Sharpley, A.N. 1995. Dependence of runoff phosphorus on extractable soil phosphorus. J. Environ. Qual. 24:920-926.

Sharpley, A.N. 1993. An innovative approach to estimate bioavailable phosphorus in agricultural runoff using iron oxide-impregnated paper. J. Environ. Qual. 22:597-601.

Sharpley, A.N. 1985. Depth of surface soil-runoff interaction as affected by rainfall, soil slope, and management. Soil Sci. Soc. Am. J. 49:1010-1015.

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.

Sharpley, A.N., S.J. Smith, O.R. Jones, W. A. Berg, and G.A. Coleman. 1992. The transport of bioavailable phosphorus in agricultural runoff. J. Environ. Qual. 21:30-35.

Sharpley, A.N., L.R. Ahuja, M. Yamamoto, and R.G. Menzel. 1981. The kinetics of phosphorus desorption from soil. Soil Sci. Soc. Am. J. 45:494-496.

List of Tables and Figures

Table 2.1. Relationship between STP and P in runoff as reported in the literature. Management refers to either the agricultural management or how the soils were handled in the study. The presence or absence of recent manure application in the 12 months preceding runoff collection is indicated under the heading "Manure?". Depth (cm) indicates soil sampling depth for STP determination.

Reference	Management	Soil type	Manure?	STP method	Depth (cm)	Slope	Intercept	R^2
McDowell and Sharpley, 2001	Grass	Calvin	?	MIII	0-7.5	0.0017	0.14	0.65
McDowell and Sharpley, 2001	Grass	Watson	?	MIII	0-7.5	0.0019	0.03	0.62
Cox and Hendrix, 2000	Tilled	Hiwassee	Yes	MIII	0-20	0.00141	forced 0*	0.9
Cox and Hendrix, 2000	No till	Hiwassee	Yes	MIII	0-20	0.00135	forced 0	0.62
Pote et al., 1999a	Grass	Captina	No	MIII	0-2	0.0022	0.255	0.62
Pote et al., 1999a	Grass	Captina	No	DI H_2O	0-2	0.018	0.087	0.71
Pote et al., 1999b	Grass	Nella	Yes	MIII	0-2	0.0036	-0.45	0.82
Pote et al., 1999b	Grass	Nella	Yes	DI H_2O	0-2	0.0107	-0.18	0.85
Pote et al., 1999b	Grass	Linker	Yes	MIII	0-2	0.0035	-0.38	0.84
Pote et al., 1999b	Grass	Linker	Yes	DI H_2O	0-2	0.0104	-0.11	0.86
Pote et al., 1999b	Grass	Noark	Yes	MIII	0-2	0.0016	0	0.87
Pote et al., 1999b	Grass	Noark	Yes	DI H_2O	0-2	0.0055	-0.03	0.94
Pote et al., 1996	Grass	Captina	No	MIII	0-2	0.0026	0.3	0.72
Pote et al., 1996	Grass	Captina	No	DI H_2O	0-2	0.0118	0.1	0.82
Pote et al., 1996	Grass	Captina	No	Fe_2O_3 paper	0-2	0.0077	0.1	0.82
Sharpley, 1995	Packed	San Sabra	Yes	MIII	0-1	0.0016	ng**	0.9
Sharpley, 1995	Packed	Shermoer	Yes	MIII	0-1	0.0047	ng	0.94
Sharpley, 1995	Packed	Stigler	Yes	MIII	0-1	0.0072	ng	0.96
Daniel et al., 1993	Grass	Captina	?	MIII	0-5	0.003	0.513	0.025
Daniel et al., 1993	Tilled	Captina	?	MIII	0-5	0	0.152	0.0004

* The authors calculated the regression slope from the origin after determining the intercepts to be negligible.
** Values for intercepts were not given by the author.

Table 2.2. P extracted (mg kg^{-1}) from soil samples by Mehlich III, distilled water, and iron oxide sink methods.

	Depth								
	0-2 cm			0-5 cm			0-10 cm		
STP method	STP range	Mean	CV†	STP range	Mean	CV	STP range	Mean	CV
	--------------------mg kg^{-1}--------------------								
Mehlich III	32-460	148a‡	79	28-403	134ab	78	31-357	120b	81
DI H_2O	4-67	24a	81	4-59	19b	84	5-56	15c	85
Fe2O3 Paper	10-86	32a	67	6-62	23b	70	4-61	17c	83

† Coefficient of variation. ‡ Means within each STP method row followed by the same letter are not different ($p < 0.01$).

Table 2.3. Relationship between STP method (mg kg^{-1}) and average P in runoff.

		Relationship with average runoff total P (mg L^{-1})					
STP method		Intercept	Slope	R^2	RMSE	Prob > F	n
Mehlich III							
	0-2 cm	0.43	0.0020	0.58	0.2129	0.0003	18
	0-5 cm	0.40	0.0025	0.69	0.1834	<0.0001	18
	0-10 cm	0.44	0.0025	0.58	0.2115	0.0002	18
DI H_2O							
	0-2 cm	0.42	0.0131	0.64	0.1958	<0.0001	18
	0-5 cm	0.42	0.0165	0.68	0.1853	<0.0001	18
	0-10 cm	0.46	0.0180	0.55	0.2199	0.0004	18
BAP							
	0-2 cm	0.38	0.0112	0.58	0.2129	0.0003	18
	0-5 cm	0.36	0.0162	0.69	0.1828	<0.0001	18
	0-10 cm	0.47	0.0155	0.48	0.2364	0.0015	18
		Relationship with average runoff DRP (mg L^{-1})					
STP method		Intercept	Slope	R^2	RMSE	Prob > F	n
Mehlich III							
	0-2 cm	0.15	0.0017	0.56	0.1794	0.0004	18
	0-5 cm	0.13	0.0020	0.64	0.1612	<0.0001	18
	0-10 cm	0.15	0.0020	0.58	0.1747	0.0002	18
DI H_2O							
	0-2 cm	0.14	0.0108	0.65	0.1604	<0.0001	18
	0-5 cm	0.14	0.0136	0.68	0.1525	<0.0001	18
	0-10 cm	0.16	0.0153	0.58	0.1746	0.0002	18
BAP							
	0-2 cm	0.11	0.0090	0.54	0.1819	0.0005	18
	0-5 cm	0.11	0.0126	0.61	0.1683	0.0001	18
	0-10 cm	0.18	0.0123	0.44	0.2019	0.0028	18
		Relationship with average runoff BAP (mg L^{-1})					
STP method		Intercept	Slope	R^2	RMSE	Prob > F	n
Mehlich III							
	0-2 cm	0.15	0.0016	0.60	0.1610	0.0002	18
	0-5 cm	0.14	0.0019	0.69	0.1420	<0.0001	18
	0-10 cm	0.16	0.0019	0.59	0.1654	0.0002	18
DI H_2O							
	0-2 cm	0.15	0.0104	0.68	0.1451	<0.0001	18
	0-5 cm	0.15	0.0129	0.69	0.1423	<0.0001	18
	0-10 cm	0.18	0.0141	0.56	0.1693	0.0004	18
BAP							
	0-2 cm	0.11	0.0089	0.60	0.1611	0.0002	18
	0-5 cm	0.10	0.0126	0.69	0.1412	<0.0001	18
	0-10 cm	0.18	0.0123	0.49	0.1816	0.0012	18

Table 2.4. Multiple regression results for P in runoff.

P runoff form	STP method	Intercept	STP	pH	Fe	Position[†]	R^2 initial	R^2 final
TP	MIII	0.36	0.002	---------	0.0012	-0.1026	0.69	0.79
	DI H_2O	0.27	0.014	---------	0.0015	-0.0955	0.68	0.83
	BAP	-0.67	0.007	0.215	0.0009	-0.0978	0.58	0.77
DRP								
	MIII	-0.54	0.001	0.144	0.0009	-0.1095	0.56	0.78
	DI H_2O	0.06	0.011	---------	0.0012	-0.0956	0.68	0.84
	BAP	-0.64	0.006	0.151	0.0009	-0.0998	0.54	0.78
BAP								
	MIII	-0.35	0.001	0.093	0.0009	-0.0757	0.69	0.83
	DI H_2O	-0.64	0.007	0.157	0.0007	-0.0758	0.68	0.86
	BAP	-0.77	0.006	0.176	0.0008	-0.0779	0.60	0.83

[†] Refers to landscape position for which we used the following code; 1 = Top, 2 = middle, and 3 = bottom.

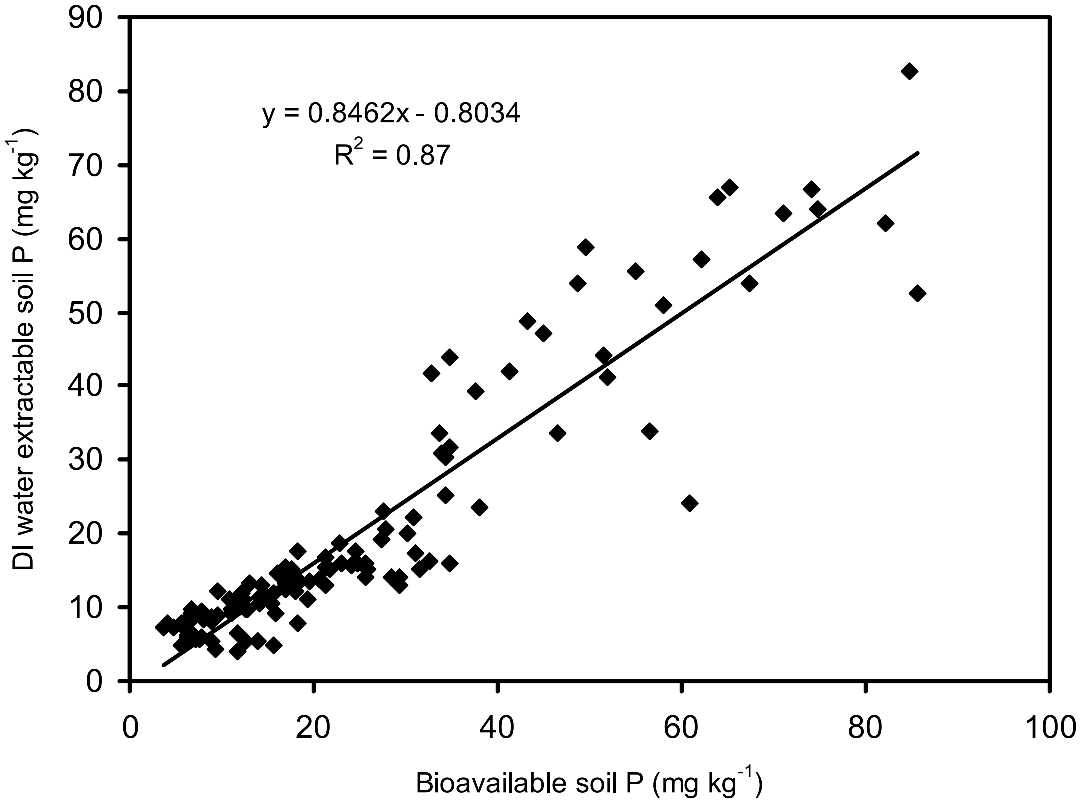

Figure 2.1. The relationship between P extracted from soil samples by Fe_2O_3 Paper (bioavailable P) and de-ionized water.

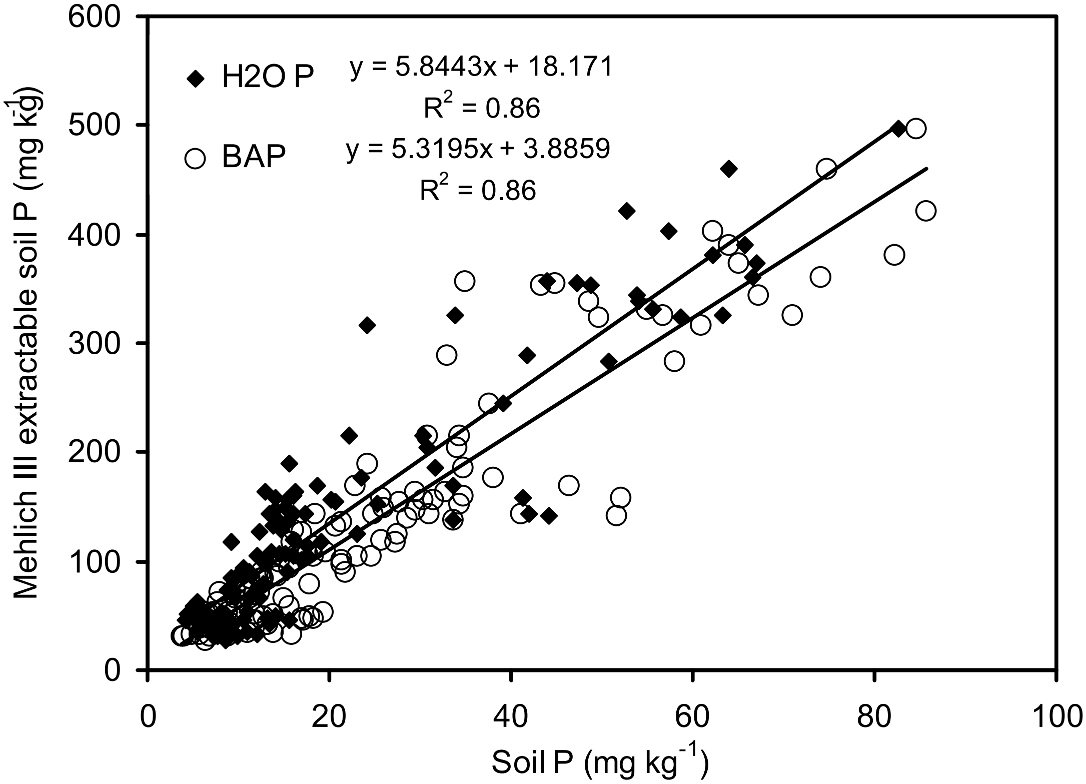

Figure 2.2. The relationship between soil P extracted by DI water and Fe_2O_3 Paper and soil P extracted by Mehlich III.

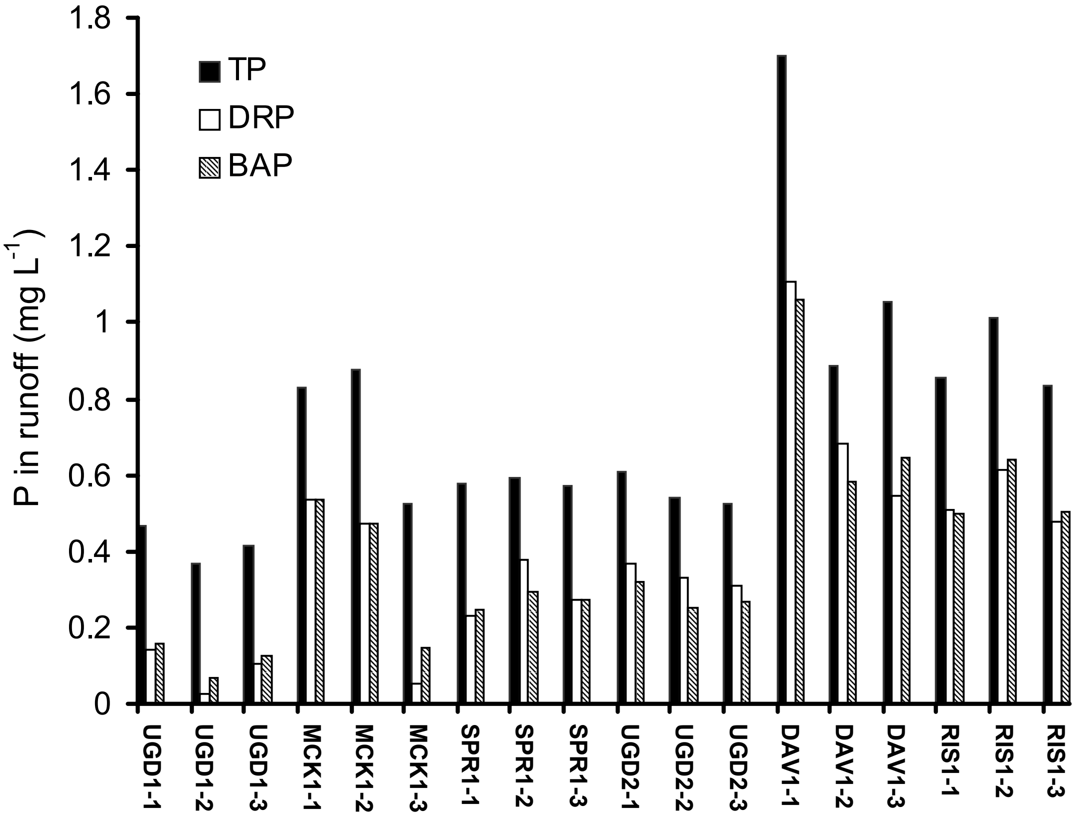

Figure 2.3. Flow weighted average total P (TP), dissolved reactive P (DRP), and bioavailable P (BAP) in runoff from each site.

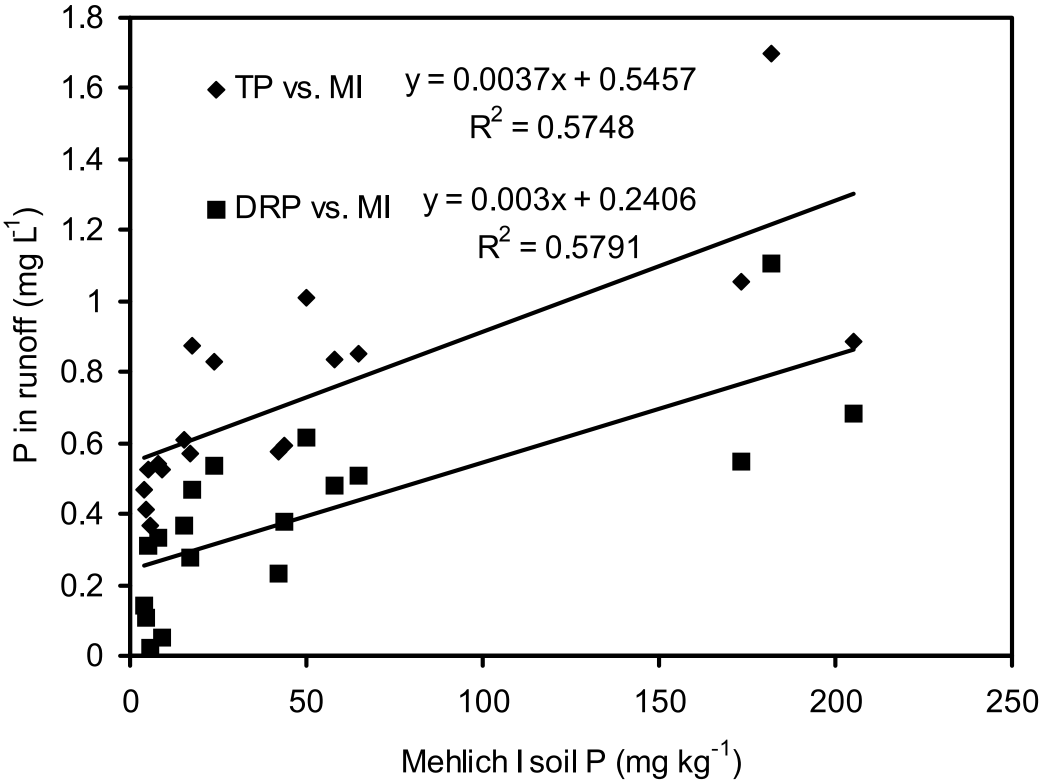

Figure 2.4. The relationship between Mehlich I extractable P and both TP and DRP in runoff.

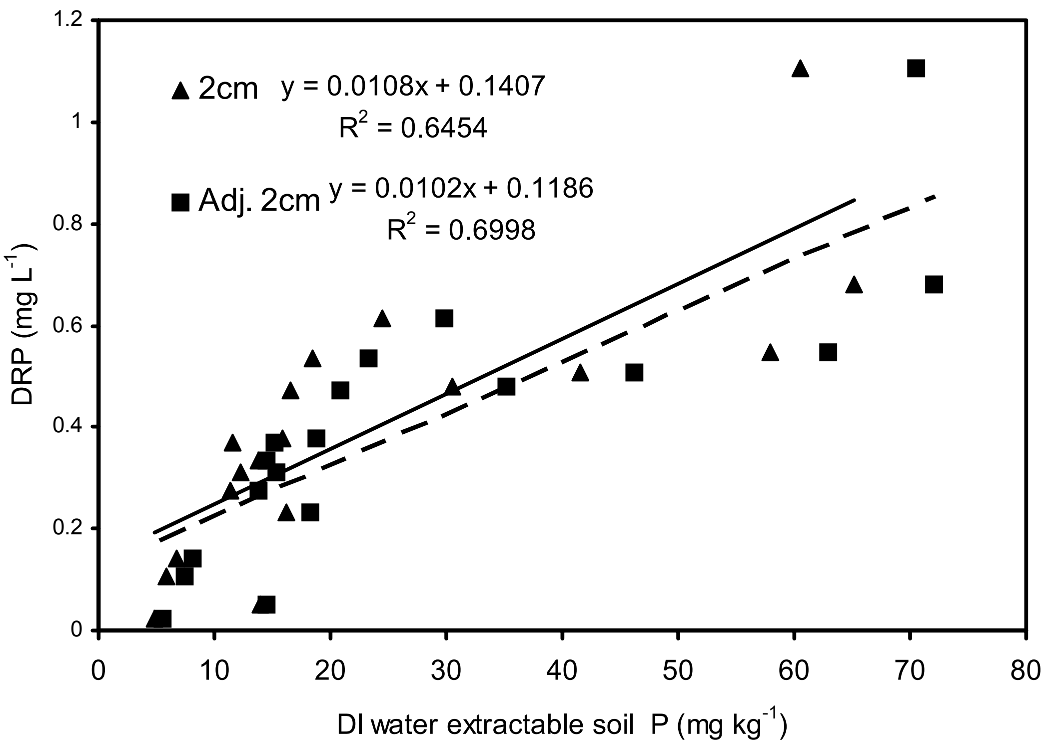

Figure 2.5.Comparison of the relationship between runoff DRP concentration (mg L^{-1}) and soil P extracted by DI water following the third rainfall vs. soil P extracted by water corrected for soluble P lost to runoff and infiltration.

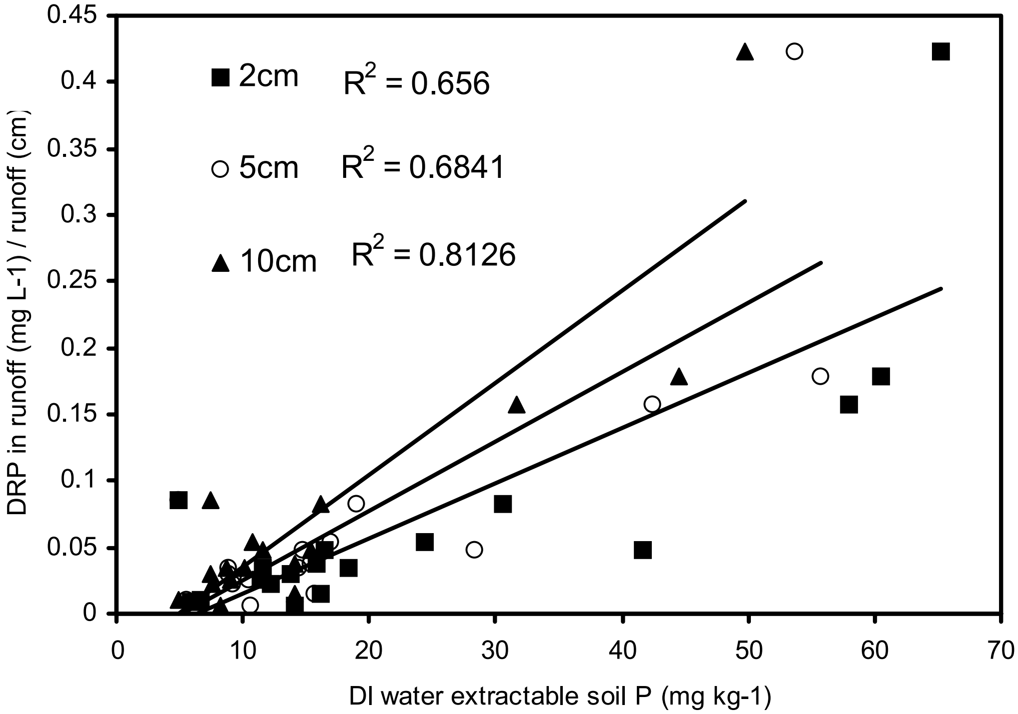

Figure 2.6. The relationship between soil P extracted by DI water and normalized (concentration divided by runoff depth) dissolved reactive P (DRP) in runoff.

CHAPTER 3

EFFECT OF RAINFALL TIMING AND APPLICATION RATE ON PHOSPHORUS LOSS FROM SURFACE APPLIED POULTRY LITTER[1]

[1] P. D. Schroeder, D. E. Radcliffe, M. L. Cabrera and C. D. Belew. To be submitted to J. Environ. Qual.

ABSTRACT

Phosphorus (P) in runoff from poultry litter may be a significant contributor to eutrophication of lakes and streams in GA and other areas in the South Eastern United States. The objectives of this research were to determine the effects of application rate and initial runoff timing on the long-term loss of P in runoff from surface applied poultry litter. Manure application rates were 2, 7, and 13 Mg ha^{-1}. Rainfall scenarios included; 1) sufficient rainfall to produce 30 min. of runoff immediately after litter application (R1); 2) no rainfall for 30 days after manure application, then sufficient rainfall to produce 30 min. of runoff (R2); and 3) small rainfall events every 7 d (5 min. at 75 mm hr^{-1}) for 30 days and then sufficient rainfall to produce 30 min. of runoff (R3). These three litter rates and rainfall scenarios were applied to 1 x 2 m plots in a 3 x 3 randomized complete block design with three replications. P loss was the greatest from the high application rate (13 Mg ha^{-1}) and from the immediate runoff (R1) plots. After initial runoff events, runoff was collected every 2 weeks for 6 months. Non-linear regression procedures were used to develop equations relating to P loss over time. The resulting equations produced fairly accurate prediction of P concentration (0.68 to 0.91 R^2) under the conditions in this study. Further analysis of the ability of these equations to predict P loss from more larger scale plots will be the subject of further research.

INTRODUCTION

Over the past decade, controlling non-point source pollution has come to the forefront in efforts to improve water quality in the United States and elsewhere. The principal components of agricultural non-point source pollution are sediment, bacteria, nitrogen (N), and phosphorus (P). Of these, P is the element most commonly associated with eutrophication in freshwater systems because these systems are usually P limited (Correll, 1998).

For economic reasons, it is unlikely that a soil's fixation capacity will be exceeded when the P source is chemical fertilizer. However, in areas of concentrated livestock production, land application of large volumes of manure, enriched with imported nutrients, is commonplace. Typically, animal manures have approximately equal amounts of available N and P, but crops generally require much more N than P. Application rates designed to meet a crop's N needs may, therefore, result in over-application of P.

The State of Georgia is one of the top broiler (*Gallus gallus domesticus*) producing regions of the United States with 1.2 billion broilers raised in 1998 (National Agricultural Statistics Service, 1999). Besides broiler production, agriculture in Central and North Georgia is generally limited to the production of beef cattle and its associated pastures and hayfields. Many farms in this region combine cattle and broiler production.

Forms of P in runoff can include dissolved P (DP) in both organic and inorganic forms as well as particulate P (PP) associated with mineral or organic particles transported in the runoff. Because erosion rates from hayfields and pastures are low, DP is usually the dominant form of phosphorus in hayfield and pasture runoff (Sharpley et al., 1992). Dissolved P in runoff results from the interaction of rainwater with a thin layer of the soil's surface. The effective depth of this interaction is dependent on soil aggregation, percent ground cover, slope, and rainfall intensity (Sharpley, 1985b). In two studies on effective depth of interaction, this depth ranged from 0.2 cm to more than 3.7

cm depending not only on the factors mentioned above but also on the experimental method used (Sharpley et al., 1981, Sharpley, 1985).

High application rates of animal manures typically increase the concentration of P in runoff. Total P concentrations were 4.5 mg L^{-1} higher in runoff from bermudagrass plots receiving poultry litter at a rate of 11 Mg ha^{-1} than from unfertilized control plots (Heathman et al., 1995). A positive relationship exists between the rate of manure application and the concentration of TP, DP and PP in runoff (McLeod and Hegg, 1984; Mueler et al., 1984; Edwards and Daniel, 1994; Vervoort et al., 1998; Wood et al., 1999). Additionally, there are currently no federal or state standards for P levels in surface waters. However, the US EPA has proposed ambient water quality criteria for Ecoregion IV of 0.037 mg P L^{-1} for rivers and streams and 0.02 mg p L^{-1} for lakes

The majority of the published research shows that most of the P is lost in the first runoff event from fields where manure has been surface applied (Edwards and Daniel, 1994; Sauer et al., 1999; Sharpley, 1997). However, the relationship between P loss in runoff from fields treated with animal manure, and storm interval and intensity is not clearly understood. Several researchers have shown after an initial spike, P levels in runoff remain above background levels for up to 18 months. Sharpley (1997) incubated 10 Oklahoman soils with poultry litter for 1 to 35 days prior to application of simulated rainfall, and found that although the P level in runoff declined through 10 consecutive rainfall events, it was still significantly above the P level in runoff from an unfertilized control plot. Similarly, in a study of surface applied poultry litter with and without tillage, Heathman et al. (1995) observed that both total P and soluble P concentrations in runoff were higher from plots receiving poultry litter than from an unfertilized control plot even after 10 rainfall events. In a multiple year study, Pierson et al. (2001) found increased P in runoff for up to 18 months after an application of poultry litter to a small watershed in Georgia.

In contrast, some researchers report P levels in runoff returning to background levels after just a few rainfall events. For example, Sauer et al. (1999) report that soluble reactive P concentration in runoff from poultry litter amended plots dropped from 13.5 mg L^{-1} in the first runoff event to 1.2 mg L^{-1} in the second runoff event. Similarly, Edwards and Daniel (1994) found that levels of DRP and TP in runoff from plots receiving poultry litter did not significantly differ from control plots after two rainfall events.

These variable results may be due to differences in soil P sorption capacities and the fertility histories of the studied sites, in that there are significant differences in the levels of soluble P running off of the unfertilized controls. For example, soluble P in runoff from controls reported by Heathman et al. (1995) were an order of magnitude lower than those reported by Sauer et al. (1999). The plots used by both Sauer et al. (1999) and Edwards and Daniel (1994) had Mehlich III extractable P levels ranging from 100 to 223 mg kg^{-1}, (Sauer et al., 1999), whereas Sharpley (1997) and Heathman et al. (1995) reported Mehlich III extractable P levels of only 7 to 39 mg kg^{-1}. The higher STP levels may produce higher "background" levels of P in runoff, which in turn may make it impossible to detect the long-term effects of manure application.

Because most of the P lost from manure applications is lost in the first runoff event, several researchers investigated the effect of time between manure application and a runoff-producing rainfall on P loss. Conceptually, the P in animal manure should become less susceptible to loss in surface runoff over time due to the adsorption and fixation of P by soil surfaces. Delay times to first runoff event of one hour to three days after manure application were studied by Westerman and Overcash (1980). They reported a 90% reduction in P concentration in runoff from fescue plots treated with liquid poultry manure for three days of delay. Similarly, soluble P concentrations in runoff from soils mixed with poultry litter were reduced from 0.74 to 0.45 mg L^{-1} when incubation time was increased from 1 to 35 days (Sharpley, 1997). Both of these studies conclude that because time to first runoff event reduces P losses, producers should plan manure applications at times when runoff producing rainfall events were less likely. It should be noted however, that these two studies did not address the issue of surface-applied dry manure because Westerman and Overcash spread liquid manure and Sharpley incorporated poultry litter in the soil. These distinctions may have contributed to their conclusions, because in both situations the manure comes into direct contact with the soil and this direct contact may have facilitated the adsorption of soluble P by the soil. In contrast, when dry poultry litter is surface applied to pastures and hayfields, a surface layer of thatch is likely to prevent direct contact between the litter and the soil and reduce the possibility that P in the manure will be adsorbed immediately by the soil. To address the effects of interval to first runoff event on P runoff from fescue plots treated with dry poultry litter, Edwards et al. (1994a) observed that intervals of 4, 7, and 14 days between manure application and rainfall had no effect on either concentration or mass of ortho-P or TP in runoff. In a related study, time to first runoff event had no effect on the transport of manure particles from poultry litter treated sites (Edwards et al., 1994b). In a more recent study, Pierson et al. (2001) observed that DRP loss from pastures fertilized with broiler litter was strongly affected by interval to first runoff event. In this study broiler litter was applied to 0.75 ha pastures and runoff was collected from natural rainfall events. Over the course of the study, drying intervals of less than 12 days and more than 7 months to the first runoff event occurred. Dissolved reactive P (DRP) concentrations in the first runoff event were < 5 mg L^{-1} for the 7 month interval but were > 5 mg L^{-1} when events occurred within 12 days of application.

From the preceding discussion, it is apparent that there is some debate over the relative importance of the various sources of P to the actual concentration of P in runoff. Additionally, the impact of these sources of P on sensitive water bodies will be affected by parameters that control runoff volume. To address these issues and develop a method for assessing the overall risk of P loss from individual sites, the P Index was created (Lemunyon and Gilbert, 1993). The original P Index was an 8 x 5 matrix of landform site characteristics including erosion, runoff class, STP, fertilizer application rate, and manure application rate. Each of these factors was assigned a rating from 0 to 8 depending on the risk level of that factor, 0 being no risk and 8 being very high risk. Once each characteristic was assigned a value, that value was multiplied by a "weighting factor". The weighted values were then summed to arrive at the P Index rating. One problem with this approach has been the assignment of the weighting factors. These factors were based on the "professional judgment" of the scientists involved.

Another approach to the P Index has been taken in Georgia, Iowa, and Virginia, where the P index is an estimate of the mean annual soluble, particulate, and leached P load loss from a site (Cabrera et al., 2002, and G.Mullins and A. Mollinari, personal communication, 2001). Whether the P index is based on weighting factors or estimates of mean annual P load, most approaches consider three sources for P in runoff: a source related to the P pool in the soil (STP), a source related to the P pool associated with inorganic fertilizer P applications, and a source related to the P pool associated with organic (manure) P applications. The relative importance of these pools is expressed through the weighting factors or through the estimate of relative P load losses. Since the timing of the contribution from the manure pool is poorly understood in surface applications to grasslands, determining the relative weight or the estimated P load loss for this pool presented difficulties in P index development.

The objective of this research was to determine the effects of application rate and initial runoff timing on the long-term loss of P in runoff from surface applied poultry litter.

MATERIALS and METHODS

Experimental Design

To determine the effect of manure application rate and initial runoff timing, we chose three poultry litter application rates and three runoff scenarios. Manure application rates were 2, 7, and 13 Mg ha^{-1} (equivalent to 1, 3, and 6 tons per acre), typical application rates in North Georgia. Rainfall scenarios included; 1) sufficient rainfall to produce 30 min. of runoff immediately after litter application (R1); 2) no rainfall for 30 days after manure application, then sufficient rainfall to produce 30 min. of runoff (R2); and 3) small rainfall events every 7 d (5 min. at 75 mm hr^{-1}) for 30 days and then sufficient rainfall to produce 30 min. of runoff (R3). These scenarios were chosen based on the contradictory nature of previous research related to time to first runoff event. The third scenario is unlike any previously reported and we believe it to be the most realistic.

Based on almost 30 years of weather data from the Athens, GA area, the probability that 25.4 mm of rain will fall on any single day is approximately 7 % and the probability of receiving 50 mm of rain on any given day is less than 4%. However, the probability that 6 mm of rain will fall on a given day is about 18 %. Based on these probabilities it seems more realistic to expect that after a manure application some small rainfall events will occur before an event large enough to produce runoff.

These three litter and rainfall scenarios were applied to 1 x 2 m plots in a 3 x 3 randomized complete block design with three replications. In addition to these treatments, three control plots were included (one for each block) to allow for correction of P loss not associated with litter application. Therefore the experiment consisted of thirty plots (3 x 3 x 3 + 3 = 30).

Following the implementation of the initial rainfall scenarios, all plots received sufficient rainfall to produce 30 min. of runoff on a biweekly basis for five months from May through October 2001. Control plots were subjected to biweekly runoff events over the same five-month period under the first rainfall scenario (R1) described above. Thanks to several days of unseasonably warm weather in January of 2002, we had the opportunity to conduct additional rainfall simulations on the 13 Mg and control plots.

This experiment was installed on a hayfield with a fairly uniform 8 % slope at the University of Georgia Plant Sciences Farm near Athens GA in the early spring of 2001. The soil in the study area is a fine, kaolinitic, thermic Typic Kanhapludult that is mapped as the Cecil series. In late February 2001, Kentucky 31 tall fescue was planted on the site to supplement the relatively thick stand of fescue that was present. At that time 14-3-12 starter fertilizer (as N, P, K) was applied at a rate of 448 kg ha^{-1}. Approximately 48.5 mm of natural rain fell on the experimental area between March 1st and May 14th, the date litter treatments were applied.

From March 3rd to May 1st, thirty 1 x 2-m plots were installed at the site in three blocks of ten plots. The blocks were orientated so that the long axis of all plots was aligned parallel with the slope. Plot borders consisting of approximately 0.3-cm thick sheet metal 15-cm tall were pressed into the ground to a depth of at least 7 cm to isolate runoff. Aluminum flumes were installed at the down-slope edge of each plot to divert surface runoff to a collection point. Poultry litter was applied by hand on the 14th, 15th and 16th of May to plots in blocks one, two, and three respectively. The 2, 7 and 13 Mg ha^{-1} litter rates resulted in the application of 69.7, 20.90, and 41.79, g P per plot.

Simulated rainfall was applied to each plot at a rate of 75 mm hr^{-1} with a JOERNS INC. (West Lafayette, Indiana) rainfall simulator. Collection of runoff from the field plots began after steady runoff commenced and continued for 30 minutes. Runoff was collected *in toto* and a 500-mL composite sample was taken and immediately placed on ice. Total runoff volume was recorded and a source water sample taken. Clear polymer covers were placed over each plot between simulations to prevent the plots from receiving natural rainfall. These covers consisted of 6 mil polyethylene stretched over wooden frames that were pitched to direct rainwater away from the plots. Wooden blocks 10 cm tall were used to support the covers above the plots. This resulted in a gap of at least 10 cm at each end of the plot. Paired thermocouples (in and out of the plot) installed in five plots showed that the soil temperature under the covers differed from ambient soil temperature by less that 0.5°C throughout the experiment. Soil moisture content in the plots and adjacent areas was monitored with time domain reflectometry (TDR). One 30-

cm TDR probe was installed in each plot. The probes were installed at an angle sufficient to ensure that the moisture content was measured over the upper 10 cm of the soil. All plots were mowed to a height of approximately 10 cm every two weeks for the duration of the experiment, and clippings were removed to prevent the loss of P to runoff from decaying grass. Local well water was used as the water source. After the last rainfall event, soil samples (composites of ten random samples) were collected to a depth 0-10 cm from within each plot.

Litter Sampling

The litter used in this study was collected on May 10, 2001 from a broiler farm in North Georgia. A 300 kg composite sample consisting of thirty sub-samples was collected from a single broiler house and transported to the University of Georgia Plant Sciences Farm where it was mixed in a commercial mortar mixer until most of the cake was broken up (approximately 15 min.). From this composite sample, treatment samples were weighed and a sub-sample was collected for total P (TP) analysis.

Sample Analysis

Soil samples were air-dried and ground to pass a 2-mm sieve. Soil pH was determined in a 1:2 soil/water mixture using a glass electrode. Extractable P in each plot soil sample was determined by two methods, Mehlich III (Mehlich, 1984) and DI water extraction (Pote et al., 1996). Total P of unfiltered runoff samples was determined colorimetrically following Kjeldahl digestion (Murphy and Riley, 1962). In the lab, 125 mL of each runoff sample was filtered (0.45-µm pore diameter) to remove particulate matter and stored at -20°C until analyzed. The DRP content of filtered runoff samples was also determined colorimetrically.

Total P content of the poultry litter determined colorimetrically following Kjeldahl digestion was 23.98 g kg^{-1}. Water-soluble P in poultry manure determined colorimetrically after shaking 20 g of manure in 4 L of DI water for 4 hours was 6.2 g kg^{-1}. The pH of the litter was 8.46 as determined in a 1:5 litter/water mixture using a glass electrode. Litter moisture content was determined to be 542 g kg^{-1}, after oven drying at 65^{o} C for 48 hours.

Data Analysis

Total and soluble P mass lost from each plot was calculated for each runoff event based on P concentration and runoff volume. Overall total P loss and cumulative P loss by runoff event were calculated. Average TP and DRP loss from control plots was subtracted from treatment plot P loss so that only P loss associated with litter application would be analyzed. Statistical analyses were performed using the Statistical Analysis System (SAS Institute, 1987). Analysis of variance techniques were used to determine treatment effects and to check for interaction across all ten runoff events. Treatment effects were also studied with ANOVA for runoff events individually. The method of least significant difference was used to analyze treatment means. Non-linear regression was used to develop predictive equations relating P loss in runoff from surface applied poultry litter to P application rate, runoff depth, cumulative rainfall, days since manure application, antecedent soil water content, and temperature. Regression analysis was also employed to determine if soil test P (STP) levels were related to P application rate, runoff depth, cumulative rainfall, or pH.

The principle of conditional error (Bose, 1949; Milliken and Johnson, 1984) was used to evaluate regression equations relating P loss to litter application, runoff and days since litter application. This is a technique for obtaining the sum of squares due to deviations from a hypothesis for linear models. The null hypothesis was that one equation could be used to describe TP or DRP loss as a function of litter application, runoff and days since litter application for all three rainfall treatments. This procedure provided an estimate of the residual sum of squares for the null hypothesis. Secondly, the alternative hypothesis that a separate equation was needed for each relationship was tested. The summation of the residual sum of squares for each separate equation provided an estimate of the residual sum of squares for the alternative hypothesis. The difference between the residual sum of squares of the null and alternative hypotheses provided an estimate of the residual sum of squares due to deviations from the null hypothesis. This residual sum of squares was then used in an F test against the residual sum of squares of the alternative hypothesis to determine if the relationship between rainfall treatments were significantly different.

RESULTS AND DISCUSSION

Rainfall and Litter treatment effects

Analysis of variance for TP and DRP in runoff across all events revealed significant interaction between both rainfall and litter treatments and runoff events so each event was analyzed as an independent experiment to assess treatment effects. For each runoff event the litter application rate and the rainfall treatments were significant ($p<0.0001$), but there was no significant interaction between them.

The effects of the three rainfall scenarios on runoff TP and DRP loss can be seen in Table 2.1. Phosphorus loss, both TP and DRP, was greatest from the R1 (immediate runoff producing rainfall) treatment. The R2 and R3 treatments did not produce significantly different cumulative TP or DRP losses ($p<0.05$). The fact that the most realistic initial rainfall treatment (R3) produced less cumulative TP and DRP loss (45% and 40 %, respectively) than R1 suggests that under "real world" conditions P losses from surface applied manure may be considerably less than the "worst case " scenario represented by the R1 treatment. The rainfall effect was most pronounced in the first runoff event in which the R1 treatment showed both the greatest TP and DRP loss (4.41, and 3.22 kg ha^{-1}, respectively), and the highest percentage P loss (59.6%, and 57.3%, respectively). The R3 treatment produced the smallest TP and DRP loss as well as the smallest % TP and DRP losses in the first event. Over the remainder of the events there was little difference in P loss or percentage P loss among the rainfall treatments, except that the R3 treatment produced higher % P loss for the 3rd and 4th events. This may indicate that although initial and overall P loss was less from the R3 treatment the P loss was more evenly distributed across runoff events.

The fact that cumulative P loss was not different between the R2 and R3 treatments, for any of the ten events, indicates that the application of small amounts of rain probably had two effects. First, the rainfall probably transported some of the manure P into the soil where it was adsorbed, leading to reduced P in runoff. The second effect of adding water may have been to increase mineralization of organic P thereby negating some of the effects associated with the adsorption of inorganic P.

As expected, the 13 Mg ha^{-1} litter treatments produced much higher cumulative TP and DRP loss than the 2 or 7 Mg ha^{-1} treatments ($p<0.05$). The highest litter application rate produced the highest TP and DRP loss for all individual runoff events (Table 2.2). The first runoff event showed the most dramatic differences among the three litter application rates with the 13, 7 and 2 Mg ha^{-1} rates producing TP loses of 4.75, 2.38 and 1.26 kg ha^{-1}, and DRP losses of 3.59, 2.03, and 0.72 kg ha^{-1}, respectively. By the second runoff event P losses from all three treatments decreased considerably, to 1.35, 0.45, and 0.24 kg ha^{-1} TP, and 1.04, 0.39, and 0.28 kg ha^{-1} DRP. For the second and later events TP and DRP losses from the 7 and 2 Mg ha^{-1} treatments were not significantly different.

The average TP and DRP loss in runoff from the L3 (13 Mg ha^{-1}) treatments for the simulations conducted in January 2002 were 0.285 and 0.241 kg ha^{-1} respectively. These values were somewhat higher than those seen in the tenth rainfall event when TP and DRP losses of 0.18 and 0.15 kg ha^{-1} were observed for the L3 treatment. The increase in P loss in these final runoff events is attributed to higher runoff volume due to wetter antecedent conditions. The L3 treatment was still producing relatively high levels of both TP and DRP, indicates that a single surface application of poultry litter can produce elevated runoff P concentrations for more than 10 months. In fact, Pierson et al. (2001) observed DRP concentrations in excess of 1 mg L^{-1} for over 18 months following 4 years of poultry litter application.

Although STP increased threefold for some treatments (Table 2.3), the fact that after 10 runoff events only 11.2, 6.2, and 5.7 % of applied P was lost from the 2, 7 and 13 Mg ha^{-1} litter treatments indicates that a significant portion of the applied P remained on the surface. This residual surface P will continue to mineralize over time and produce elevated P levels in runoff for a significant period as reported by Pierson et al. (2001).

The results of this experiment both affirm and contradict previous research on P loss from poultry litter application. It is quite clear from our research that a 30-day delay interval between litter application and runoff significantly reduced both the initial TP and DRP loss as well as the long-term P loss when compared to runoff immediately following manure application. These results appear to agree with results published by Westerman and Overcash (1980) and Sharpley (1997). These two studies differ from the present study in that they either used liquid manures (Westerman and Overcash) or incorporated the manure into the soil (Sharpley). These two differences would likely amplify the delay effect because of the close contact between soil and manure inherent in the studies. Contradictory findings were reported by Edwards et al. (1994), who concluded that delay intervals of 4, 7 and 14 days, did not affect TP and DRP loss in runoff from poultry litter surface applied to fescue plots. They reasoned that delay interval did not affect P loss because the grass cover limited contact between the litter and the soil, i.e. conditions were not optimal for soil fixation of litter P. The differences between the Edwards et al. and the present study are likely due to several factors including the longer delay interval (14 vs. 30 days), lower maximum litter application rate (5.6 vs. 13 Mg ha^{-1}), and the fact that Edwards et al. did not include an immediate runoff treatment in their study. It is possible that the delay effect occurs in a short period of time due to rapid fixation of P and without an immediate runoff treatment this effect was not seen. However, it seems that this is unlikely due to the restricted contact between soil and litter caused by the grass cover. It is possible that the delay effect seen in the present study was due to the settling of manure and movement of manure by soil organisms over time and subsequent increase in soil contact; also the heavy daily dewfall and high humidity common in this area may have aided in the movement of soluble P into the soil.

Soil Phosphorus Levels

After the final rainfall event, soil samples were collected from the upper 10 cm of each plot (Table 2.3). Soil pH ranged from 5.63 to 5.92, DI water extractable P ranged from 0.3. to 6.8 kg ha^{-1}, and Mehlich III extractable P ranged from 21.8 to 65.9 kg ha^{-1}. Stepwise multiple regression was performed in order to determine the most important factors controlling soil test P (STP) change due to poultry litter application. Soil test P was regressed against manure P application rate, pH, cumulative rainfall, and runoff depth. Mehlich III STP was related to P application rate and soil pH (R^2 = 0.60) by the following equation:

$$Mehlich\ III = -112.12 + 23.55\ (pH) + 0.17\ (Litter\ TP)$$

where litter TP is the P applied in litter (kg ha^{-1}). Due to the relatively small amount of P extracted by DI water (mean = 2.80 kg ha^{-1}) and high degree of variability (STDEV = 3.10) across the treatments, no significant relationships were found between DI water extractable P and any of the regressors used.

Predicting P Loss

Since there have been few attempts to model the contribution to P loss in runoff from P in surface-applied manure, we tried to develop equations to predict P loss from surface-applied poultry litter over time. We attempted to fit equations of several different forms, both linear and non-linear, using multiple and non-linear regression procedures in SAS. Of all the equations that we attempted to fit to the data, the following decay function was found to give the best fit.

$$P(t) = (P_0 \bullet a)\ e^{-kt}$$

Where $P(t)$ is the concentration of P lost (mg L^{-1}) at time t, and P_0 is the litter P application rate (kg ha^{-1}). a and k are constants related to the maximum P concentration predicted and the effect of time, respectively. This equation was fitted to data collected from all ten runoff events. The principle of conditional error, (Bose, 1949; Milliken and Johnson, 1984) was used to determine if one equation would adequately describe the relationship between P loss and litter P application rate, days since litter application and runoff depth in mm, for all rain treatments. This analysis indicated that there were significant differences ($p< 0.05$) between the three rainfall treatments and that separate equations were needed to model the data. Through the fitting process, attempts were made to include factors such as rainfall, runoff, temperature, soil moisture, and days since litter application, in the equation. Of all these factors, inclusion of time since application alone produced the best prediction. As a result the following six equations were developed:

R1

$$TP(t) = (P_0 \bullet 0.231)\ e^{-0.041t}$$

$$DRP(t) = (P_0 \bullet 0.118)\ e^{-0.042t}$$

R2

$$TP(t) = (P_0 \bullet 0.343)\ e^{-0.045t}$$

$$DRP(t) = (P_0 \bullet 0.266)\ e^{-0.043t}$$

R3

$$TP(t) = (P_0 \bullet 0.116)\ e^{-0.024t}$$

$$DRP(t) = (P_0 \bullet 0.080)\ e^{-0.021t}$$

Where P_0 is the amount of P (kg ha^{-1}) applied in litter, t is the number of days since litter application.

In these equations the first term, $P_0 \bullet a$, represents the maximum P concentration in runoff when t is equal to zero. The value of a in the equations for the R3 treatment is about half the value of a in the equations for the R1 treatments. This reflects the combined effect of the delay in runoff and the application of small rainfall events prior to runoff. However, the value of a in the equations for R2 is much higher than the value of a in either the R1 or R3 equations. Throughout the fitting process, the R2 data set produced nonsensical equations and efforts to find erroneous data points or sets of points proved unsuccessful resulting in a values for the R2 equations that are excessively large.

The constants related to the number of days since litter applications were similar for the R1 and R2 equations. However the k value for the R3 equations were about half that of the R1 and R2 equations. The smaller k values for the R3 equations may represent the effect of the small rainfall events applied between litter application and the initial runoff event. This effect may be due to immobilization of P in the litter over time through biological uptake and or fixation on the surface of soil organic matter and Fe and Al oxides. Consequently, in soils with high STP levels this effect may be less apparent since most P fixing sites would already be occupied.

Figures 2.1 through 2.4 show the decay equation for TP and DRP developed for the R1 and R3 scenarios. The best fit was obtained for the R1 equations followed by the R3 equations (Table 2.4). The R2 equations were the least accurate predictors of P concentration. Because of the problems associated with fitting the R2 equations and because they were the least accurate they may be of limited use. However, the R2 rainfall treatment is also the least realistic of the three employed in this study.

The R1 and R3 equations should allow the prediction of long-term P loss from surface application of poultry litter, under conditions similar to those of this study. These equations may also provide a more scientifically based assessment of the risk of P loss associated with surface application of poultry litter. These types of equations could be included in a load based P index such as the one developed in Georgia.

CONCLUSIONS

The rainfall treatment where runoff-producing rainfall was applied immediately following litter application produced significantly greater TP and DRP loss ($p < 0.05$) for the first runoff event. It appears quite clear that rainfall timing (i.e. time to first runoff event) and litter application rate have a dramatic effect on P loss in runoff. Additionally, after 10 runoff events P losses in runoff were still significant and resulted in P concentrations exceeding 1 mg L^{-1}, the value that has been proposed as the maximum desirable P concentration in agricultural runoff (USEPA, 1986). Total mass of P lost from all ten runoff events represented 5 to 11 % of the P applied indicating that a significant pool of litter P remained after ten events. Most of this P pool may be organic P that remains on the soil surface rather than inorganic P adsorbed to the soil because only modest increases were seen in STP levels. Additionally, an equation relating litter P application rate to changes in Mehlich III STP was developed. About 60% of the variability in Mehlich III STP was explained by this equation

Non-linear regression equations that may be useful in predicting P losses in runoff from surface applied poultry litter were developed. These equations were able to explain

68 to 91 % of the variability seen in P losses. A computer model will be developed, based on these equations and previous work by the authors relating STP levels to P in runoff, to predict P loss from poultry litter applied to pastures and hayfields. Such a model would improve the ability to assess the vulnerability of a site to P loss from surface application of poultry litter and may be helpful in designing comprehensive nutrient management plans. Because the levels of P loss seemed to rebound after several months as reflected in the runoff from the January 2002 rainfall simulations, future research in this area should be directed at determining the effect of the interval between runoff events on P loss.

REFERENCES

Cabrera, M.L., D.H. Franklin, G.H. harris, V.H. Jones, H.A. Kuykendall, D.E. Radcliffe, L.M. Risse, and C.C Truman. 2002. The Georgia Phosphorus Index. (Available on-line with updates at http://www.ga.nrcs.usda.gov/gatechnical/afo/ga-p-index-v12bsdl.doc)

Correl, D.L. 1998. The role of phosphorus in the eutrophication of receiving waters: A review. J.Environ. Qual. 28:261-266.

Edwards, D.R., and T.C. Daniel. 1994. Quality of runoff from fescuegrass plots treated with poultry litter and inorganic fertilizer. J. Environ. Qual. 23:579-584.

Edwards, D.R., and T.C. Daniel, P.A. Moore, Jr., and P.F. Vendrell. 1994a. Drying interval effects on quality of runoff from fescue plots treated with poultry litter. Trans. ASAE. 37:831-843.

Edwards, D.R., and T.C. Daniel, P.A. Moore, Jr., and A.N. Sharpley. 1994b. Solids transport and erodibility of poultry litter surface-applied to fescue. Trans. ASAE. 37:771-776.

Gburek, W.J., A.N. Sharpley, L. Heathwaite, and G.J Folmar. 2000. Phosphorus management at the watershed scale: A modification of the phosphorus index. J. Environ. Qual. 29:130-144.

Heathman, G.C., A.N. Sharpley, S.J. Smith, and J.S. Robinson. 1995. Land application of poultry litter and water quality in Oklahoma, U.S.A. Fert. Res. 40:165-173.

Lemunyon, J.L. and R.G. Gilbert. 1993. The concept and need for a phosphorus loss assessment tool. J. Prod. Agric. 6:483-486.

McLeod, R.V. and R.O. Hegg. 1984. Pasture runoff water quality from application of inorganic and organic nitrogen sources. J. Environ. Qual. 13:122-126.

Mehlich, A. 1984. Mehlich III soil test extractant: A modification of Mehlich II extractant. Commun. Soil Sci. Plant Anal. 15:1409-1416.

Meuler, D.H., R.C. Wendt, and T.C. Daniel. 1984. Phosphorus losses as affected by tillage and manure application. Soil Sci. Soc. Am. J. 48:901-905.

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. 1999. USDA-NASS Agricultural Statistics 1999. U.S. Department of Agriculture Washington, D.C.

Pierson, S.T., M.L. Cabrera, G.K. Evanylo, H.A. Kuyrkendall, C.S. Hoveland, M.A. McCann, and L.T. West. 2001. Phosphorus and ammonium concentrations in surface runoff from grasslands fertilized with broiler litter. J. Environ. Qual. 30:1784-1789.

Pote, D.H., T.C. Daniel, D.J. Nichols, A.N. Sharpley, P. A, Moore, Jr., D.M. Miller, and D.R. Edwards. 1999. Relationship between phosphorus levels in three ultisols and phosphorus concentrations in runoff. J. Environ. Qual. 28:170-175.

Sauer T.J., T.C. Daniel, P.A. Moore, Jr., K.P. Coffey, D.J. Nichols, and C.P. West. 1999. Poultry litter and grazing animal effects on runoff water quality. J. Environ. Qual. 28:860-865.

Sharpley, A.N. 1981. The depth of rainfall-runoff-soil interaction as determined by ^{32}P. Water Res. Res. 17:969-974.

Sharpley, A.N. 1985a. Phosphorus cycling in unfertilized and fertilized agricultural soils. Soil Sci. Soc. Am. J. 49:905-911.

Sharpley, A.N. 1985b. Depth of surface soil-runoff interaction as affected by rainfall, soil slope, and management. Soil Sci. Soc. Am. J. 49:1010-1015.

Sharpley, A.N. 1997. Rainfall frequency and nitrogen and phosphorus runoff from soil amended with poultry litter. J. Environ. Qual. 26:1127-1132.

Sharpley, A.N., S.J. Smith, O.R. Jones, W.A. Berg, and G.A. Coleman. 1992. The transport of bioavailable phosphorus in agricultural runoff. J. Environ. Qual. 21:30-35.

USEPA 2000. Ambient water quality criteria recommendations. EPA 822-B-00-019.

Vervoort, R.W., D.E. Radcliffe, M.L. Cabrera, and M. Latimore, Jr. 1998. Field scale nitrogen and phosphorus losses from hayfields receiving fresh and composted broiler litter. J. Environ. Qual. 27:1246-1254.

Westerman, P.W. and M.R. Overcash. 1980. Short-term attenuation of runoff pollution potential for land applied swine and poultry manure. p. 289-292. *In* R.J. Smith (ed) Livestock waste: A renewable resource. Proc. 4th International Symp. On Livestock Wastes. Amarillo, Tx, 15-17 Apr. ASAE, St. Joseph, MI.

Wood, B.H., C.W. Wood, K.H. Yoo, K.S. Yoon, and D.P. Delaney. 1999. Seasonal surface runoff losses of nutrients and metals from soils fertilized with broiler litter and commercial fertilizer. J. Enviorn. Qual. 28:1210-1218.

List of Tables and Figures

Table 3.1. Total and dissolved P lost from each of ten runoff events and % of total lost in each event for the three rainfall treatments averaged across litter application rates. The column headings R1, R2, and R3 refer to immediate runoff, 30 days no runoff, and small rainfalls before runoff, respectively. Event refers to biweekly 30-minute runoff events.

	R1		R2		R3	
Event	TP (kg ha^{-1})	% Total	TP (kg ha^{-1})	% Total	TP (kg ha^{-1})	% Total
1	4.41	59.59	2.37	51.52	1.61	39.46
2	1.10	14.86	0.49	10.65	0.45	11.03
3	0.39	5.27	0.33	7.17	0.47	11.52
4	0.34	4.59	0.48	10.43	0.62	15.20
5	0.34	4.59	0.20	4.35	0.14	3.43
6	0.23	3.11	0.24	5.22	0.34	8.33
7	0.17	2.30	0.17	3.70	0.12	2.94
8	0.16	2.16	0.13	2.17	0.14	3.43
9	0.15	2.03	0.12	2.61	0.11	2.70
10	0.11	1.49	0.10	2.17	0.08	1.96
Total	7.40a†		4.63b		4.08b	
	DRP (kg ha^{-1})	% Total	DRP (kg ha^{-1})	% Total	DRP (kg ha^{-1})	% Total
1	3.22	57.30	1.84	48.94	1.28	37.87
2	0.85	15.12	0.44	11.70	0.41	12.13
3	0.28	4.98	0.28	7.45	0.38	11.24
4	0.26	4.63	0.42	11.17	0.52	15.38
5	0.29	5.16	0.11	2.93	0.11	3.25
6	0.18	3.20	0.21	5.59	0.32	9.47
7	0.15	2.67	0.15	3.99	0.12	3.55
8	0.14	2.49	0.13	3.46	0.09	2.66
9	0.14	2.49	0.11	2.93	0.10	2.96
10	0.11	1.96	0.07	1.86	0.05	1.48
Total	5.62a		3.76b		3.38b	

† Totals followed by the same letter are not different according to LSD ($p<0.05$).

Table 3.2. Total and dissolved P lost from each of ten runoff events and % of total lost in each event for the three litter treatments averaged across rainfall treatments. The column headings L1, L2, and L3 refer to litter application rates 3 7 and 13 Mg ha^{-1}, respectively. Event refers to biweekly 30-minute runoff events.

	L1		L2		L3	
Event	TP (kg ha^{-1})	% Total	TP (kg ha^{-1})	% Total	TP (kg ha^{-1})	% Total
1	1.26	61.17	2.38	56.40	4.75	49.17
2	0.28	11.65	0.45	10.66	1.35	13.98
3	0.13	6.31	0.27	6.40	0.79	8.18
4	0.22	10.68	0.41	9.72	0.81	8.39
5	0.05	2.43	0.16	3.79	0.48	4.97
6	0.08	3.40	0.24	5.69	0.49	5.07
7	0.03	1.46	0.01	0.31	0.29	3.00
8	0.02	0.49	0.10	2.37	0.28	2.90
9	0.03	1.46	0.11	2.61	0.24	2.48
10	0.02	0.97	0.09	2.13	0.18	1.86
Total	2.12c†		4.22b		9.66a	
	DRP (kg ha^{-1})	% Total	DRP (kg ha^{-1})	% Total	DRP (kg ha^{-1})	% Total
1	0.72	47.37	2.03	55.62	3.59	47.24
2	0.28	18.42	0.39	10.68	1.04	13.68
3	0.12	7.89	0.19	5.21	0.63	8.29
4	0.19	12.50	0.35	9.59	0.66	8.68
5	0.03	1.97	0.10	2.74	0.39	5.13
6	0.08	5.26	0.20	5.48	0.42	5.53
7	0.03	1.97	0.12	3.29	0.26	3.42
8	0.02	1.32	0.10	2.74	0.25	3.29
9	0.03	1.97	0.10	2.74	0.21	2.76
10	0.02	1.32	0.07	1.92	0.15	1.97
Total	1.52c		3.65b		7.60a	

† Totals followed by the same letter are not different according to LSD ($p<0.05$).

Table 3.3. Surface horizon pH, extractable P concentration, and P sorption index averaged by treatment in samples taken after last runoff event.
† Means ($n = 3$) followed by the same letter are not different ($p < 0.05$)

Treatment		pH	DI water P	Mehlich III P
Rain	Manure		mg kg^{-1}	mg kg^{-1}
R1	L1	5.92a†	1.48ab	27.82b
R1	L2	5.84a	1.77ab	34.43b
R1	L3	5.93a	6.81a	63.60a
R2	L1	5.63a	2.49ab	35.43b
R2	L2	5.76a	3.41ab	46.64b
R2	L3	5.88a	3.44a	65.92a
R3	L1	5.79a	2.24ab	33.68b
R3	L2	5.69a	0.72ab	31.33b
R3	L3	5.73a	5.29a	57.84a
R1	Control	5.70a	0.30b	21.76b

Table 3.4. Statistical parameters associated with P loss prediction equations.

Treatment	P form	R^2	RMSE (mg L^{-1})	n
R1	TP	0.91	2.93	90
	DRP	0.91	2.30	90
R2	TP	0.73	2.52	90
	DRP	0.68	2.32	90
R3	TP	0.86	1.13	90
	DRP	0.82	0.95	90

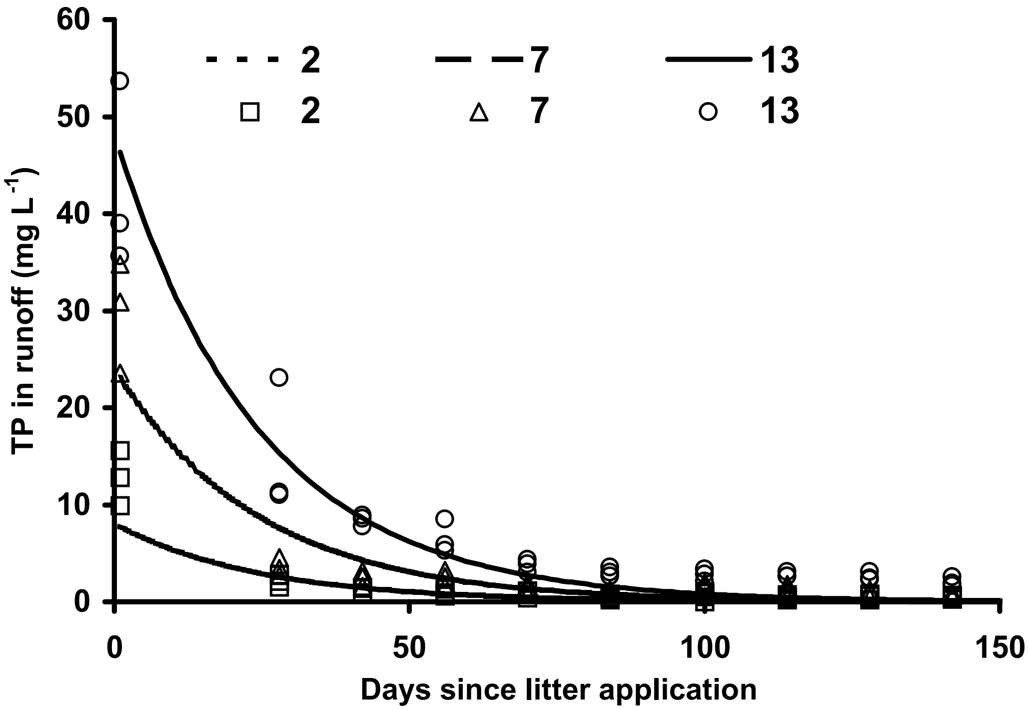

Figure 3.1. Symbols indicate observed TP in runoff (mg L^{-1}) for 2,7 and 13 Mg ha^{-1} litter applications and lines indicate TP in runoff from 2, 7, and 13 Mg ha-1 litter treatments as predicted with the equation developed from the R1 treatment data.

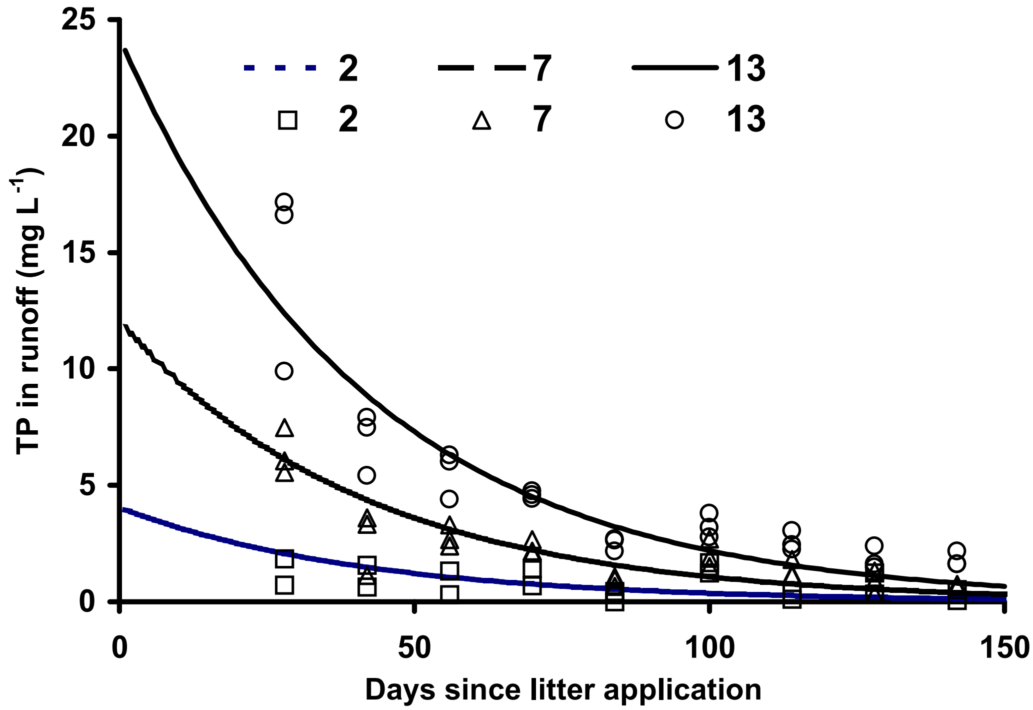

Figure 3.2. Symbols indicate observed TP in runoff (mg L^{-1}) for 2,7 and 13 Mg ha^{-1} litter applications and lines indicate TP in runoff from 2, 7, and 13 Mg ha-1 litter treatments as predicted with the equation developed from the R3 treatment data.

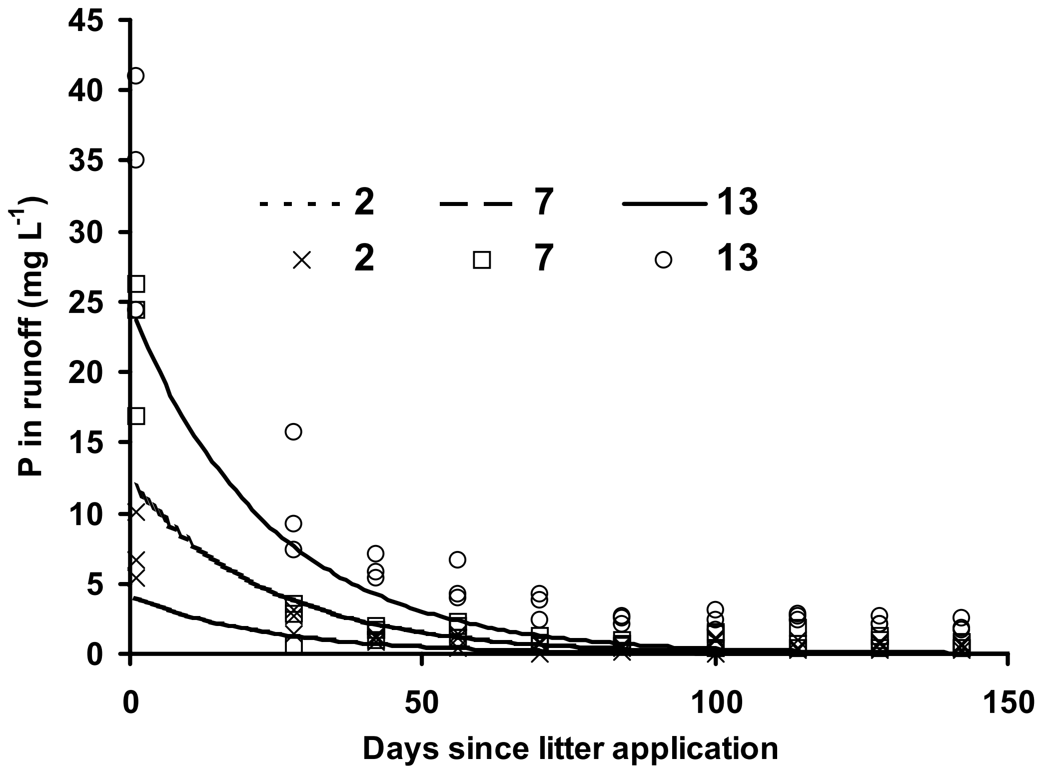

Figure 3.3. Symbols indicate observed DRP in runoff (mg L^{-1}) for 2,7 and 13 Mg ha^{-1} litter applications and lines indicate DRP in runoff from 2, 7, and 13 Mg ha-1 litter treatments as predicted with the equation developed from the R1 treatment data.

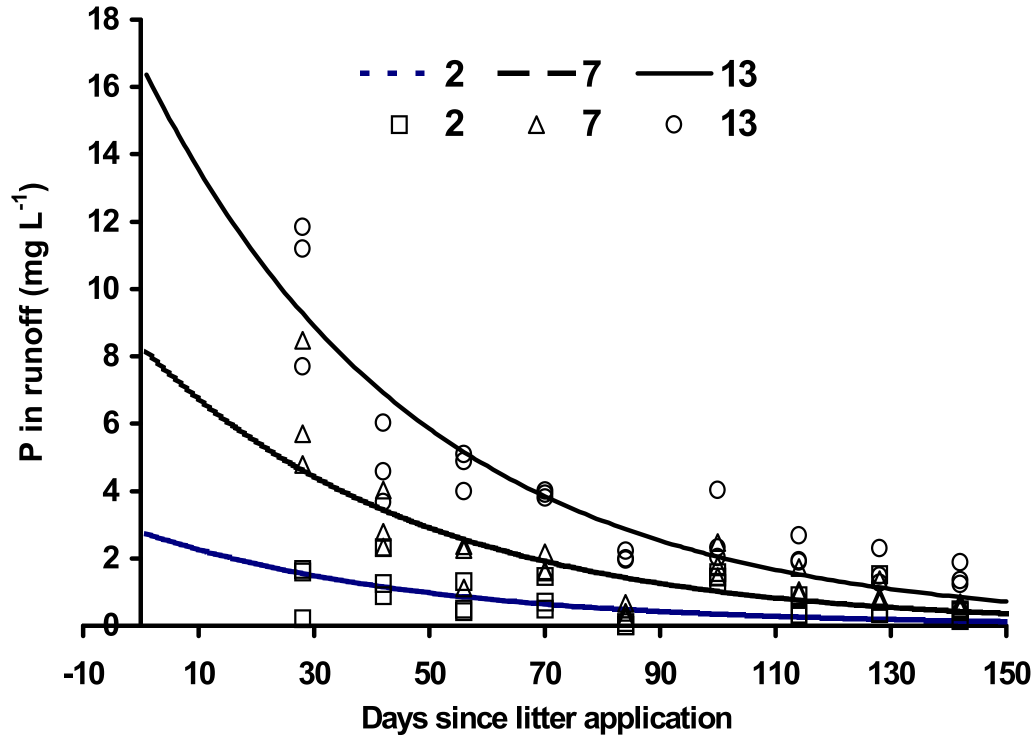

Figure 3.4. Symbols indicate observed DRP in runoff (mg L^{-1}) for 2,7 and 13 Mg ha^{-1} litter applications and lines indicate DRP in runoff from 2, 7, and 13 Mg ha-1 litter treatments as predicted with the equation developed from the R3 treatment data.

CHAPTER 4

MODELING SOLUBLE PHOSPHORUS LOSSES FROM GRASSLANDS FERTILIZED WITH POULTRY LITTER[1]

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[1] P. D. Schroeder, D. E. Radcliffe, M. L. Cabrera and C. D. Belew. To be submitted to J. Environ. Qual.

ABSTRACT

The loss of Dissolved Reactive Phosphorus (DRP) in runoff from pastures and hayfields where poultry litter (manure and bedding) is applied may be a significant contributor to agriculture's impact on the eutrophication of Georgia's lakes and streams. Loss of DRP from grasslands fertilized with poultry (Gallus gallus domesticus) litter was modeled with equations developed previously by the authors (Schroeder et al., 2002). Modeling results were compared to runoff DRP losses observed over a two-year period from five poultry litter treated fields previously reported by Pierson et al. (2001). Results indicate that the model was effective at predicting DRP concentration in runoff ($r = 0.8$). The prediction seemed to be most accurate for runoff events that occurred shortly after litter application. However, predictions became less accurate for events that occurred long after application. The reduction in prediction accuracy was due to the tendency of observed DRP concentration to fluctuate over time. These fluctuations may be explained by changes in the pool of soluble P. Fluctuations in the soluble P pool may be related to variable source areas within the field and lysis of microbial cells caused by rapid wetting during large rainfall events.

INTRODUCTION

The principal components of agricultural non-point source pollution are sediment, bacteria, nitrogen (N), and phosphorus (P). Of these, P is the element most commonly associated with eutrophication in freshwater systems because these systems are usually P limited (Heckey and Kilham, 1988).

Forms of P in runoff can include both organic and inorganic forms as well as P associated with mineral or organic particles transported in the runoff. Because erosion rates from hayfields and pastures are low, Dissolved reactive P (DRP) is usually the dominant form of phosphorus in hayfield and pasture runoff (Sharpley et al., 1992). Dissolved P in runoff results from the interaction of rainwater with a thin layer of the soil's surface and any surface applied materials that contain P such as animal manures.

Chemical extractants that simulate the availability of soil P have been used for decades to predict crop yield response to added P in fertilizer. The traditional interpretations of these tests as predictors of plant available P are based on extensive research. However, recent research supports the use of these "agronomic" tests for environmental purposes. Pote et al. (1999) report that P in runoff was highly correlated with STP in the upper 3 cm of pasture plots. Similarly, in a review of several P runoff studies, Sharpley et al. (1996) report that Bray-1 P extracted from the upper 0 to 5 cm of several soils accounted for 58 to 98% of the variation in DP concentration in runoff.

Application of animal manures can increase the concentration of P in runoff. Total P (TP) concentrations were 4.5 mg L^{-1} higher in runoff from bermudagrass plots receiving poultry (Gallus gallus domesticus) litter at a rate of 11 Mg ha^{-1} than from unfertilized control plots (Heathman et al., 1995). A positive relationship has also been shown between the rate of manure application and the concentration TP, DRP and

Particulate P (PP) in runoff (McLeod and Hegg, 1984; Mueler et al., 1984; Edwards and Daniel, 1994; Vervoort et al., 1998; Wood et al., 1999).

The majority of the published research shows that most of the P is lost in the first runoff event from fields where manure has been surface applied (Edwards and Daniel, 1994; Sauer et al., 1999; Sharpley, 1997). However, the relationship between P loss in runoff from fields treated with animal manure, and storm interval and intensity is not clearly understood. Several researchers have shown after an initial spike, P levels in runoff remain above background levels for up to 18 months. Sharpley (1997) incubated 10 Oklahoma soils with poultry litter for 1 to 35 days prior to application of simulated rainfall, and found that although the P level in runoff declined through 10 consecutive rainfall events, it was still significantly above the P level in runoff from an unfertilized control plot. Similarly, in a study of surface applied poultry litter with and without tillage, Heathman et al. (1995) observed that both TP and DRP concentrations in runoff were higher from plots receiving poultry litter than from an unfertilized control plot even after 10 rainfall events. In a multiple year study, Pierson et al. (2001), found that a single application of poultry litter to a small watershed in Georgia produced increased P in runoff for up to 18 months.

Several computer models have been developed to simulate the fate and transport of non-point source pollutants including pesticides, metals, and nutrients. Several models, including CREAMS (Knisel, 1980), SWAT (Arnold et al., 1998), AGNPS (Young et al., 1989), and EPIC (Sharpley and Williams, 1990) have been developed to predict the impact of agricultural practices on water quality. In all these models DRP loss in surface in surface runoff is based on the concept of partitioning pesticides into the solution and sediment phases as described by Leonard and Wauchope (1980).

In EPIC, for example, most of the P lost in runoff is assumed to be associated with the sediment phase, and sediment transport of P is simulated with a loading function developed by McElroy et al. (1976) and modified for application to single events by Williams and Hann (1978). The function is

$$YP = 0.001\ (Y)\ (c_p)\ (ER)$$

where YP is the concentration of P in runoff (kg ha^{-1}), c_p is the concentration of P in the top soil layer (g t^{-1}), Y is the sediment yield (kg ha^{-1}), and ER is the enrichment ratio. The enrichment ratio is the concentration of P in the sediment divided by the P concentration in the soil.

Because most of the P lost in runoff is assumed to be associated with the sediment phase, soluble P loss is described by the following simple equation

$$YSP = 0.01\ (c_{LPl})\ (Q) / k_d$$

where YSP is soluble P (kg ha^{-1}) lost in runoff, Q is runoff volume, c_{LPl} is the available P (g Mg^{-1})) in horizon l , and k_d is the concentration of P in sediment divided by the P concentration of the runoff water. The default value of k_d used in EPIC is 175. This value was chosen as a best-fit estimate by Williams (1995).

Recently, a modification has been developed for EPIC's soluble P loss equation because the original EPIC equation underestimated soluble P losses. The modified soluble P equation uses a nonlinear function of organic P to improve the soluble P loss estimate

$$YSP = ((AP)(Q)(RTO))/0.1(Wt)(k_d) \text{ and}$$

$RTO = 10(Wp/Wt)$

Where *AP* is the labile P content of the upper 10 cm of the soil, *Wt* is soil weight, and *Wp* is the organic P content of the soil. For the most part, P Loss in EPIC as well as most of the other models is based on the interaction between runoff and surface soil. Furthermore, these models generally treat unincorporated surface applied manure as inorganic P fertilizer and do not account for direct loss of P from manure to runoff.

The purpose of this research was to bring together previously developed equations relating DRP loss in runoff to STP and surface applied poultry litter, and to evaluate the accuracy of P losses predicted by these equations.

MATERIALS AND METHODS

Description of Monitored Site

Data presented in this study as "observed" data was initially reported by Pierson et al., (2001). In their study, five fescue (Festuca arundinacea Schreb.)-common Bermudagrass [Cynodon dactylon (L.) Pers.] fields (0.72 to 0.79 ha) were bordered by earthen berms and fitted with H-flumes and ISCO (Lincoln, Ne) refrigerated samplers. Soil series present at the Eatonton, GA site include Cecil (fine, kaolinitic, thermic typic kanhapludults), Altavista (fine-loam, mixed, semiactive, thermic, Aquic Hapludults), Helena (fine, mixed, semiactive, thermis Aquic Hapludults), and Sedgefield (fine, mixed, active, thermic, Aquic Hapludults). Precipitation and runoff volume were recorded at 5-minute intervals. During the two years studied, poultry litter was applied four times: March 16, 1995 (102 kg P ha^{-1}), October 30, 1995 (112 kg P ha^{-1}), March 4, 1996 (174 kg P ha^{-1}), and September 25, 1996 (103 kg P ha^{-1}). Litter samples were analyzed for TP as described by Kuykendall et al. (1999). Runoff samples were filtered (0.45 μm) and analyzed for DRP by the molybdate blue method (Murphy and Riley, 1962).

Description of Modeling Equations

P loss from surface-applied litter is estimated with equations that were developed from previous fieldwork involving P loss from surface applied poultry litter under simulated rainfall where three litter application rates of 2, 7, and 13 Mg ha^{-1} were studied (Schroeder et al., 2002). There were two rainfall scenarios: sufficient rainfall to produce 30 min. of runoff immediately after litter application (R1), and small rainfall events every 7 d (5 min. at 75 mm hr^{-1}) for 30 days and then sufficient rainfall to produce 30 min. of runoff (R3) were studied. The three litter application rates and two rainfall scenarios were applied to 1 x 2 m plots in a 3 x 3 randomized complete block design with three replications. After initial runoff scenarios, simulated rainfall was applied and runoff collected every 2 weeks for 6 months. Non-linear regression was used to develop equations to predict P concentration in runoff as a function of time and initial P application. The equation for the immediate runoff treatment (R1) was:

$$DRP\ (mg\ L^{-1}) = P_o \bullet 0.116 \bullet e^{-0.024t} \qquad [1]$$

For delayed runoff (R3) the equation was:

$$DRP\ (mg\ L^{-1}) = P_o \bullet 0.080 \bullet e^{-0.021t} \qquad [2]$$

where P_o is the total P applied as litter in kg ha^{-1}, and t is the number of days since litter application. P loss in kg ha^{-1} is calculated from P concentrations and runoff volume In addition to the above equations relating litter application and time to P loss in runoff, we looked at the effect of litter application on soil test P. Litter application and STP data from the plot study described above was used to develop an equation relating increases in

STP to litter application. Stepwise multiple regression was performed to determine the most important factors controlling soil test P (STP) change due to poultry litter application. Soil test P was regressed against manure P application rate, pH, cumulative rainfall, and runoff depth. Mehlich III STP was strongly related to P application rate and soil pH (r = 0.77) by the following equation:

$$Mehlich\ III = -112.12 + 23.55\ (pH) + 0.17\ (litter\ TP) \qquad [3]$$

where manure TP is the total P applied in poultry litter in kg P per ha.

The relationship between STP and P loss in runoff was modeled with equations developed by the author in previous research (Schroeder et al., 2002), which focused on the relationship between STP and P in runoff in the absence of poultry litter. In this study, a total of 54 rainfall simulations (3 rainfall events, 3 paired plots, and 6 sites) were conducted (Schroeder et al., 2002). Total P and DRP in runoff and Mehlich III extractable soil P were determined. Phosphorus loss from soil P was described by the following equations that relate TP and DRP in runoff to Mehlich III STP for samples taked to a depth of 10 cm..

$$DRP\ (kg\ ha^{-1}) = (0.0018 \bullet STP + 0.15) \bullet (RUNOFF \bullet 0.01) \qquad [4]$$

Where RUNOFF is in mm. The initial function ($0.0018 \bullet STP + 0.15$) calculates DRP in mg L^{-1}which is then converted to kg ha^{-1} by the second factor ($RUNOFF \bullet 0.01$).

Modeling Runs

Thirty-nine runoff events with corresponding rainfall data were identified over the period from January 1st, 1995 to December 31st, 1996. In order to model changes in STP, initial values were used for pH and Mehlich III STP recorded at the Eatonton sites (6 and 28 (mg kg^{-1}), respectively) were used. Actual runoff depths, used to model P loss, were also obtained from Pierson et al. (2001). Simulations were run from the date of each litter application described above to the next application. The predicted for Mehlich III STP from each litter application was used as the initial STP value for the next simulation run.

Data Analysis

To evaluate modeling performance we used the following measures: correlation coefficient (r), a measure of the linear correlation between observed and simulated results; root mean square error (RMSE), which measures the inherent error in the simulation; and relative root mean square error (RRMSE = RMSE/observed mean $\times$ 100), RRMSE is a measure of error in relationship to the mean. In addition to the above analysis, we also regressed observed against simulated results and analyzed the intercepts and slopes to determine if they were different from 0 and 1, respectively (SAS Institute, 1994).

RESULTS AND DISCUSSION

STP Prediction

Soil samples were collected on three dates from the research plots at Eatonton, GA. The average STP from these plots as well as the simulated STP are presented in Table 3.1. The observed and simulated STP values for 1995 reflect the effects of the first litter application. The STP values for 1996 reflect the effect of the second litter application, while the 1997 STP value reflects the effect of both the third and fourth litter applications. It appears that the STP equation produced a fairly reasonable prediction of changes in STP levels associated with litter application. However, due to the limited number of observations reported here it is impossible to determine statistical significance.

P loss Prediction

Equations one and two, and four, were effective (Table 3.2.) at predicting event DRP loss (kg ha^{-1}) when actual event runoff values were used (Fig. 3.1). The fact that the equations predicted P loss accurately when modeled with observed runoff indicates the importance of runoff prediction accuracy in prediction P losses.

There was little difference in the accuracy of prediction of DRP concentration with either equation 1 or equation 2 (Fig. 3.2, and Table 3.2). The RMSE values for both equations were greater than 89 % indicating that the average predicted DRP concentration had an error equal to 89 % of the mean observed DRP concentration. Regression analysis revealed that the slope and intercept of the equation 2 line was significantly different than one and zero, respectively. This indicates that equation 2 tended to under predict DRP concentration in runoff.

To further understand the sources of variation between observed and predicted DRP concentrations we looked at individual events (Fig 3.3 & 3.4). The initial five runoff events of 1995 (Fig. 3.3) reflect the prediction of DRP concentration based on STP (equation four). For these events equation four predicted a DRP concentration of 0.20 (mg L^{-1}). The average observed DRP concentration for these five events was 0.33 (mg L^{-1}) indicating that equation four underestimated DRP concentration.

With either equation one or two, predicted DRP concentrations for runoff events that occurred soon after litter application were closer to observed values than predictions for runoff events that occurred many months after litter application. For events within 15 days of litter application equation one tended to over-predict P loss whereas equation two tended to under-predict P loss. After the first litter application (March 16, 1995, day # 75) there was a period of almost 90 days in which no runoff producing rainfalls occurred. When significant runoff was recorded, on day 277, observed P loss was much higher than predicted. For the first three runoff events after the second litter application (October 30, 1995), equation one over-predicted P loss while equation two under predicted for the first event and over-predicted for the second and third events (Fig.3.3). After the first three events following the second litter application both equations under-predicted P loss. It is interesting to note that as the number of days since litter application increased, predicted P loss based on equation two was closer to the observed P loss than predictions based on equation one. This is due to the fact that equation two was developed from plots that had delayed runoff. The effect of the delayed runoff was a lower slope on the decay equation allowing for higher P loss as the number of days since litter application increases.

For the 1996 simulations (fig 3.4) we initially saw a general continuation of the pattern seen in the last three events of 1995, i.e. both equations under-predicted DRP concentration. The under-prediction was most pronounced with the last four events prior to the third litter application. For these events the observed DRP concentration increased markedly without any additional P application. Interestingly, the runoff volume associated with the events that occurred on the 26th, 31st, and 33rd day of 1996 was three to ten times greater than the runoff volume associated with the first five runoff events of 1996. The fact that DRP concentration seems to increase as runoff volume increases is counterintuitive, but was seen in several instances throughout this study.

After the third litter application (March 4th , day 65, 1996) both equations under-estimated DRP concentration on the 66th and 67th days, but the under-estimation was much greater with equation 2. The fact that equation one was a better predictor of DRP concentration for these two events is due to the fact that equation one was developed from plots where runoff immediately followed litter application. For the remainder of the events before the final litter application, except for the last event, both equations over-predicted DRP concentration. Following the final litter application (September 25, day 269, 1996), both equations drastically over-estimated DRP concentration. The observed DRP concentration on this date is somewhat suspect because only one of the five plots at the Eatonton site recorded runoff that day and the volume was very low (0.7 mm).

As the above discussion implies the observed DRP concentration did not follow a predictable decrease over time following litter applications. Since the prediction equations used in this study were simple decay equations they could not predict these increases in DRP concentration. The question follows, why does the DRP concentration increase in the absence of added P and why did the increased concentrations coincide with large volume runoff events? We propose two possible explanations that are may be interrelated.

First, the concept of variable source area (VSA) may play a role in this phenomenon. The concept of VSA is that for any given field there is a limited area within a field that contributes runoff to a stream and that the size of this area changes over time (Dune and Black, 1970). The size of the source area depends on the size of the storm and the antecedent moisture, topography, and soil type. Based on the VSA concept, increases in DRP concentration could be due to the larger source area supplying runoff during larger storms. In effect, frequent small storms remove P from the same area and may be depleting the soluble P pool in these areas. This results in a decrease in DRP concentration over time. However, when a large storm occurs the VSA increases and areas that may have a large pool of soluble P contribute to runoff, and DRP concentration increases. Alternatively, the low volume, low DRP concentration events may be the result of overland flow due to hydrophobic soil surface conditions in the area close to the collection flume. This kind of hydrophobicity is often seen in very dry surface soils.

Secondly, microbial biomass turnover due to prolonged soil desiccation during periods without rain and rapid rewetting during rainfall events may contribute significantly to the size of the soluble P pool in the VSAs. In a study of C and N fluctuations, Van Gestel et al. (1993) observed that desiccation and rewetting contributed to C and N mineralization. Kieft et al. (1987) who studied microbial response to rapid increases in water potential, reported that 17-58% of soil biomass was released upon rapid wetting of dry soils. They concluded that a rapid water potential increase could be a potent catalyst for the turnover of soil C as well as other nutrients such as N and P. The phosphorus in cellular components, such as phospholipids and nucleic acids, that are released upon cell lysis may pass through the typical 0.45 μm filter used to differentiate dissolved P from particulate P. Several studies have shown that organic P may be hydrolyzed by and react with molybdate in the commonly used molybdate-blue method of determining DRP in water samples (Haygarth et al., 1997; Ron Vaz et al., 1993; Tarapchak, 1993). In effect the increases in DRP observed may be partly due to a flux of organic P released from microbial cells and subsequently mischaracterized as inorganic DRP.

CONCLUSIONS

The results of this study indicate that the equations used may be effectively employed to predict DRP loss from poultry litter that is surface applied to pastures and hayfields when runoff is calibrated. This prediction was most accurate for runoff events that occur soon after litter application, which consequently are the events that produce the greatest DRP loss. The biggest source of variation between observed and modeled DRP concentration was associated with instances when observed DRP concentration increased in the absence of additional P application. These counter-intuitive increases in DRP concentration may be explained by a combination of processes including variable source area and microbial turnover. Additionally, further study of the dynamics of P cycling in the surface horizon and thatch layer of soils where poultry liter has been applied should be pursued.

REFERENCES

Arnold, J.G., B.A. Engel, and R. Surinivasan. 1993. A continuous time, grid cell watershed model. Proceedings of the Application of Advanced Information Technologies: Effective Management of Natural Resources Conference. ASAE, St. Joseph, MI.

Dune, T., and R.D. Black. 1970. Partial area contributions to storm runoff in a small New England watershed. Water Resour. Res. 6:12961311.

Edwards, D.R., and T.C. Daniel. 1994. Quality of runoff from fescuegrass plots treated with poultry litter and inorganic fertilizer. J. Environ. Qual. 23:579-584.

Haygarth, P.M., M.S. Warwick, and W.A. House. 1997. Size distribution of colloidal molybdate reactive phosphorus in river waters and soil solution. Water res. 31:439-442.

Heathman, G.C., A.N. Sharpley, S.J. Smith, and J.S. Robinson. 1995. Land application of poultry litter and water quality in Oklahoma, U.S.A. Fert. Res. 40:165-173.

Heckey, R.E., and P. Kilham. 1988. Nutrient limitation of phytoplankton in freshwater and marine ecosystems: A review of recent evidence of the effects of enrichment. Limnol. Oceanogr. 33:796-822.

Kieft, T.L., E Soroker, and M.K. Firestone. 1987. Microbial biomass response to a rapid increase in water potential when dry soil is wetted. Soil Biol. and Biochem. 19:119-126.

Knisel, W.G. (ed.). 1980. CREAMS: A field scale model for chemical runoff and erosion from agricultural management systems. Conserv. Res. Rep. 20. USDA-ARS, Washington, DC.

Leonard, R.A. and R.D. Wauchope. 1980. Chapter 5: The pesticide submodel. p. 88-112. *In* Knisel, W.G. (ed). CREAMS: A field-scale model for chemicals, runoff, and erosion from agricultural management systems. U.S. Department of Agriculture, Conservation research report no. 26.

McElroy, A.D., S.Y. Chiu, J.W. Nebgen. 1976. Loading functions for assessment of water pollution from nonpoint sources. Environ. Prot. Tech. Ser., EPA 600/2-76-151, 445pp.

McLeod, R.V. and R.O. Hegg. 1984. Pasture runoff water quality from application of inorganic and organic nitrogen sources. J. Environ. Qual. 13:122-126.

Mehlich, A. 1984. Mehlich III soil test extractant: A modification of Mehlich II extractant. Commun. Soil Sci. Plant Anal. 15:1409-1416.

Meuler, D.H., R.C. Wendt, and T.C. Daniel. 1984. Phosphorus losses as affected by tillage and manure application. Soil Sci. Soc. Am. J. 48:901-905.

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.

Pierson, S.T., M.L. Cabrera, G.K. Evanylo, H.A. Kuyrkendall, C.S. Hoveland, M.A. McCann, and L.T. West. 2001. Phosphorus and ammonium concentrations in surface runoff from grasslands fertilized with broiler litter. J. Environ. Qual. 30:1784-1789

Pote, D.H., T.C. Daniel, D.J. Nichols, A.N. Sharpley, P. A, Moore, Jr., D.M. Miller, and D.R. Edwards. 1999. Relationship between phosphorus levels in three ultisols and phosphorus concentrations in runoff. J. Environ. Qual. 28:170-175.

Ron Vaz, M.D. A.C. Edwards, C.A. Shand, and M.S. Cresser. 1993. Phosphorus fractions in soil solution: Influence of soil acidity and fertilizer applications. Plant Soil. 148:179-183.

Sauer T.J., T.C. Daniel, P.A. Moore, Jr., K.P. Coffey, D.J. Nichols, and C.P. West. 1999. Poultry litter and grazing animal effects on runoff water quality. J. Environ. Qual. 28:860-865.

Schroeder, P.D. 2002. Characterization of phosphorus loss in runoff from pastures and hayfields related to soil test phosphorus and poultry litter application. Ph.D. diss. Univ. of Georgia. Athens.

Sharpley, A.N. 1997. Rainfall frequency and nitrogen and phosphorus runoff from soil amended with poultry litter. J. Environ. Qual. 26:1127-1132.

Sharpley A.N., and J.R. Williams, (eds).1990. EPIC: Erosion/Productivity Impact Calculator; I. Model Documentation. USDARS Tech. Bull. 1768.

Sharpley, A.N., S.J. Smith, O.R. Jones, W.A. Berg, and G.A. Coleman. 1992. The transport of bioavailable phosphorus in agricultural runoff. J. Environ. Qual. 21:30-35.

Sharpley, A.N., T.C. Daniel, J.T. Sims, and D.H. Pote. 1996. Determining environmentally sound soil phosphorus levels. J. Soil Water Cons. 51:160-166.

Shuman, L.M., P.l. Raymer, J.L. Day, and M.J. Cordonnier. 1988. Comparison of four phosphorus extraction methods on three acid southeastern soils. Commun. In Soil Sci. Plant Anal. 19:579-595.

Tarapchak, S.J. 1993. Soluble reactive phosphorus in lake water: Evidence for molybdate enhanced hydrolysis. J Environ. Qual. 12:105-108

Van Gestel, M. R Merckx and K. Vlassak. 1993. Microbial biomass responses to soil drying and rewetting: The fate of fast and slow-growing microorganisms in soils from different climates. Soil Biol. and Biochem. 25:109-123.

Vervoort, R.W., D.E. Radcliffe, M.L. Cabrera, and M. Latimore, Jr. 1998. Field scale nitrogen and phosphorus losses from hayfields receiving fresh and composted broiler litter. J. Environ. Qual. 27:1246-1254.

Williams, J.R. 1995. The EPIC model. pp. 909-1000. *In* V.P. Singh (ed.) Computer models of watershed hydrology. Water Res. Publications, Littleton, CO.

Wood, B.H., C.W. Wood, K.H. Yoo, K.S. Yoon, and D.P. Delaney. 1999. Seasonal surface runoff losses of nutrients and metals from soils fertilized with broiler litter and commercial fertilizer. J. Enviorn. Qual. 28:1210-1218.

Young, R.A., C.A. Onstad, D.D. Bosch, and W.P. Anderson. 1989. AGNPS, a nonpoint source pollution model for evaluating agricultural watersheds. J. Soil Water Conserv. 44:168-173.

List of Table and Figures

Table 4.1. Observed and predicted Mehlich I STP values.

Date	Observed Mehlich I STP	Simulated Mehlich I STP	RMSE
	kg ha^{-1}		
10/16/95	47	41*	
2/14/96	68	69	37
3/15/97	197	130	

* Values for Mehlich I taken as (Mehlich III • 0.58 – 1.71) (Mehlich, 1985 and Shuman et al., 1988).

Table 4.2. Root mean square error (RMSE), relative root mean square error (RRMSE), intercept, slope, correlation coefficient (r) and number of observations (n) for the relationship between observed and simulated runoff and DRP loss.

Variable	RMSE	RRMSE	Intercept	Slope	r	n
	mg L^{-1}		mg L^{-1}			
DRP EQ 1	3.08	89.6	0.18	0.89	0.79	39
DRP EQ 2	3.08	89.7	1.40*	0.62*	0.72	39
	kg ha^{-1}		kg ha^{-1}			
DRP EQ 1	0.30	70.0	-0.09	1.02	0.97	39
DRP EQ 2	0.33	78.0	-0.01	0.76*	0.98	39

* Indicates significantly different from zero (intercept) or one (slope) at p = 0.05.

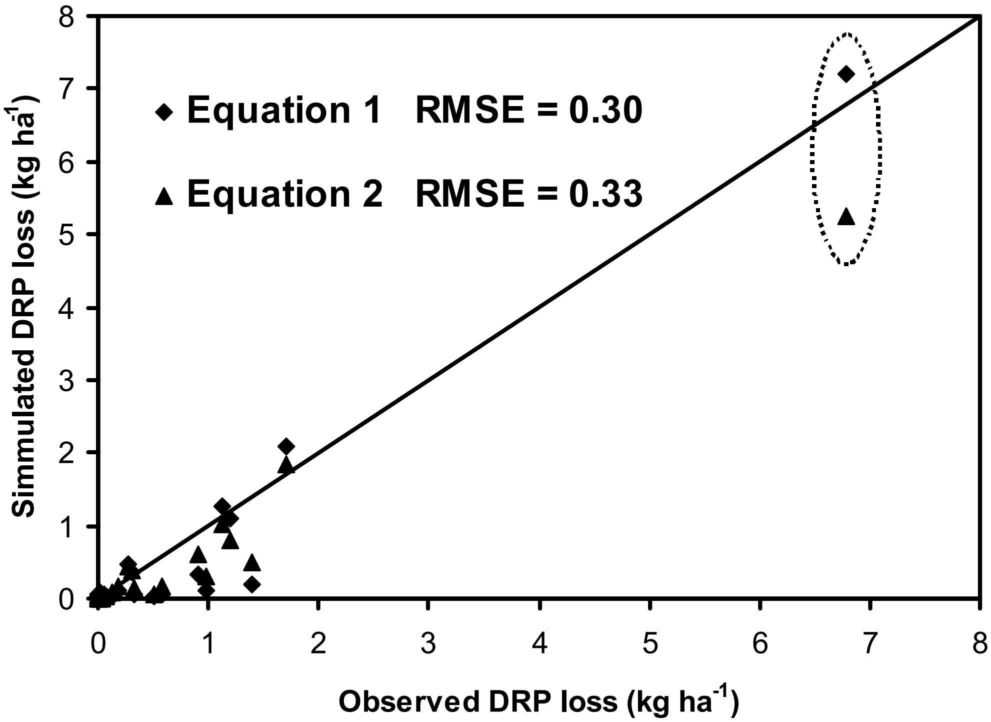

Figure 4.1. The relationship between observed and predicted DRP loss (kg ha^{-1}) when observed runoff was used (n = 39). Dashed oval indicates points that strongly influenced the relationship

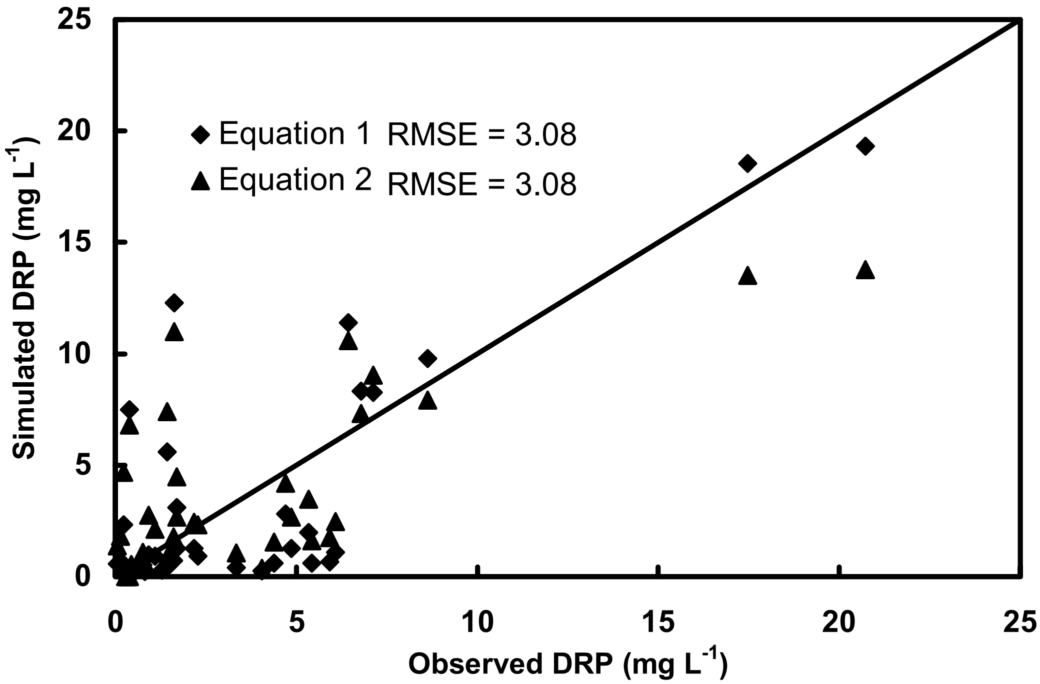

Figure 4.2. The relationship between observed and simulated DRP concentration (mg L^{-1}).

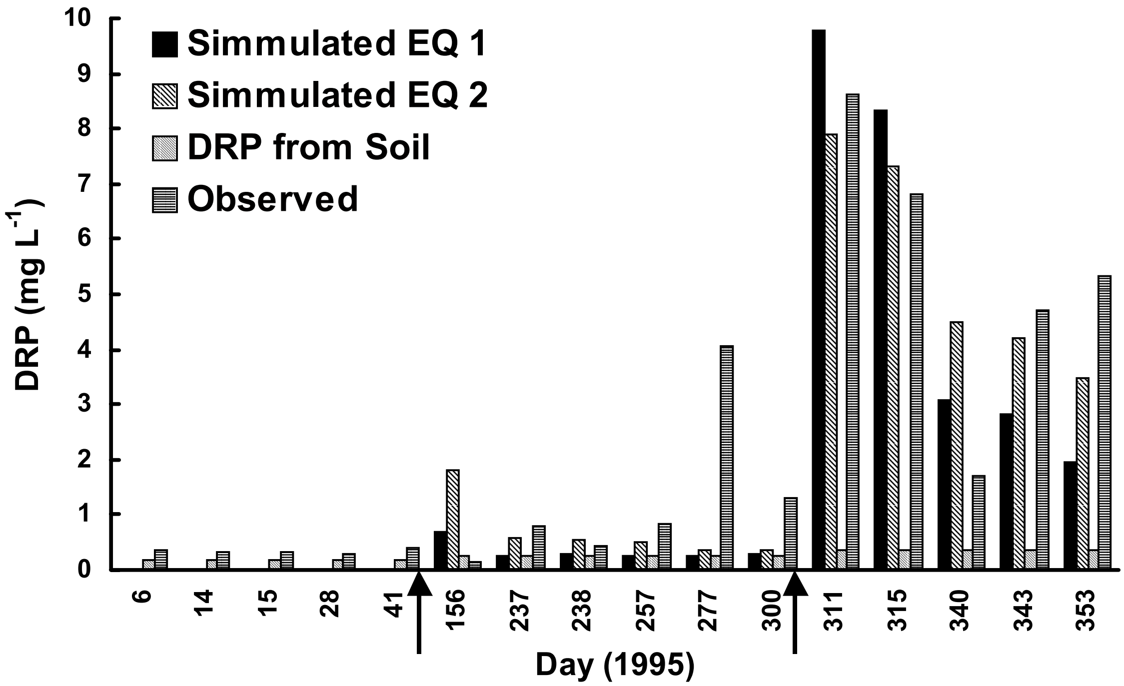

Figure 4.3. Concentration of DRP (mg L^{-1}) observed and simulated for runoff events observed in 1995. Arrows indicate the approximate dates when poultry litter was applied.

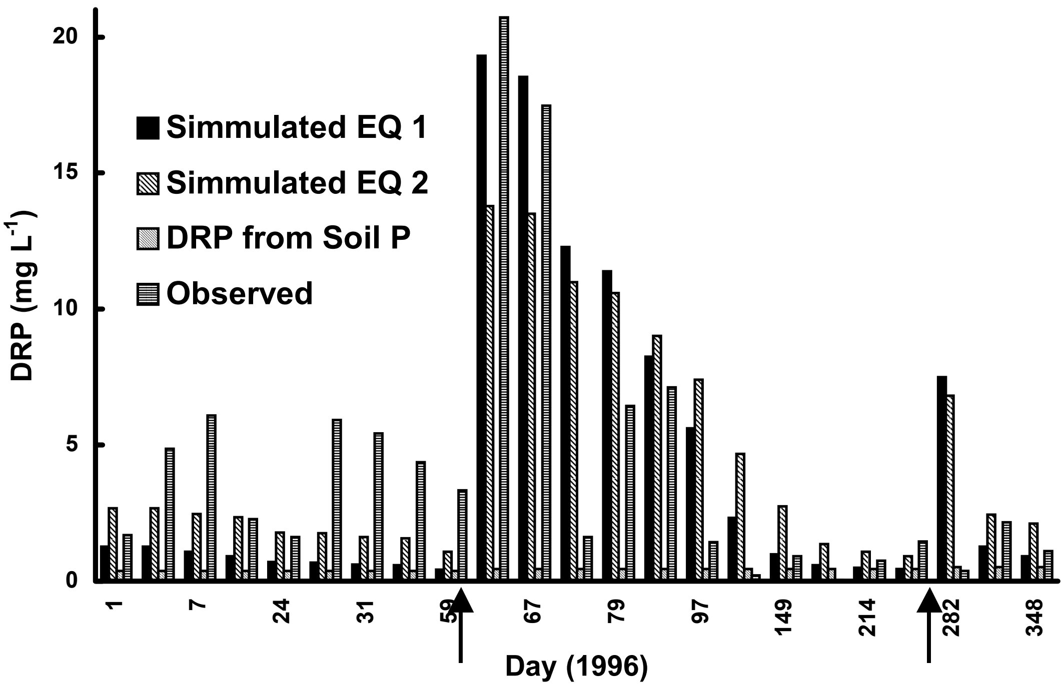

Figure 4.4. Concentration of DRP (mg L^{-1}) observed and simulated for runoff events observed in 1996. Arrows indicate the approximate dates when poultry litter was applied.

CHAPTER 5

OVERALL CONCLUSIONS

This research focused on the relationships between STP and surface applied poultry litter, and P that is lost in runoff from pastures and hayfields. There were significant correlations between all forms of P in runoff and all soil P test methods used. However, STP correlated best with field runoff P when extracted with Fe_2O_3 Paper or DI water. The strongest relationship (R^2 = 0.69) between STP and P in field runoff was obtained with BAP soil test on 5-cm samples and BAP in runoff. The highest correlation between STP method and DRP in runoff occurred with DI water extractable P and the 0-5 cm sampling depth (R^2 = 0.68). Normalizing DRP to reduce variability associated with differences in runoff volume resulted in improved correlation between DRP and DI water extractable P. The effect was most pronounced with the 0-10 cm soil sample where R^2 increased from 0.58 to 0.81.

Although STP levels were different among the three soil sampling depths, comparison of the slopes and intercepts of the regression lines for each combination of STP and depth vs P in runoff revealed that for all sampling depths were no statistical differences between the slopes and intercepts of the regression lines. Also there was no difference between the DRP vs STP and the BAP vs STP regression lines indicating that one equation could be used to predict either form of P in runoff.

By relating STP to P in runoff we were able to account for between 50 and 80% of the variability in P loss. However, for all forms of P in runoff and all STP methods, R^2

increased with the inclusion of extractable Fe and landscape position in the regression equation. The coefficients on the Fe and pH factors were both positive indicating that an increase in extractable Fe or pH produced an increase in P loss. Conversely, the coefficient of the landscape position terms were all negative indicating that P loss was lower from the lower landscape positions.

Additionally, the results of this study indicate that, for Piedmont soils in hay or pasture, no matter what STP method is used or form of runoff P measured, P loss can be predicted with reasonable accuracy. Furthermore, the addition of a few easily obtainable soil and site characteristics can significantly improve the prediction of P loss compared to STP alone. Also, the fact that extractable Fe, pH, and landscape position had an effect helps explain why the relationship between STP and runoff P varies among some soils. The rainfall treatment where runoff-producing rainfall was applied immediately following litter application produced significantly greater TP and DRP loss ($p < 0.05$) for the first runoff event. It appears quite clear that rainfall timing (i.e. time to first runoff event) and litter application rate have a dramatic effect on P loss in runoff. Additionally, after 10 runoff events P losses in runoff were still significant and resulted in P concentrations exceeding 1 mg L^{-1}, the value that has been proposed as the maximum desirable P concentration in agricultural runoff (USEPA, 1986). Total mass of P lost from all ten runoff events represented 5 to 11 % of the P applied indicating that a significant pool of litter P remained after ten events. Most of this P pool may be organic P that remains on the soil surface rather than inorganic P adsorbed to the soil because only modest increases were seen in STP levels. Additionally, an equation relating litter P application rate to changes in Mehlich III STP was developed. About 60% of the variability in Mehlich III STP was explained by this equation

Non-linear regression equations that may be useful in predicting P losses in runoff from surface applied poultry litter were developed. These equations were able to explain 68 to 91 % of the variability seen in P losses. A computer model will be developed, based on these equations and previous work by the authors relating STP levels to P in runoff, to predict P loss from poultry litter applied to pastures and hayfields. Such a model would improve the ability to assess the vulnerability of a site to P loss from surface application of poultry litter and may be helpful in designing comprehensive nutrient management plans. Because the levels of P loss seemed to rebound after several months as reflected in the runoff from the January 2002 rainfall simulations, future research in this area should be directed at determining the effect of the interval between runoff events on P loss.

Equations developed through this research may be effectively employed to predict DRP loss from poultry litter that is surface applied to pastures and hayfields. This prediction was most accurate for runoff events that occur soon after litter application, which consequently are the events that produce the greatest DRP loss. The biggest source of variation between observed and modeled DRP concentration was associated with instances when observed DRP concentration increased in the absence of additional P application. These counter-intuitive increases in DRP concentration may be explained by a combination of processes including variable source area and microbial turnover. The results of this study indicate that the equations reported here may be valuable predictors of P loss if they can be coupled with accurate prediction of runoff events. Additionally, further study of the dynamics of P cycling in the surface horizon and thatch layer of soils where poultry liter has been applied should be pursued.