ABSTRACT

JOHNSON, AMY MARIE. Phosphorus Loss Assessment in North Carolina. (Under the direction of Deanna Osmond.)

Increased concern about potential losses of phosphorus from agricultural fields receiving animal waste has resulted in the implementation of new state and federal regulations related to nutrient management. In response to strengthened nutrient management standards that require consideration of phosphorus (P), North Carolina has developed a site-specific P indexing system called the Phosphorus Loss Assessment Tool (PLAT) to predict relative amounts of potential P loss from agricultural fields.

The objectives of this study were threefold: (i) to predict the percentage and types of farms that will be forced to change management practices due to implementation of the PLAT index (ii) to evaluate the predictive ability of PLAT using sensitivity/uncertainty analysis; (iii) to establish whether the method of predicting soil P used in PLAT, Mehlich-3 soil test P, is an adequate proxy for a more descriptive measure of soil P, namely the oxalate degree of P saturation.

Based on a statistically random sampling of agricultural sites in all 100 counties, approximately 8% of producers in the state will be required to apply animal waste or inorganic fertilizer on a P basis rather than a nitrogen basis, with the percentage increasing for farmers applying animal waste (~27%). The tool predicted the areas in the state that are known to be disproportionately vulnerable to P loss due to histories of high P applications, high densities of animal units, or soil type and landscapes that are most susceptible to P loss. Statistical evaluation of the tool showed that soluble loss pathways, surface runoff and subsurface drainage, impact predictions of P loss more than either loss of P through erosion or applied source. Sensitivity to input factors related to the different methods for estimating soluble losses between naturally and artificially drained conditions varied depending on input values. Predicted P losses from organic soils were drastically more sensitive to soil test P than other soil types. Results from this analysis will help determine which areas to focus resources on in an attempt to improve upon the accuracy of PLAT's predictions of P loss.

This study showed that PSR_{ox} was significantly related to Mehlich-3 STP, especially when soils are grouped according to P retention properties. Mehlich-3 extracted 49% of P_{ox}, 88% of Al_{ox}, and 15% of Fe_{ox}. Phosphorus saturation measurements did not appear to be valid for organic soils, as they did not

follow the assumption that amorphous iron and aluminum are responsible for the majority of P retention. Phosphorus saturation as determined by Mehlich-3 extraction was significantly correlated to P saturation as determined by oxalate extraction. However, caution should be used before substituting Mehlich-3 PSR for oxalate P saturation due to the inability of Mehlich-3 to account for extractable Fe and thus a soil's P sorption capacity.

PHOSPHORUS LOSS ASSESSMENT IN NORTH CAROLINA

by
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BIOGRAPHY

Amy Marie Johnson was born in Satellite Beach, Florida on June 28, 1974. She was raised and lived in Satellite Beach until attending college at Warren Wilson College in Asheville, North Carolina. It was here that she became involved with agriculture by working with Ernst Laursen on the Warren Wilson College farm. During this time she began to develop an interest in soils. She received her B.A. in Environmental Studies in 1996. She then began working at the Mountain Horticultural Crops Research Center in Fletcher, North Carolina, where she received a research assistantship to work on a Masters Degree with Greg Hoyt at North Carolina State University.

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Chapter 1

The Use of Mehlich-3 as an Environmental Index of P Saturation in NC Soils

INTRODUCTION

The loss of phosphorus (P) from agricultural fields has been cited as a major contributor to decreased water quality in the United States (USEPA, 1996) and is especially aggravated by land application of large amounts of animal wastes (Breeuwsma et al., 1995). Therefore, the need to accurately predict losses of soluble P in runoff and subsurface leaching from agricultural fields has become a top priority in many states as public policy demands a better accounting of P movement (Sharpley et al., 1996; Sims et al., 2000; Sims and Coale, 2002). States, such as North Carolina, that have large animal production industries located near or adjacent to nutrient sensitive, yet economically important water bodies are likely to continue to see an increase in lawsuits and forced implementation of new regulations or strengthening of existing policies. Consequently, both state and national regulations have been implemented to limit excessive applications of P from animal wastes and other sources. Individual states, most notably those states in the Chesapeake Bay Watershed, have had their own laws in effect for some time providing for more restrictive management of agricultural nutrients (Sims and Coale, 2002).

One of the most important national initiatives related to P management has been the revision of the Natural Resources Conservation Service's Standard 590 to include a requirement that land applicators of animal waste implement P-based management in certain situations. The adoption of this change requires each state to implement one of three methods to account for potential P loss; an agronomic threshold, an environmental threshold, or a site-specific P index. To this effect, soil test phosphorus (STP) is used either directly as a single indicator of potential P loss or indirectly as one factor among many embedded within a P-loss site assessment. Currently, forty-nine states use some measure of STP to help predict loss of P in surface and subsurface flow (Sharpley et al., 2003).

Because STP was originally intended to predict plant response to soil P, there is some question as to its validity as an indicator of soluble P loss. This has led to numerous studies attempting to correlate readily available STP data with more specific, but more tedious measures of soil P loss (Pote et al., 1996; Sharpley et al., 1996; Pote et al., 1999; Paulter and Sims, 2000; McDowell and Sharpley, 2001; Maguire and Sims, 2002b; Sims et al., 2002). The current method of relying on STP to estimate potential dissolved P loss lacks the ability to differentiate between different soil types' distinct abilities to retain P. A measure of soil P saturation, which relates not only the amount of P a soil has accumulated but also the soil's capacity to continue to sorb additional P, should theoretically represent a significant improvement over STP. For this reason, P saturation may be a better predictor of a soil's proclivity to release P across a range of different soil types. Indeed, many studies have found that the amount of soluble P that can be released from the soil matrix, and therefore potentially degrade water quality, can be related to the degree of phosphorus saturation (Lookman et al., 1996; Sharpley, 1995; Beauchemin et al., 1996; Pote et al., 1996; Pote et al., 1999; Hooda et al., 2000; Maguire et al., 2000b; Paulter and Sims, 2000).

One method of determining P saturation that has received much attention is the use of oxalate degree of phosphorus saturation (DPS_{ox}) as measured by ammonium oxalate extraction. The sum of oxalate-extractable iron and aluminum has been correlated to a soil's capacity to sorb P (Freese et al., 1992; Yuan and Lavkulich, 1994; Darke and Walbridge, 2000; Paulter and Sims, 2000; Nair and Graetz, 2002; D'Angelo et al., 2003), although the correlation varies across different soil types (Beachemin and Simard, 1999). Oxalate in darkness extracts organic bound iron and aluminum as well as poorly crystalline iron and aluminum oxides, which are considered to be the more reactive forms of Fe and Al and the main sorbents of P in acid soils (van der Zee and van Riemsdijk, 1988). Paulter and Sims (2000) found that the percentage of reversibly sorbed P that was released into soil solution as measured by three different methods increased as the P sorption capacity became increasingly saturated. Hooda et al. (2000) determined that across different soil types, ranging from loamy sand to sandy clay loam, DPS_{ox} best predicted desorbable P compared to other techniques. Pote et al. (1996) also reported that DPS_{ox} was significantly related to dissolved P in runoff from field plots. However, this study involved only one soil type and the authors cautioned that uncertainty remains about how well this relationship will hold across drastically differing soil textures. A later study by Pote et al. (1999) reinforced the existence of a relationship between DPS and dissolved P but contended that each soil type had a unique regression equation describing this correlation. Lookman et al. (1995) reported a significant relationship between P sorption capacity and soil texture, although light alluvial soils did not fit this relationship due to the much higher iron content of these samples. Saunders (1964) was able to relate P retention of New Zealand soils to differing degrees of weathering (essentially, the degree of crystallinity of iron and aluminum oxides). Therefore, it may be possible to group soils according to properties affecting P retention and set individualized guidelines for limiting P buildup in each group (Beauchemin and Simard, 1999).

In response to a number of very public animal waste related mishaps, North Carolina has enacted state regulations affecting animal waste operators that are either more restrictive than or directly based on the NRCS 590 nutrient management standard. In compliance with Standard 590, North Carolina has developed a site-indexing tool called the Phosphorus Loss Assessment Tool (PLAT) to estimate phosphorus losses from four distinct pathways: erosion P loss, surface runoff P loss, subsurface drainage P loss, and applied source P loss (Havlin et al., 2001). The accuracy of predicting the losses from the first three pathways depends upon the ability to quantify how much sorbed P can be dissolved or released as particulates from soil constituents. The present version of PLAT relies on Mehlich-3 STP to assist in this prediction. The question of how STP relates to a more rigorous environmental indicator of potentially soluble P such as DPS_{ox} for soils of varying properties is therefore of interest in the assessment of the PLAT index.

Since this study involved a diverse group of soils with contrasting properties, categorizing soils according to characteristics related to P retention was considered beneficial for predicting soluble P loss. In North Carolina, all soils in the state have been categorized by NC extension personnel into “soil management groups”, originally developed to differentiate soils, based on their unique properties, into divisions that would theoretically have similar yield and management (see www.soil.ncsu.edu/nmp/SMG_Final_2000.pdf). These properties are based on pedogenesis, physiographic region, drainage class, and in some cases, landscape position. From these 62 soil management groups, four classifications called “P threshold groups” were developed for use in the PLAT index and are based on properties affecting P retention and mobility, including soil texture and depth to Bt horizon (Table 1). Comparisons and analysis of study soils were performed on these soil groupings as well as the data set as a whole.

The P threshold concept is used in North Carolina’s P-loss assessment index and implies that, in general, the higher percentage of clay a soil has, the greater its ability to sorb P (Cox and Hendricks, 2000). A “P threshold" has been defined for use in the PLAT index as the Mehlich-3 P (M3P) value at which soil P will be dissolved into soil solution as runoff at a concentration of 1 mg L^{-1} from a particular soil. A concentration of 1 mg L^{-1} was used as the critical dissolved P concentration because this is the limit that applies to point source wastewater treatment plants (USEPA, 1996). Threshold M3P values for each of the four soil groups were determined from past studies of NC soils (Fox and Kamprath, 1970; Cox and Hendricks, 2000; Tarkalson, 2001; Kamprath, E.J., unpublished data). Based on this criterion, a threshold value for clay soils was set at 500 mg kg^{-1} M3P because these soil types are capable of sorbing greater amounts of P as compared to coarser-textured soils. Greater amounts of aluminum and iron oxides are associated with a higher P sorption capacity of soils. On the other extreme are organic soils with their inability to hold significant amounts of P due, presumably, to competition by organic matter functional groups for P sorption sites. These shallow organic soils (<66 cm of organic material) have been found to have a greater dissolution of P over successive extractions when compared to mineral soils, as well as to mineralize a substantial amount of organic P (Daughtrey et al. 1972). Therefore, the threshold value for soils with an organic surface layer of 40 cm or greater was set at 50 mg kg^{-1}. Threshold values for loamy soils and sandy soils were set at 200 mg kg^{-1} and 100 mg kg^{-1}, respectively. For simplicity, these threshold groupings will subsequently be referred to as organic, sand, loam and clay soil groups.

Table 1. Description of North Carolina's P threshold groups.

P threshold groups	**Location**	**P threshold (mg kg^{-1})**
Organic	Coastal Plain	50
Sand	Coastal Plain and alluvial areas of Piedmont and Mountains	100
Loam	Piedmont and Coastal Plain	200
Clay	Piedmont and Mountains	500

Accordingly, the first objective of this study was to determine the phosphorus saturation status of diverse soils in agricultural areas of North Carolina having histories of animal waste application or long-term fertilizer use. We also wished to examine the suitability of separating soils into relatively homogenous groups (P threshold groups) and how each of these proposed soil groupings behaved with respect to the STP parameters studied. Soil P recommendations in North Carolina are based on M3P and it seems unlikely that soil test laboratories will logistically be able to perform oxalate extractions on a large scale. Therefore, our second objective was to establish whether Mehlich-3 ST is an adequate proxy for the more descriptive oxalate P saturation measurement by evaluating the relationship between the two measurements in the study soils. It may be possible to use the Mehlich-3 extractant to predict P saturation as has been suggested by Khiari et al., (2000) for use in Canada, and by Sims et al., (2002) in Delaware. Thus, our third objective was to ascertain whether Mehlich-3 soil P saturation correlates with the standard oxalate P saturation measurement. In association with this, differences between the ability of oxalate and Mehlich-3 to extract P, Al and Fe in soils with diverse properties were assessed.

MATERIALS AND METHODS

A total of eight counties, encompassing each of North Carolina's three physiographic regions (Mountains, Piedmont, and Coastal Plain), were selected for sampling (Figure 1). Counties were chosen to represent a range of soil types and to target counties having histories of heavy phosphorus applications, via either animal wastes or inorganic fertilizer. Table 1 shows the percentage of agricultural soils in each selected county submitted to the NC Dept. of Agronomic and Consumer Services, Soil Testing Division in each selected county that had M3P values rated agronomically high or very high. Soils in these categories have M3P levels >53 mg kg^{-1} and experience no crop response to further additions of P (SERA-IEG-6, 2001).

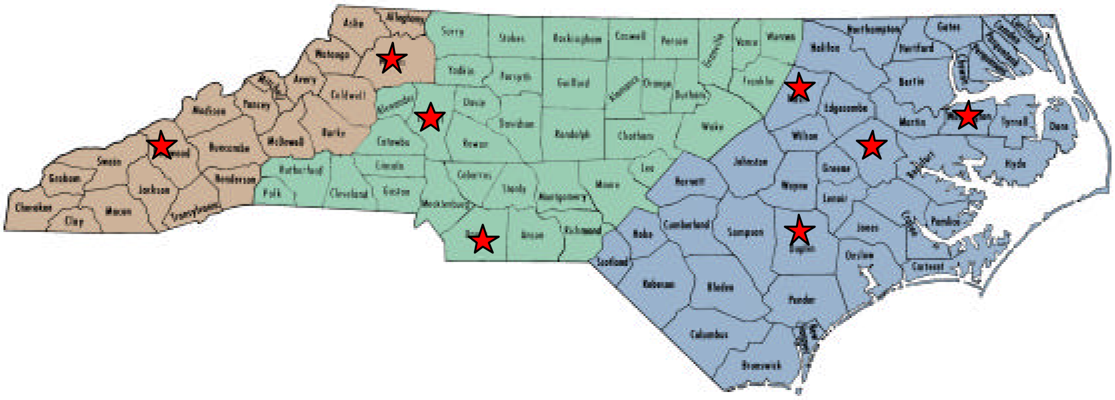

Figure 1. Location of counties selected for sampling.

Sampling was performed to assure that soils from farm fields applying swine, poultry and dairy wastes, in addition to those receiving inorganic fertilizers, were obtained. Sites were selected in each county to represent the predominant P source applied in that county (Table 2). Choice of sample fields was distributed between both pastured and row-cropped fields.

Soils were collected from a total of 86 sites, 15 sites in the Mountains, 16 sites in the Piedmont, and 55 sites in the Coastal Plain, including a subset of eight sites comprising organic soil systems. The Coastal Plain occupies 45% of NC's land area and 84% of the state's agricultural land and therefore represented a higher proportion of the samples taken. Samples were collected to a depth of 80 cm (32 inches) in 10 cm (4 inch) increments. Composite soil samples were obtained by combining five to ten separate soil cores (7.62-cm diameter) that had been taken randomly across each field. Trends with depth were shown by keeping the sample-by-depth structure intact. When attempting to correlate chemical soil properties, i.e. Mehlich-3 with oxalate P, each individual depth increment (10 inches) was analyzed separately to give a

Table 2. Selected properties and sampling scheme of counties selected for sampling.

County	Physiographic Region	% of soils above 53 mg M3P kg^{-1}†	# of sample sites for each fertilizer type
Duplin	Coastal Plain	90	8 – poultry 8 – swine 8 – inorganic
Nash	Coastal Plain	89	4 – poultry 4 – swine 4 – inorganic
Pitt	Coastal Plain	88	3 – poultry 2 – swine 6 – inorganic
Washington	Coastal Plain	58	8 – inorganic
Iredell	Piedmont	62	4 – dairy 2 – inorganic
Union	Piedmont	91	5 – poultry 2 – swine 3 – inorganic
Haywood	Mountain	67	3 – dairy 2 – inorganic
Wilkes	Mountain	88	2 – dairy 7 – poultry 1 – inorganic

† N.C. Dept. of Agric. Consumer Services (2002).

total of 685 samples. At one site, the bottom three depths could not be sampled due to wetness. For basic statistics, topsoils (0-20 cm) and subsoils (20-80 cm) were grouped separately, yielding 172 topsoil samples and 513 subsoil samples.

Samples were air-dried and ground to pass through a 2-mm sieve prior to analysis. Particle size was determined by pipette analysis after removing organic matter from oven-dry soil with hydrogen peroxide and dispersing with Na-hexametaphosphate (Kilmer and Alexander, 1949). Soil pH was measured in 1:5 soil: water solution and M3P, aluminum (M3Al) and iron (M3Fe) were measured by extraction with Mehlich-3 soil test extractant (Mehlich, 1984). Oxalate-extractable P (P_{ox}), aluminum (Al_{ox}) and iron (Fe_{ox}) were measured by a modification of the McKeague and Day procedure (1966). This procedure involved dissolving 16.2 g of ammonium oxalate and 10.8 g of oxalic acid in 1L of distilled water. The pH of the combined solution was adjusted to 3.0 by the addition of dilute HCl or NH_4OH. The extraction procedure involved mixing 0.75 grams of soil and 30 ml of the 0.2M ammonium oxalate solution in 50-ml centrifuge tubes. Tubes were shaken on a horizontal reciprocating shaker in the dark for 2 hours (Parfitt, 1989). The extract was centrifuged for five minutes at 3000 RPM and filtered through a 0.45µm pore diameter syringe filter. Phosphorus, iron and aluminum concentrations in soil extracts were determined by inductively coupled plasma atomic emission spectroscopy. Extracts were diluted by a factor of 10 and a surfactant added to prevent clogging in the central torch tube (Novozamsky et al., 1986).

The degree of phosphorus saturation was described by Schoumans (2000) as follows:

$$DPS_{ox}\ (\%) = 100 \times ([P_{ox}] \ / \ \alpha\ [Al_{ox} + Fe_{ox}]) \qquad [1]$$

where ($Al_{ox} + Fe_{ox}$) is taken to represent P sorption capacity (PSC) and P_{ox}, Al_{ox}, and Fe_{ox} are expressed in mmol kg^{-1}. The parameter α is experimentally determined for a given soil type and is time and concentration dependent (van der Zee and Van Riemsdijk, 1988). Applying a α factor other than 1 for the wide variety of soils in this study would have been arbitrary and was therefore omitted in estimating DPS_{ox}, as has been suggested by other authors (Hooda et al., 2001; Kleinman and Sharpley, 2002). Therefore, we used the following equation to calculate a phosphorus saturation ratio rather than a percentage of saturation:

$$PSR_{ox} = ([P_{ox}] \ / \ [Al_{ox} + Fe_{ox}]) \qquad [2]$$

Additionally, a P saturation ratio using the Mehlich-3 extractant was calculated as has been described previously by Khiari et al. (2000) and others (Maguire and Sims, 2002a; Sims et al., 2002):

$$\text{M3-PSR I} = \text{M3P} / [\text{M3Al} + \text{M3Fe}] \quad [3]$$

$$\text{M3-PSR II} = \text{M3P} / \text{M3Al} \quad [4]$$

where M3-PSR I is the Mehlich-3 phosphorus saturation ratio involving Mehlich-3-extracted P, Al and Fe and M3-PSR II is the Mehlich-3 phosphorus saturation ratio without considering Mehlich-3-extracted Fe. Mehlich-3 P, M3Al, and M3Fe are expressed in mmol kg^{-1}.

Effects of soil groupings, depth and P source were evaluated for significance by a generalized linear model with Tukey's pairwise-comparisons. When performing statistics comparing soil properties in topsoils and subsoils, values in the 0-10 cm and 10-20 cm depths were used to represent the topsoil. In other words, each 0-10 cm depth sample and each 10-20 cm depth sample were treated as independent data points. Subsoils were represented by the 6 depths that occurred deeper than 20 cm. Relationships between extractants, P saturation models, and PSR_{ox} and M3P were quantified by least squares linear regression. All statistical analysis was performed using the GLM procedure of SAS, Version 8 (SAS Institute, 1998).

RESULTS AND DISCUSSION

Soil Test Phosphorus and Degree of Phosphorus Saturation

A total of 38 soil series and six soil orders (Ultisols, Entisols, Inceptisols, Histosols, Spodosols and Alfisols) were represented in this study. Both surface and subsoil textures of mineral soils ranged from sand to clay (see appendix, Table A1). The mean pH was 5.9 for topsoils and 5.1 for subsoil samples.

In North Carolina, M3P is used to classify a soil's ability to provide P to growing crops, with values greater than 53 mg kg^{-1} considered to be above optimum. Of the 685 study samples, 33% had a M3P above the critical agronomic threshold of 53 mg kg^{-1}. Because the recommended depth for obtaining samples for the NCDACS Soil Testing Lab is 0-20cm, we also segregated the samples into topsoils (0-20 cm) and subsoils (20-80 cm). In this case, 74% of the topsoils from this study were above the critical M3P while only 18% of subsoils exceeded 53 mg kg^{-1}. This result is consistent with expectations of the selected sampling sites, as continuous surface applications of animal wastes or inorganic fertilizers could build up M3P in surface horizons to levels at which crops are non-responsive. It is important to note that for this study we chose counties with histories of animal waste applications or high inorganic fertilization of tobacco.

Although some of the chosen counties, such as Duplin and Wilkes, represent the worst-case scenarios in terms of M3P buildup, the situation in these counties is not uncommon across NC. For instance, in 2002, 64% of soils tested by the state fell into the high (53-117 mg kg^{-1}) or very high (>117 mg kg^{-1}) range for M3P was 64%. Of 100 counties, 69 had more than 50% of agricultural soil samples tested in the High and Very High range; 37 had greater than 75% in this range (North Carolina Dept. Agric. Consumer Services, 2002). The accumulation of P in surface soils sampled for this study represents a potential risk for dissolved P loss that may contribute to degraded water quality. Numerous studies have linked increased STP levels with elevated losses of soluble P in runoff (Sharpley, 1995; Pote et al., 1999; Hooda et al., 2000; McDowell and Sharpley, 2001; Tarkalson, 2001).

Table 3 and Figure 2a illustrate the buildup of M3P in surface soils of the present study. Overall trends for M3P in surface soils (0-20 cm) confirmed the generalizations above, as clay soils held the most STP while organic soils held the least. The mean surface M3P of the sandy soil group (229 mg kg^{-1}) was well above the threshold of 100 mg kg^{-1}. Mean values for soil in the loam group were also above their respective P threshold at the surface but decreased much faster with depth than did sandy soils. The mean M3P value of clay soils never exceeded the P threshold for this soil group. However, examining Wilkes

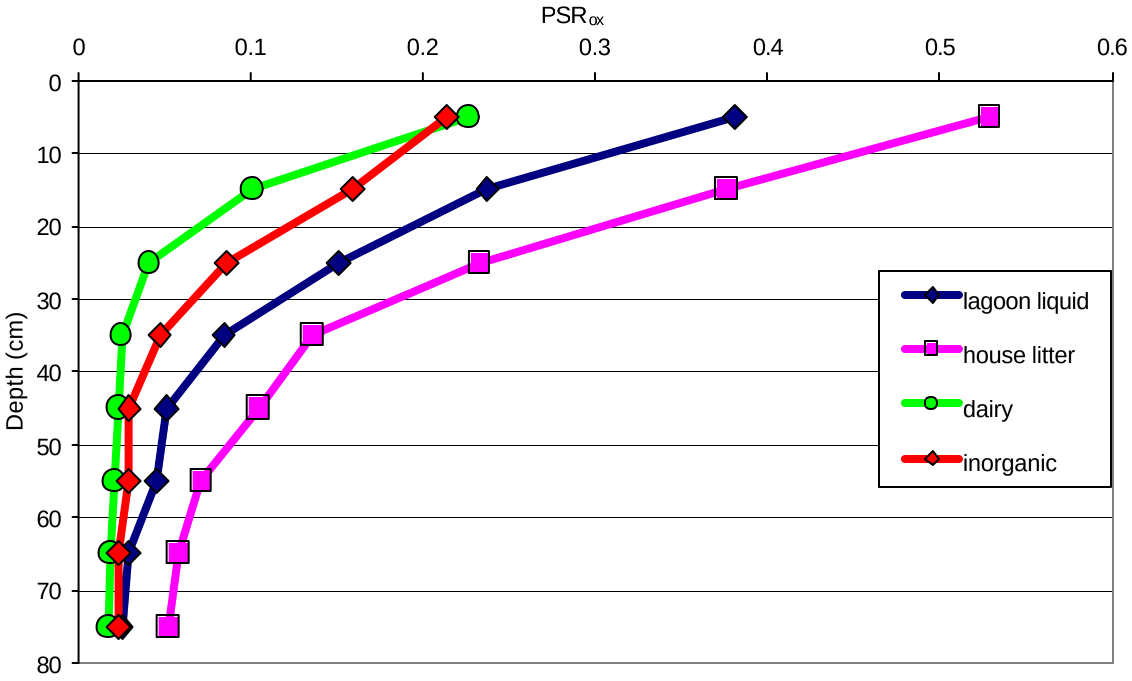

Figure 3b. Mean PSR$_{ox}$ levels with depth, segregated by P source type.

County individually, which consists of mostly clay soils and has a history of long-term application of poultry litter, illustrates that given enough time, even soils with high P retention can build up substantial levels of surface P (Figure 2b).

Figure 2a indicates that while most soils were not experiencing appreciable downward movement of P due to buildup of extractable P in the surface, trends with depth for sands and, to a smaller extent, loams show that transport of P via leaching is detectable and may become a future problem. At the deeper soil depths (>20 cm), sandy soils had a significantly greater amount of M3P (***P***<0.05) than did the other three threshold groups. Sandy soils have less P sorption capacity on Fe- and Al-oxides than finer-textured soils. Therefore, they would be expected to be more susceptible to soluble P movement downward through the soil profile as the surface horizon becomes P-saturated more rapidly. In general, most of the sandy soils in NC occur in the Coastal Plain, which is also the region where the majority of NC's agricultural land occurs. The low P sorption capacity of sandy soils creates a concern for the potential loss of soluble P through leaching in these areas.

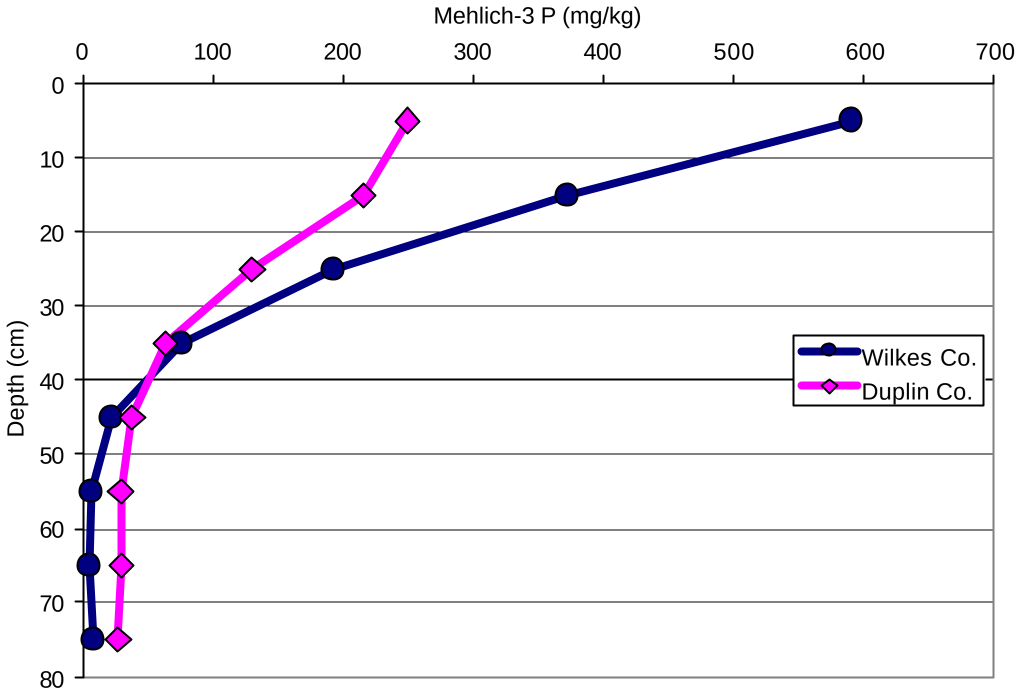

Figure 2b. Mean Mehlich-3-extractable P from sample sites in two counties with long-term P applications of poultry and swine waste.

Table 3. Selected properties of the 685 study soils. Values in parentheses are in mmol kg^{-1}.

	Soil Property	Range	Mean	Median	Standard Deviation
		Oxalate, mg kg^{-1} (mmol kg^{-1})			
All soils	P	4 – 2612 (0.13 – 84.34)	155 (5.01)	58 (1.87)	240 (7.75)
	Al	118 – 7840 (4.37 – 290.59)	1012 (37.51)	772 (28.61)	912 (37.51)
	Fe	3 – 3240 (0.05 – 58.02)	631 (11.30)	496 (8.88)	482 (8.63)
	PSR$_{ox}$,†	0.003 – 1.094	0.119	0.042	0.163
Topsoils	P	17 – 2612 (0.54 - 84.34)	390 (12.59)	285 (9.20)	349 (11.27)
	Al	118 – 5680 (4.37 – 210.53)	1032 (38.25)	762 (28.24)	961 (35.62)
	Fe	111 – 2884 (1.99 – 51.64)	762 (13.65)	668 (11.96)	486 (8.70)
	PSR$_{ox}$,†	0.019 – 1.094	0.286	0.246	0.204
Subsoils	P	4 – 822 (0.13 – 26.54)	76 (2.45)	29 (0.94)	107 (3.45)
	Al	181 – 7840 (6.71 – 290.59)	1006 (37.29)	776 (28.76)	896 (33.21)
	Fe	3 – 3240 (0.05 – 58.02)	587 (10.51)	460 (8.24)	473 (8.47)
	PSR$_{ox}$,†	0.003 – .682	0.063	0.023	0.096
		Mehlich-3, mg kg^{-1} (mmol kg^{-1})			
All soils	P	0 – 1294 (0 – 41.78)	76 (2.45)	9 (0.29)	140 (4.52)
	Al	195 – 2739 (7.23 – 101.52)	887 (32.88)	867 (32.13)	312 (11.56)
	Fe	9 – 343 (0.16 – 6.14)	93 (1.67)	74 (1.33)	63 (1.13)
	M3-PSR I‡	0 – 1.10	0.08	0.01	0.15
	M3-PSR II§	0 – 1.40	0.09	0.01	0.17
Topsoils	P	0 – 1294 (0 – 41.78)	206 (6.65)	155 (5.01)	199 (6.43)
	Al	195 – 2187 (7.23 – 81.06)	778 (28.84)	743 (27.54)	322 (37.51)

	Fe	29 – 343 (0.52 – 6.14)	154 (2.76)	146 (2.61)	62 (1.11)
	M3-PSR I‡	0 – 1.10	0.23	0.17	0.20
	M3-PSR II§	0 –1.40	0.30	0.19	0.23
Subsoils	P	0 – 490 (0 – 15.82)	32 (1.03)	0 (0)	72 (2.32)
	Al	293 – 2739 (10.86 – 101.52)	924 (34.25)	896 (33.21)	296 (37.51)
	Fe	9 – 274 (0.16 – 4.91)	73 (1.31)	57 (1.02)	50 (0.90)
	M3-PSR I‡	0 – 0.54	0.04	0	0.08
	M3-PSR II§	0 – 0.74	0.04	0	0.09

†PSR$_{ox}$ = ([P$_{ox}$] / [Al$_{ox}$ + Fe$_{ox}$]) in mmol kg^{-1}
‡ M3-PSR I = M3P / [M3Al + M3Fe]
§ M3-PSR II = M3P / M3Al

Figure 2c shows M3P trends when the study soils are segregated according to which type of P source was applied to the fields they were sampled from. Soils on which lagoon liquid was applied had a higher overall STP ($\boldsymbol{P}<0.05$) at all depths than did soils fertilized with inorganic P, but not than soils receiving dairy waste. Typically, swine farms in NC are lagoon operations in which lagoon effluent is sprayed onto pastures. Because of the higher content of soluble P in lagoon effluent, it is assumed to infiltrate rapidly into the soil surface as compared to other, less soluble animal waste types, such as dry poultry litter. We expected that fields receiving liquid forms of animal waste would exhibit the greatest downward movement of P. However, this trend was not evident (Figure 2c). Instead, soils receiving applications of dry litter, typically broiler house litter, had significantly greater M3P ($\boldsymbol{P}<0.001$) at subsurface depths (>20 cm) than did soils receiving other P sources. Sims et al. (2002) found substantial amounts of M3P in agricultural subsoils where long-term over-application of poultry wastes had occurred. Soils that received dry litter had significantly greater M3P levels ($\boldsymbol{P}<0.001$) at the soil surface (0-20 cm) also. Dry poultry litter has a greater content of total P than lagoon liquid or dairy waste (sludge or scraped) and is generally applied at higher rates. It appears that the greater P load of the poultry waste was the dominant factor controlling the level of both surface and subsurface M3P in surface rather than soil type, since poultry litter samples were obtained for a range of soil types in each of NC’s three physiographic regions.

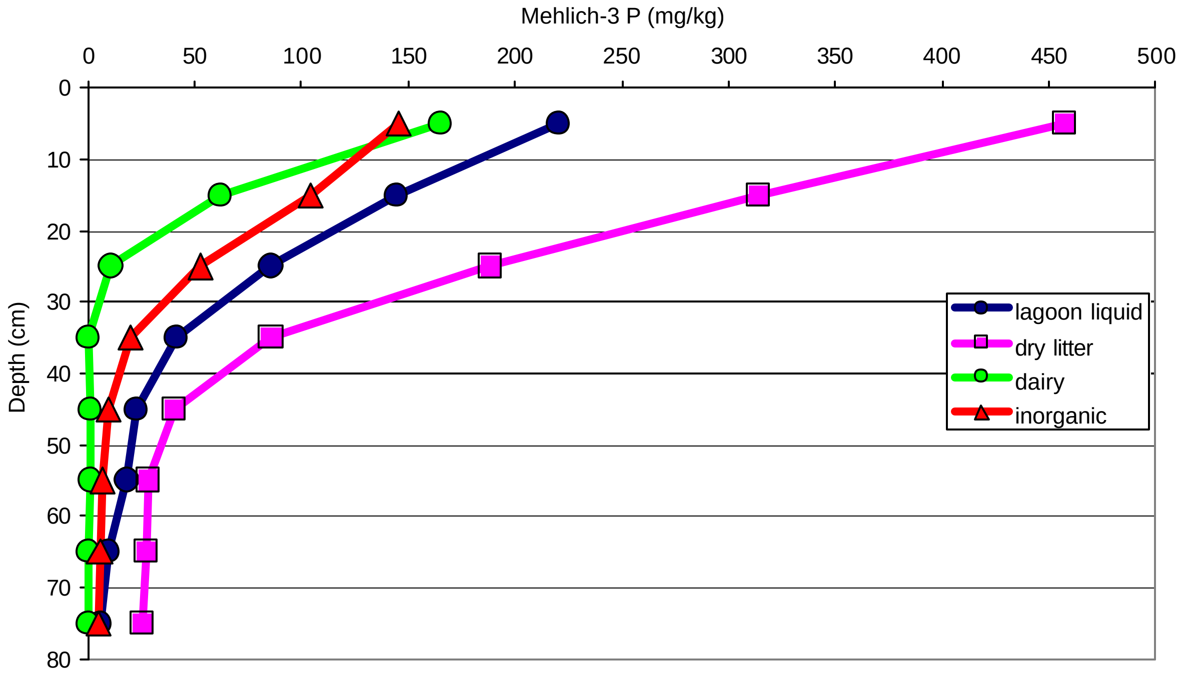

Figure 2c. Mean Mehlich-3-extractable P values with depth, segregated by P source type.

The mean PSR_{ox} of all samples was 0.12, while segregating into topsoils and subsoils yields mean PSR_{ox} values of 0.29 and 0.06. When segregated by soil group, trends observed in PSR_{ox} generally followed those of M3P described above, with saturation ratios being significantly greater ($\boldsymbol{P}$<0.001) at the soil surface and decreasing with depth (Figure 3a). When segregated by soil type, sands were more P-saturated ($\boldsymbol{P}$<0.001) in the subsurface horizon (20-80 cm) than all other soil types. This result shows that soils of the sandy threshold are more P-saturated than other soil groups. Because the exact histories of these sandy sample sites is unknown, it is impossible to conclude whether the greater P saturation is due to the effect of soil type, or that these sites have received a greater P load due to applications of animal waste, although both factors undoubtedly had an effect. The average PSR_{ox} of topsoil sands was 0.36, while they averaged 0.12 in the subsoils. The fact that these soils are highly saturated likely presents a future environmental problem as P moving downward may eventually come into contact with groundwater, which may contribute to deterioration of surface waters via subsurface drainage, especially if these fields continue to receive animal wastes. Even if these soils stopped receiving applications of animal wastes immediately, it is likely they would continue to experience P build up in the subsurface horizons for some time.

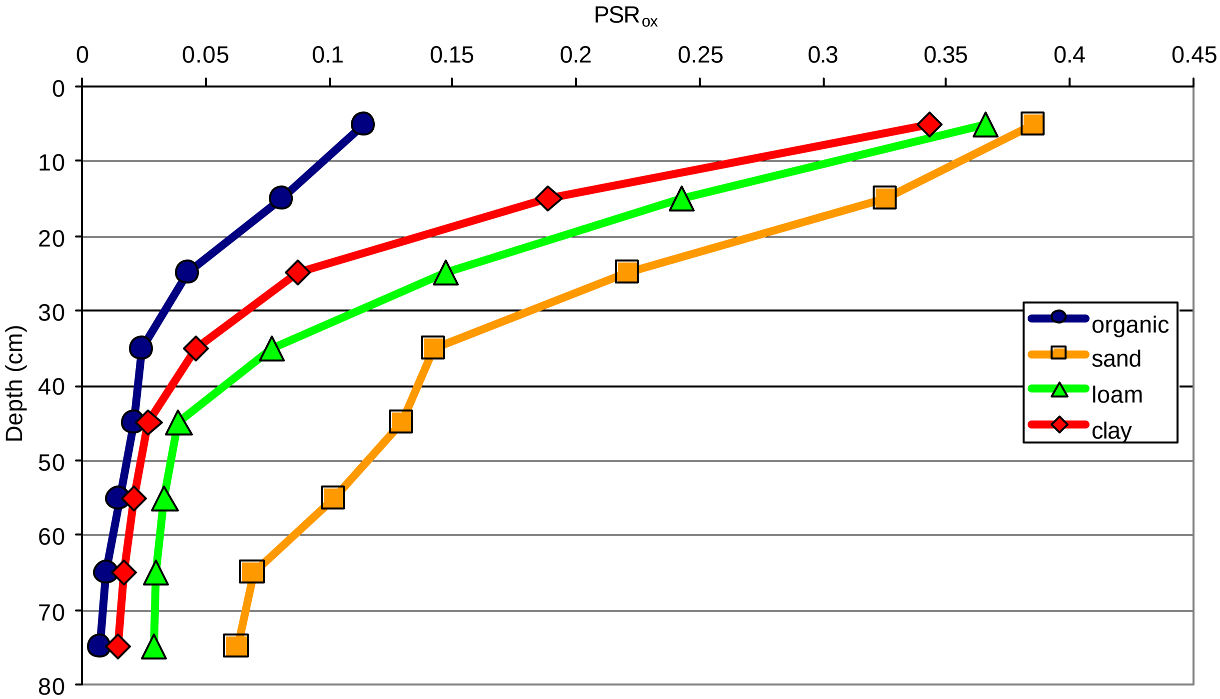

Figure 3a. Mean PSR$_{ox}$ levels with depth, segregated by soil group.

Soil types other than sand showed a greater reduction in PSR$_{ox}$ with increases in depth, as theoretically their finer-textured subsurface layers had a greater capacity to sorb P moving downward than did the coarse-textured sands. Organic soils had a significantly lower PSR$_{ox}$ than either loams ($\boldsymbol{P}$<0.001) or clays ($\boldsymbol{P}$<0.05). This result is not due to the fact that the soils in the organic threshold group had a significantly lower level of soil P, but rather that they had a much higher mean value for the quantity ($Al_{ox} + Fe_{ox}$), which, based on the definition provided in Equation 1, translates into a greater P sorption capacity. This finding and its implications are discussed in a later section.

Segregating samples by P source type showed that, again, trends of PSR$_{ox}$ mirrored those of M3P (Figure 3b). Soils on which dry litter had been applied had significantly greater P saturation ($\boldsymbol{P}$<0.001) in both topsoils and subsoils than soils receiving other types of manure or inorganic P, indicating the importance of P load.

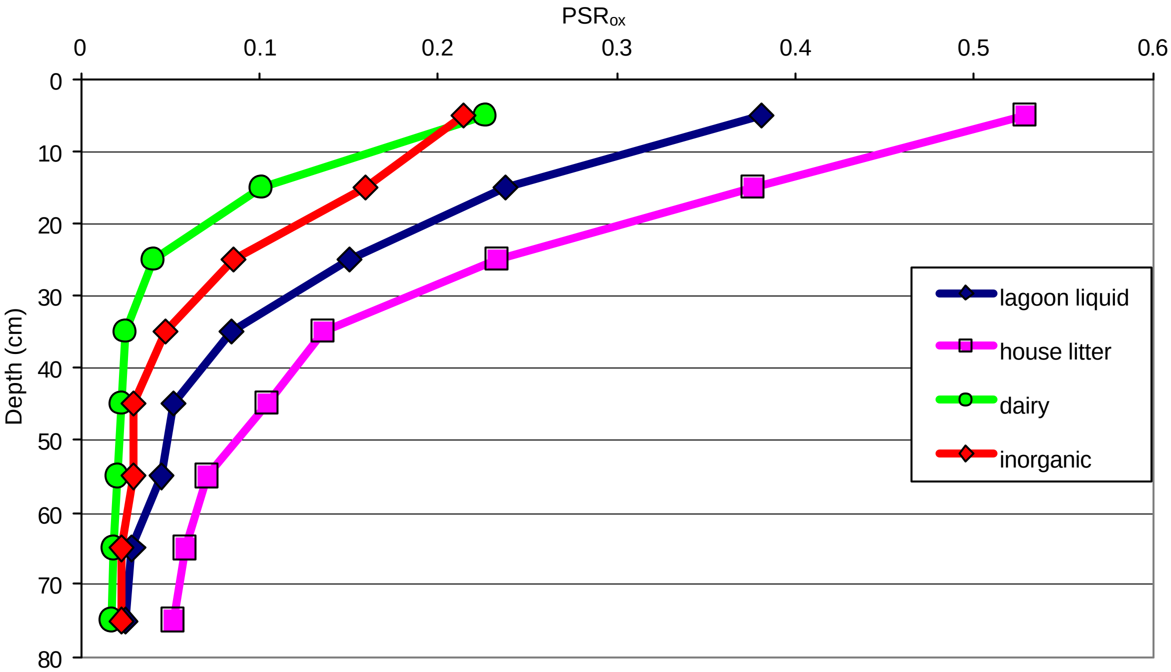

Figure 3b. Mean PSR_{ox} levels with depth, segregated by P source type.

Comparison of Mehlich-3 P and Oxalate Degree of Phosphorus Saturation

Because NC's Phosphorus Loss Assessment Tool (PLAT) relies heavily on M3P as a surrogate for soluble P, even though M3P was originally tested and implemented for agronomic purposes, we were interested in how M3P would relate to an environmental indicator such as DPS_{ox}.. Figure 4a shows that Mehlich-3 P was highly correlated to PSR_{ox} ($r^2 = 0.76$***) for the entire data set. The linear regression equation shown in Figure 4a indicates that, roughly, an increase of 10 mg kg^{-1} in M3P would cause a PSR_{ox} increase of 0.001. This rate of change is similar to retionships found between DPS_{ox} and M3P in other studies when normalized to DPS_{ox} (a=1.0), as summarized in Table 4. Note that the studies included in Table 4 used a measure of percent P saturation (DPS_{ox}), while we used a ratio. Two of the studies included a a value of 0.5 in their calculation of DPS_{ox}, while the remainder used a=1.0. Therefore, we normalized the results of their study to ours in order to compare the relationships between M3P and P saturation. The study by Kleinman and Sharpley (2002) evaluated a wide variety of soils from the National Soil Survey Laboratory archives in order to compare P sorption capacity components. The correlation between DPS_{ox} and M3P for all 37 acidic soils included in the study was weak (r^2=0.21). Ten topsoils involved in that

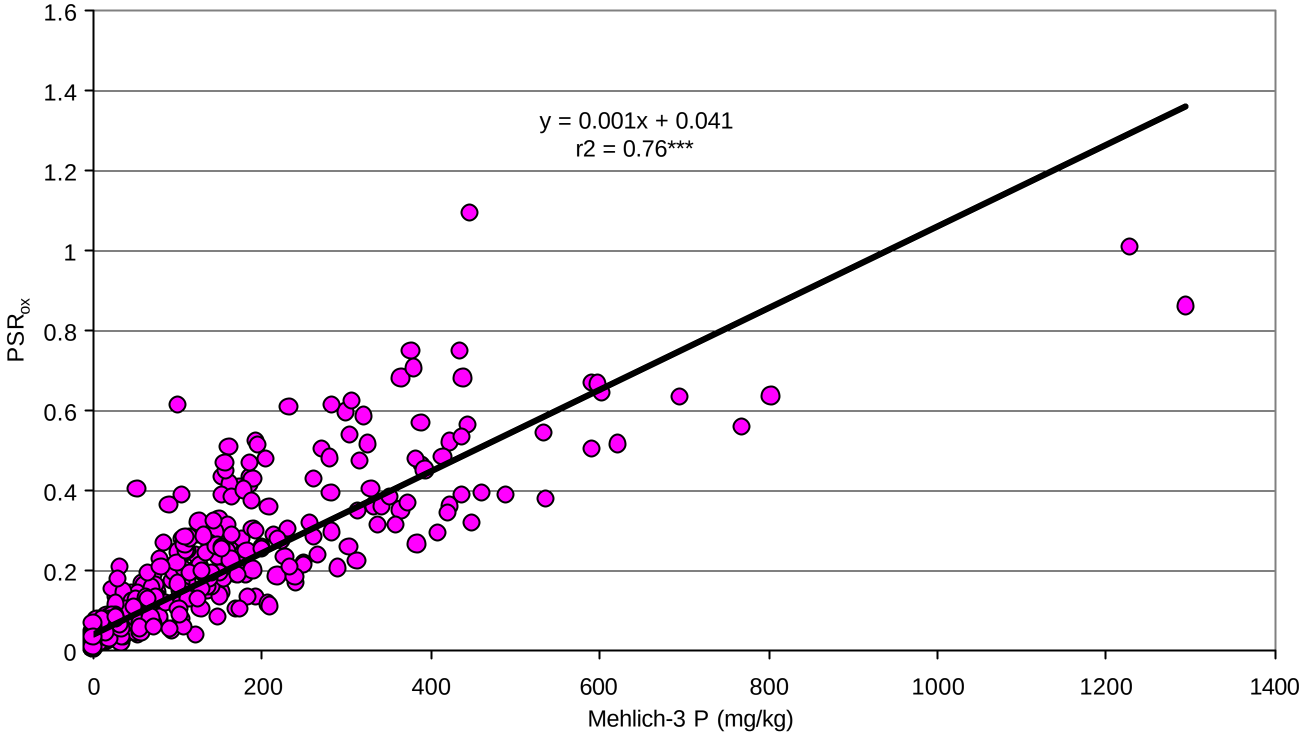

Figure 4a. Relationship between Mehlich-3 P and PSR_{ox} for all 685 study samples.

study are present in North Carolina and were singled out for regression analysis. In this case, STP was closely related to DPS_{ox} (r^2=0.92***; DPS_{ox}=0.12*M3P +2.1). The good relationship may be due to the fact that only one of the sandy soils, which tend to be the most variable of all soil groups, was included. Furthermore, when the 13 Ultisols from their study were segregated, the association between M3P and DPS_{ox} was also much improved from the whole data set (r^2=0.79; DPS_{ox}=0.13*M3P +2.5). Both of these findings suggest that grouping soils into generalized categories and setting critical limits on each will better assist in the prediction of soil P behavior (Beauchemin and Simard, 1999).

North Carolina employed the concept of P threshold groups as described earlier. These threshold groups are an integral component of NC's P indexing tool for sites requiring P-based nutrient management (Havlin et al., 2001). When soil samples from the present study were subdivided by soil P threshold group, all soil groups had significant relationships (***P***>0.001) between M3P and PSR_{ox} (Figure 4b-4f and Table 5). The correlation coefficient of this association improved from that of the whole data set in clays (r^2=0.76 to 0.91) and organics (0.76 to 0.93), while it worsened in sands (0.76 to 0.65) and loams (0.76 to 0.73). Smaller particle-sized soils produced greater slopes of the linear regression relating PSR_{ox} and M3P. This implies that coarser textured soils are more sensitive to changes in M3P, i.e. they become saturated more quickly, presumably due to their lower P sorption capacity (Fe_{ox} + Al_{ox}). Slopes of regression lines for each soil type were significantly different from each other (***P***<0.05) with the exception of sands and loams when examining the whole soil profile (Table 5). The organic soils had the smallest slope indicating that it would take a higher level of M3P to raise the PSR_{ox} of these soils by the same amounts as in the other soil groups (Figure 4b).

Table 4. Relationships between M3P and DPS_{ox} in various studies.

Reference	n	Site	Sample type	a	Regression Equation	r^2	Normalized slope¶
Current study	685	North Carolina	Topsoils and subsoils	1.0	PSR_{ox} = 0.001*M3P + 0.041	0.76	0.10
Sims et al., 2002	465	Delaware	Topsoils and subsoils	0.5	DPS_{ox} = 0.18*M3P + 12.2	0.72	0.36
Hooda et al., 2001	26	UK	Topsoils	1.0	DPS_{ox} = 0.06*M3P + 7.6	0.86	0.06
Kleinman and Sharpley, 2002	37†	NSSL‡ archives	Topsoils	1.0	DPS_{ox} = 0.14*M3P + 13.1	0.21	0.14
Kleinman and Sharpley, 2002	10	North Carolina §			DPS_{ox} = 0.12*M3P + 2.1	0.93	0.12#
Kleinman et al., 2000	59	New York	Topsoils	0.5	DPS_{ox} = 0.13*M3P + 7.1	0.91	0.26
D'Angelo et al., 2003	18	Kentucky	Topsoils	1.0	DPS_{ox} = 0.13*M3P + 3.1	0.91	0.13
Kleinman et al., 2003	72	Pennsylvania	Topsoils and subsoils	1.0	DPS_{ox} = 0.13*M3P + 2.0	0.90	0.13
							Mean ± standard deviation = 0.169 ± 0.104

†Only acidic soils from study included.

‡NRCS National Soil Survey Lab.

§Those soils from Kleinman and Sharpley (2002) that occur in North Carolina.

¶Normalized to PSR_{ox} and a = 1.0.

#Not included in calculation of overall mean ± standard deviation.

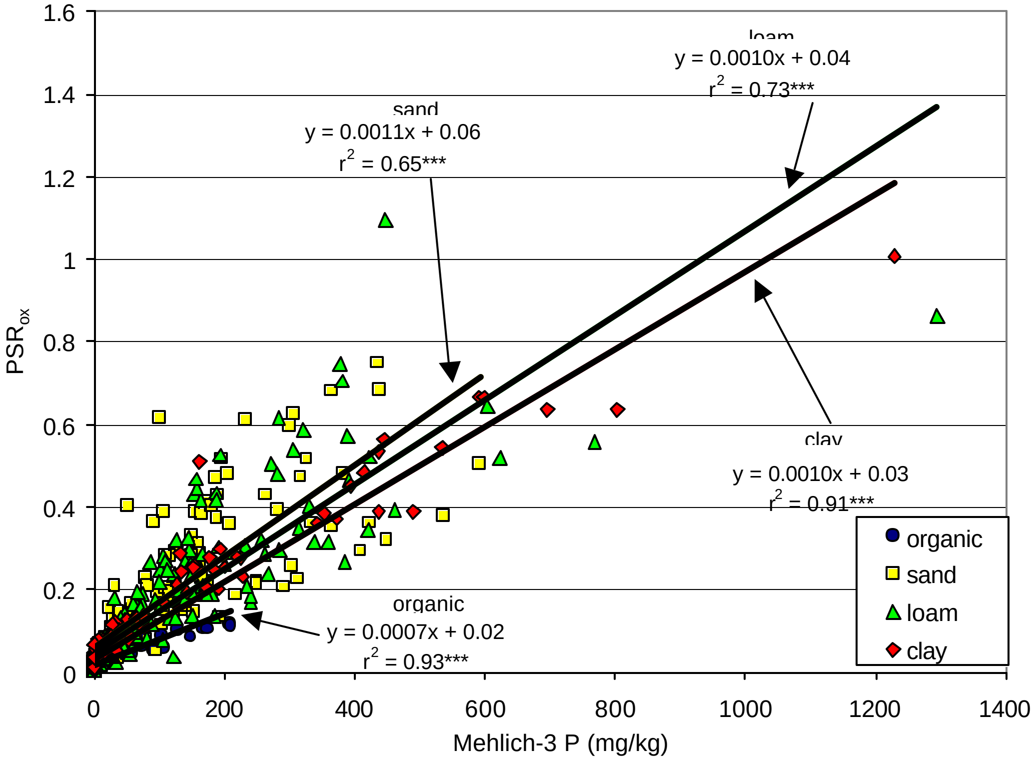

Figure 4b. Relationship between Mehlich-3 P and PSR_{ox}, segregated by soil group.

Separating into topsoils (0-20 cm) and subsoils (20-80 cm) showed that M3P was better at predicting PSR_{ox} in the subsoils than the topsoils (Figures 4c-4f). In every case, subsoil samples had higher slope values than topsoil samples, indicating that subsoils will become saturated at a lower M3P value than will topsoils (Table 5). Subsoils often contain greater amounts of Fe- and Al-oxides then surface soils. However, it is the crystallinity of oxide minerals that determines a soil's P sorption capacity (Borggaard, 1992). In general, subsoils have a lower ratio of poorly-crystalline to well-crystalline oxides and thus a lower P sorption capacity. These well-crystalline oxides tend not to dissolve in oxalate and P sorption capacity may, therefore, not follow trends in Fe_{ox} and Al_{ox}.

Table 6 illustrates the level of PSR_{ox} corresponding to the M3P threshold for each soil group. These data suggest that soils with a higher capacity to retain P can be saturated to a greater level before dissolved P reaches a critical concentration. It is important to note that P threshold values were assigned based on studies done on NC soils relating M3P and dissolved P in runoff. Therefore, these critical PSR_{ox} values hould be viewed with caution as they may not represent potential soluble P loss due to leaching. More

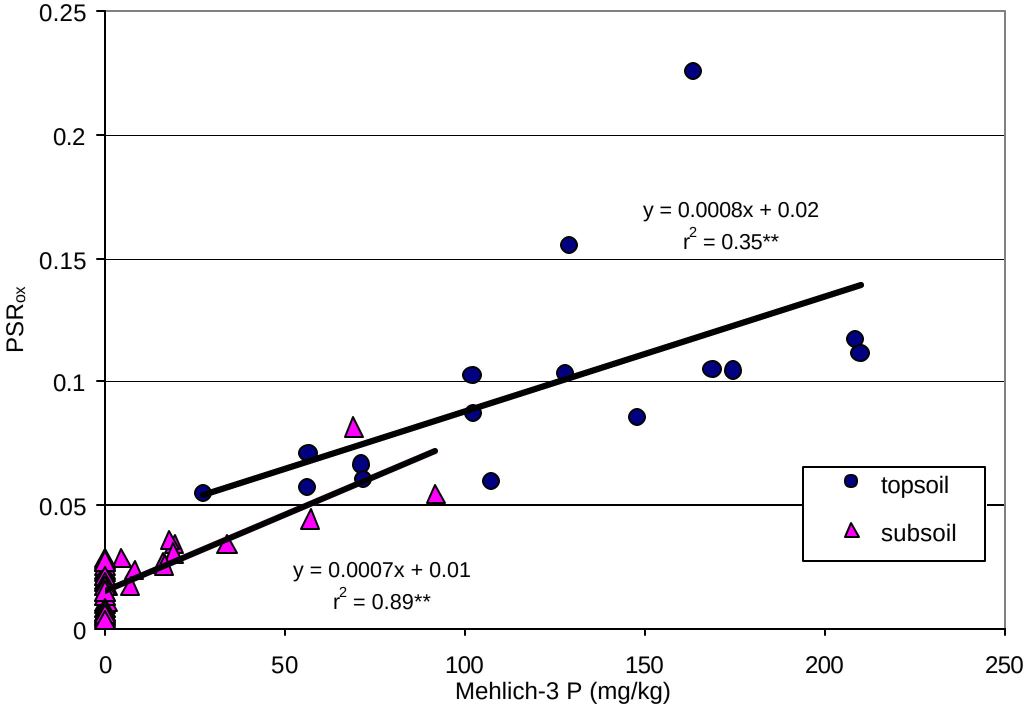

Figure 4c. Relationship between Mehlich-3 P and PSR_{ox} for soils in the organic soil group, segregated by topsoils (0-20cm) and subsoils (20-80cm).

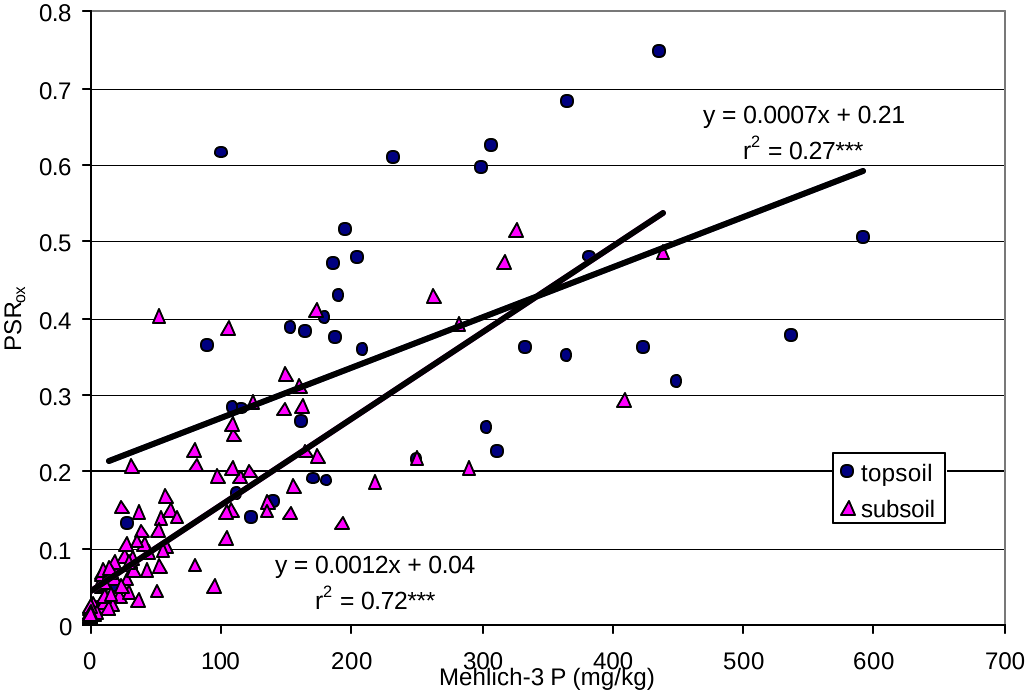

Figure 4d. Relationship between Mehlich-3 P and PSR_{ox} for soils in the sand soil group, segregated by topsoils (0-20cm) and subsoils (20-80cm).

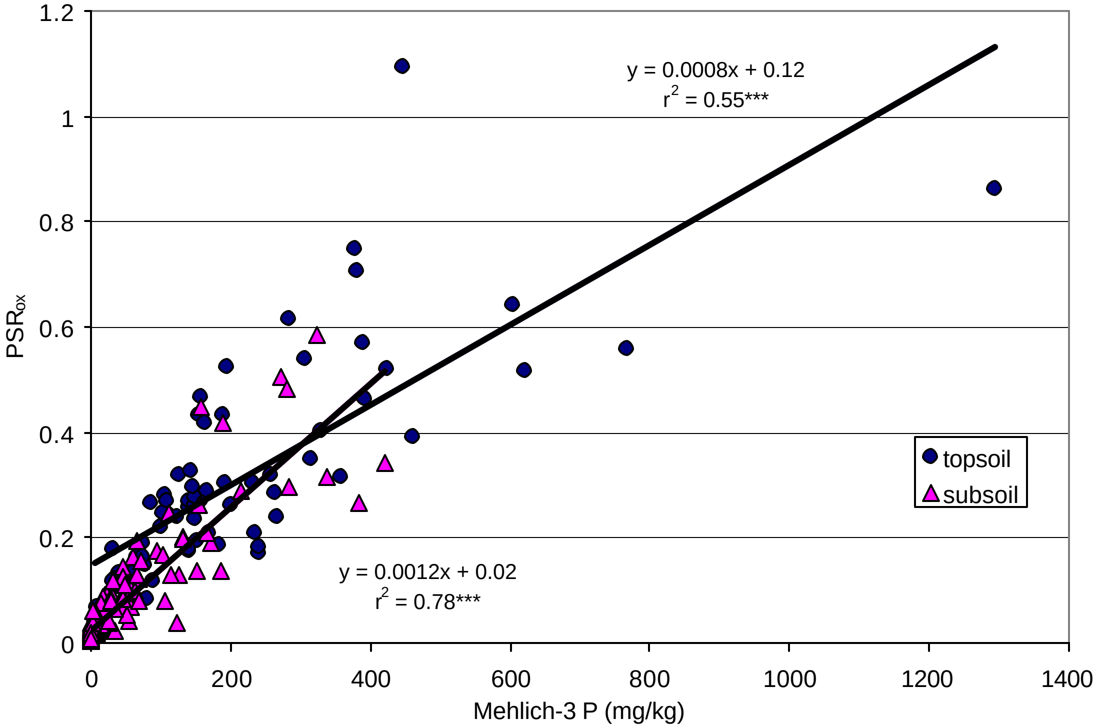

Figure 4e. Relationship between Mehlich-3 P and PSR$_{ox}$ for soils in the loam soil group, segregated by topsoils (0-20cm) and subsoils (20-80cm).

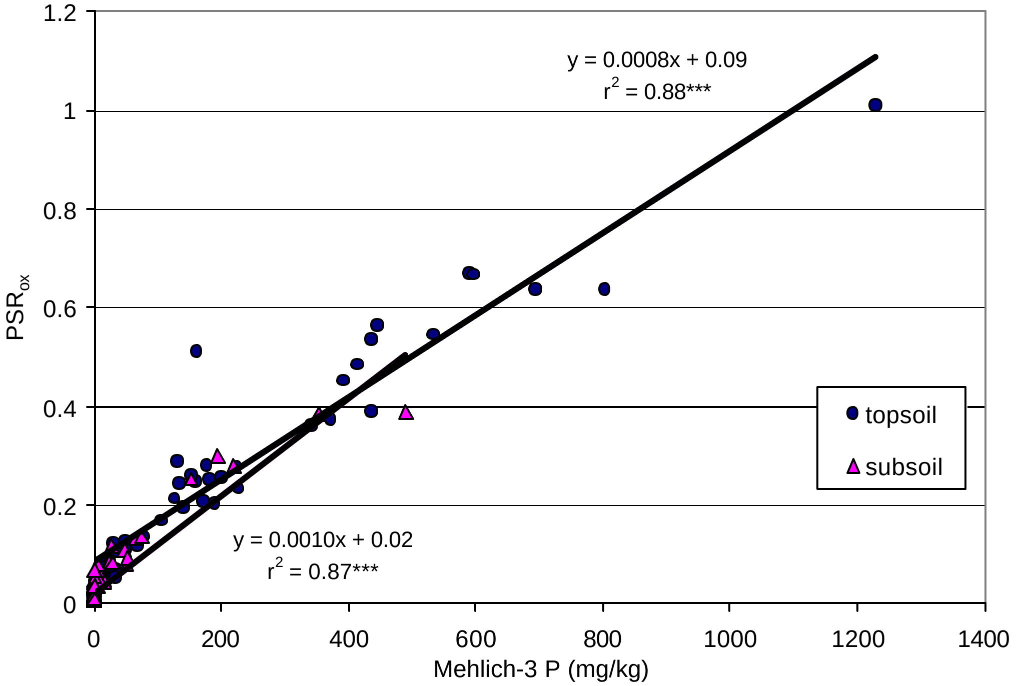

Figure 4f. Relationship between Mehlich-3 P and PSR$_{ox}$ for soils in the clay soil group, segregated by topsoils (0-20cm) and subsoils (20-80cm).

research, preferably in the form of batch studies, should be conducted in order to obtain PSR_{ox} values that directly assess soluble P loss for each soil type to compare with M3P threshold values.

Table 5. Relationships between M3P and DPS_{ox} for soils in North Carolina's P threshold groups.

Soil Group	**M3P threshold (mg/kg)**	**Sample set**	**r^2**	**Regression equation**
Organics	50	All	0.93***	PSR_{ox} = 0.0007*M3P + 0.02
		Topsoils	0.89**	PSR_{ox} = 0.0007*M3P + 0.01
		Subsoils	0.37**	PSR_{ox} = 0.0008*M3P + 0.02
Sands	100	All	0.65***	PSR_{ox} = 0.0011*M3P + 0.06
		Topsoils	0.27***	PSR_{ox} = 0.0007*M3P + 0.21
		Subsoils	0.72***	PSR_{ox} = 0.0012*M3P + 0.04
Loams	200	All	0.73***	PSR_{ox} = 0.0010*M3P + 0.04
		Topsoils	0.55***	PSR_{ox} = 0.0008*M3P + 0.12
		Subsoils	0.78***	PSR_{ox} = 0.0012*M3P + 0.02
Clays	500	All	0.91***	PSR_{ox} = 0.0010*M3P + 0.03
		Topsoils	0.88***	PSR_{ox} = 0.0008*M3P + 0.09
		Subsoils	0.87***	PSR_{ox} = 0.0010*M3P + 0.02

*** Significant at the 0.001 probability level.
** Significant at the 0.01 probability level.
* Significant at the 0.05 probability level.

Table 6. Critical PSR_{ox} levels at which the North Carolina PLAT P threshold occurs for each soil threshold group.

P Threshold Groups	**Critical PSR_{ox}†**
Organic	0.047
Sand	0.170
Loam	0.244
Clay	0.484

†Critical PSR_{ox} refers to the P saturation ratio that occurs at the respective soil group's threshold M3P according to linear regression equations in Table 5.

Comparison of Oxalate and Mehlich-3 as Extractants of P, Al and Fe

Mehlich-3 and oxalate extracted very different amounts of P, Fe and Al, with oxalate removing more than Mehlich-3 in all cases (Table 3). Mehlich-3 extracted 49% of P_{ox}, 88% of Al_{ox}, and 15% of Fe_{ox} when considering all soil depths. This result was very similar to that of Sims et al., (2002) who found that Mehlich-3 extracted 53%, 84% and 19% of P_{ox}, Al_{ox}, and Fe_{ox} respectively on 465 soils of Delaware's Coastal Plain. Ammonium oxalate is often used to obtain semi-selective dissolution of poorly-crystalline iron and aluminum oxides and organically bound Fe and Al, soil components found to be associated with a high P retention capacity (Borggaard, 1992; Freese et al., 1992; Gerke and Herman, 1992), while the Mehlich-3 extractant was designed as an agronomic measure of plant response. Mehlich-3's extracting constituents, fluoride and dilute acid, remove easily acid-soluble forms of P, calcium phosphates, and a portion of the aluminum and iron phosphates (Olsen and Sommers, 1982), while oxalate dissolves both non-crystalline and organically bound forms of Al and Fe (McKeague and Day, 1966). While Mehlich-3 and oxalate appear to have extracted similar amounts of Al, much higher amounts of Fe were extracted with oxalate. This finding suggests that the Mehlich-3 extractant was not as competitive as oxalate in terms of iron solubility. Similar results have been reported by Sims et al. (2002), Maguire and Sims (2002a), and Khiari et al. (2000). The lower P sorption capacity (PSC), i.e., lower Mehlich-3-extractable Fe (M3Fe), and to some degree Mehlich-3-extractable Al (M3Al), may explain the lower P removed by Mehlich-3, as the majority of P would be expected to be associated with Fe and Al in these soils.

The relationships between Mehlcih-3- and oxalate-extracted Al, Fe and P are shown in Figures 5-7. Good correlation was found between the two extractants for P (r^2 = 0.94) and Al (r^2 = 0.87), but Fe was only poorly correlated (r^2=0.44) to M3Fe. This observation has also been shown for certain Delaware soils by Sims et al., (2002), although Maguire and Sims (2002a) found a direct relationship between the amounts of iron extracted by Mehlich-3 and oxalate (r^2 = 0.93). Segregating into individual soil groups did not greatly improve the relationship between Fe_{ox} and M3Fe (data not shown).

Aluminum

Differences in levels of extracted Al, Fe, and P among the four soil threshold groups are shown in Table 7. General trends in extractable Al show that greater mean amounts of Al were removed with both extractants as the value of P threshold for the particular soil group increased (Figures 8 and 9). Soils in the organic group had significantly higher levels of Al (***P***<0.001) than other soil groups, as determined by both

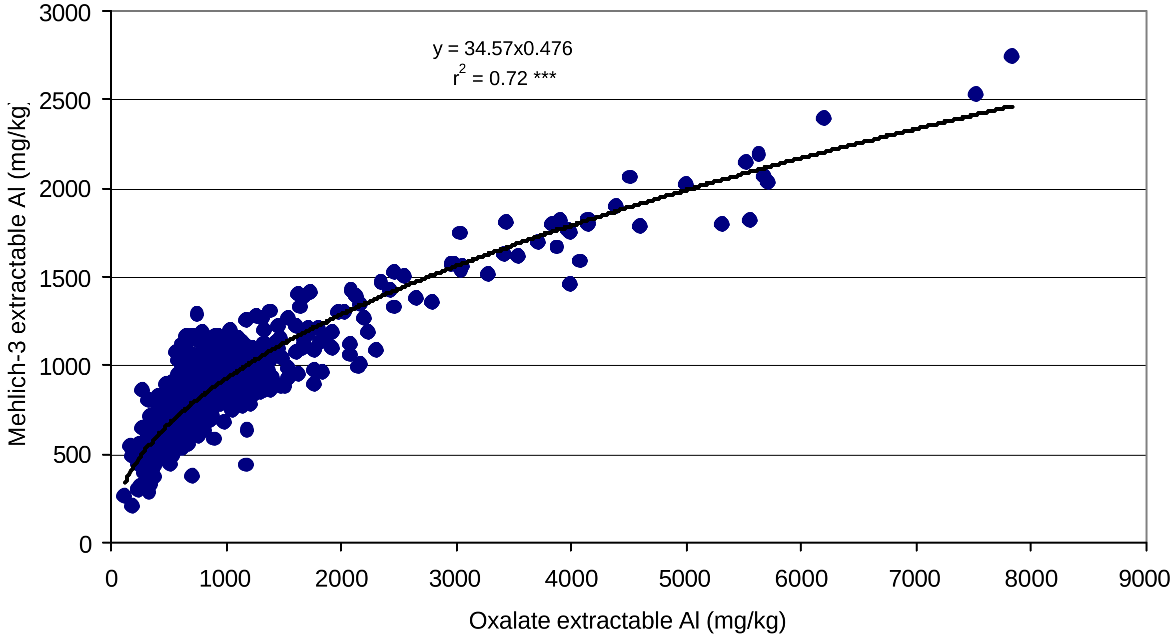

Figure 5. Relationship between oxalate- and Mehlich-3-extractable Al for all 685 study samples.

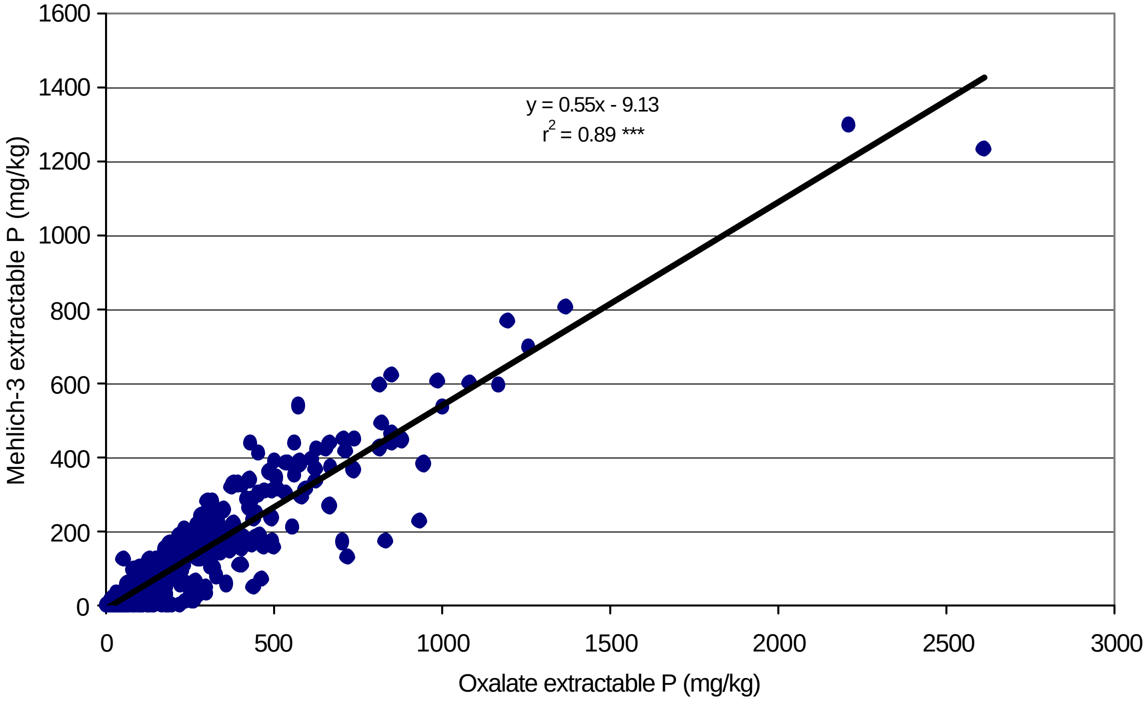

Figure 7. Relationship between oxalate- and Mehlich-3-extractable P for all 685 study samples.

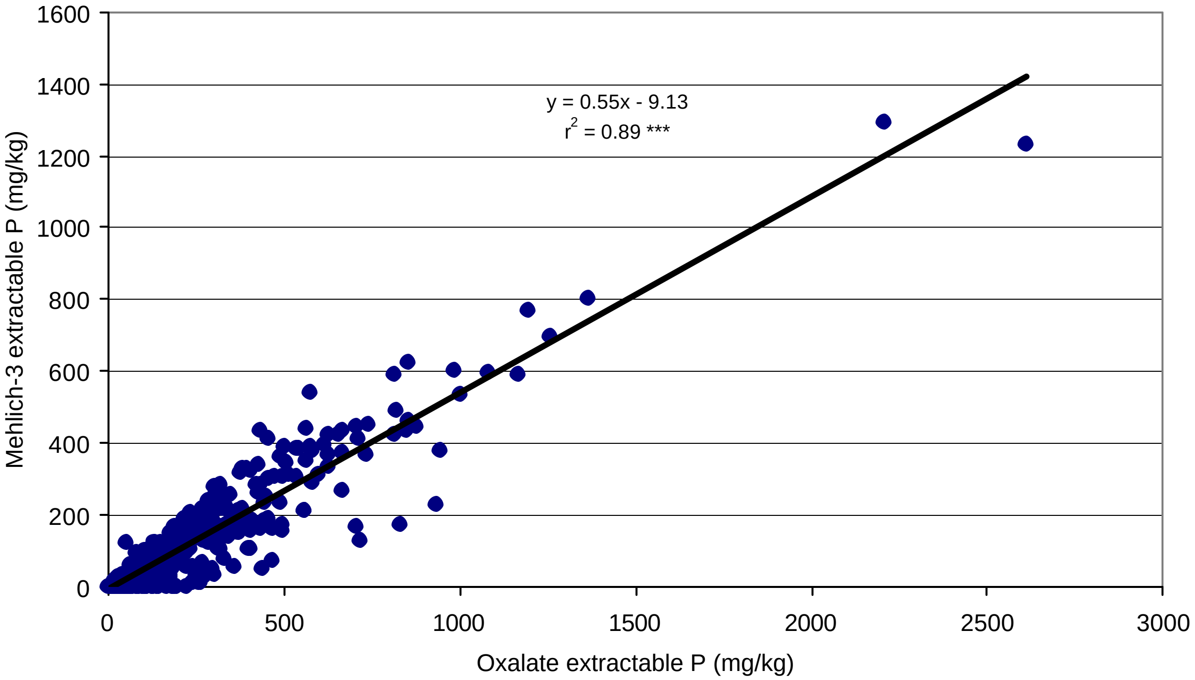

Figure 7. Relationship between oxalate- and Mehlich-3-extractable P for all 685 study samples.

extractants (Table 7). Several explanations could account for this finding. These soils were not deep organics (<66 cm) and would therefore have a supply of Al from the mineral matter that is further down in the profile. Vegetation being uprooted by wind, allowing mixing of mineral soil within the organic zone, as well as some deposition of mineral material, most likely would have occurred over time (Dolman and Buol, 1968). This occurrence, combined with the Al^{3+}-complexing power of organic matter, and the relatively low pH that exists in these soils due to the abundance of organic acids and the lack of historical lime usage, could thereby solubilize Al^{3+} from the available mineral matter. Additionally, it has been suggested that some organic acids can cause dissolution of minerals by forming soluble complexes with Al, thereby causing the release of Al^{3+} into solution and subsequent sorption to solid organic matter (He et al., 1999; Bhatti et al., 1998; Lan et al., 1995; Stumm, 1986). Other studies have indicated that the presence of organic ligands retards the formation of crystalline aluminum hydroxides, thereby keeping more Al in an amorphous form (Kwong and Huang, 1979; Kwong and Huang, 1977). One or a combination of these considerations could have been occurring. Other authors have found a correlation between Al_{ox} and organic Al, and although the exact mechanism was unknown, it is probable that the interaction will affect P retention in some manner (Lookman et al., 1996; Darke and Walbridge, 2000). McDowell and Condron, (2001) contemplated that whether this relationship will affect P sorption depends on the chemical composition and amount of organic matter. Clearly, more research on the interaction of extractable Al (and Fe) and organic matter is warranted for this group of soils in NC. In the case of mineral soils in our study,

Table 7. Mean values for soil elements extracted by Mehlich-3 and oxalate. Numerical values with the same letter are not significantly different from each other at the 0.05 probability level.

Threshold group	**P***	**Al**	**Fe**
	Mehlich-3 (mg kg^{-1})		
Sands	228 (107) A	717 A	107 A
Loams	207 (75) A	786 A	90 A
Clays	216 (65) A	976 B	67 B
Organics	120 (36) A	1471 C	153 C
	Oxalate (mg kg^{-1})		
Sands	360 (178) A	757 A	546 A
Loams	358 (139) A	763 A	568 A
Clays	458 (158) A	917 A	773 B
Organics	397 (162) A	2988 B	694 AB

*Values shown are for mean extractable P in topsoils (0-20 cm) only; mean extractable-P in soil profiles (0-80 cm) appears in parentheses; statistical comparison results are for topsoils;

clay soils yielded the highest amount of both extracted forms of Al, although only the M3Al content of clays was significantly higher (***P***<0.001). Shaw (2001) found that clays had higher amounts of amorphous Al and Fe oxides as compared to coarse soil fractions in Alabama Ultisols. Iron and aluminum oxides are associated with phyllosilicate minerals which are concentrated in the clay fraction.

Using a differential plot (Al_{ox} – M3P), it is clear that oxalate extracted significantly more Al (***P***<0.001) from soils in the organic threshold group (Figure 10). This difference between the two extractants was very pronounced, as oxalate extracted almost twice the amount of Al as Mehlich-3. This result indicates that oxalate was better able to extract Al associated with organo-Al complexes. Differences between amounts of Al extracted by the two extractants were not statistically significant in the other three soil groups, with the extractants removing essentially equal amounts of Al.

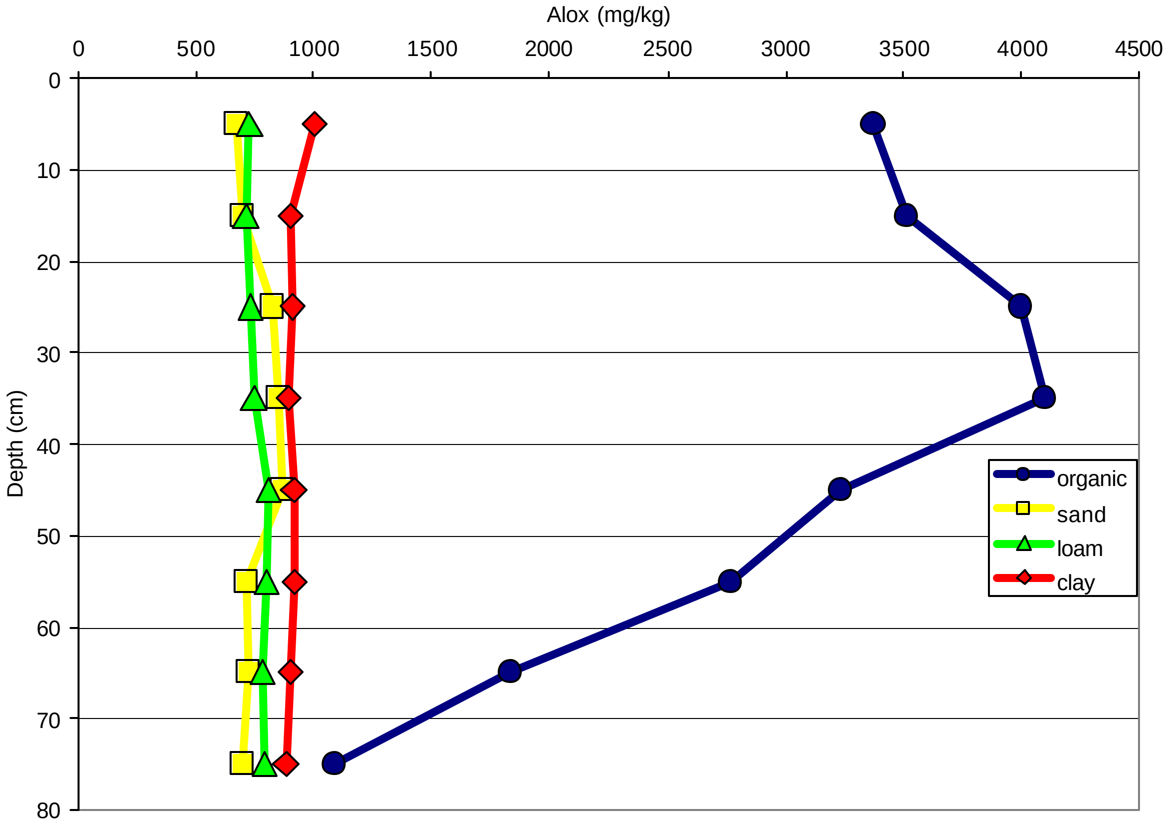

Figure 8. Oxalate-extractable Al levels with depth, segregated by soil group.

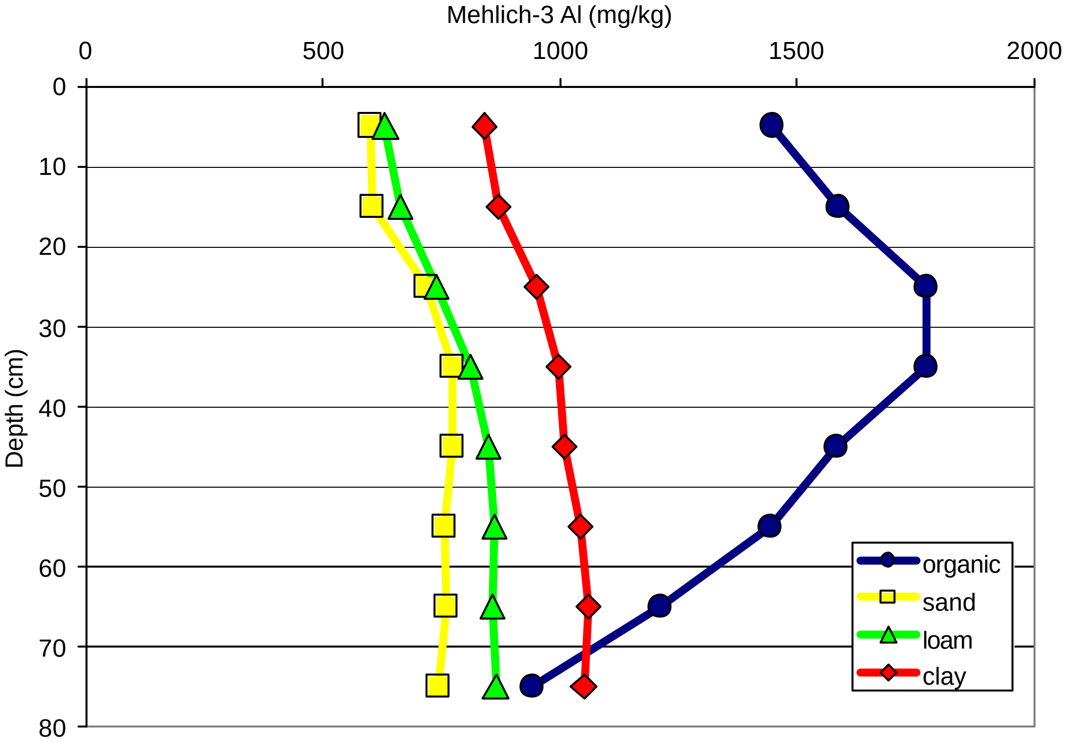

Figure 9. Mehlich-3-extractable Al levels with depth, segregated by soil group.

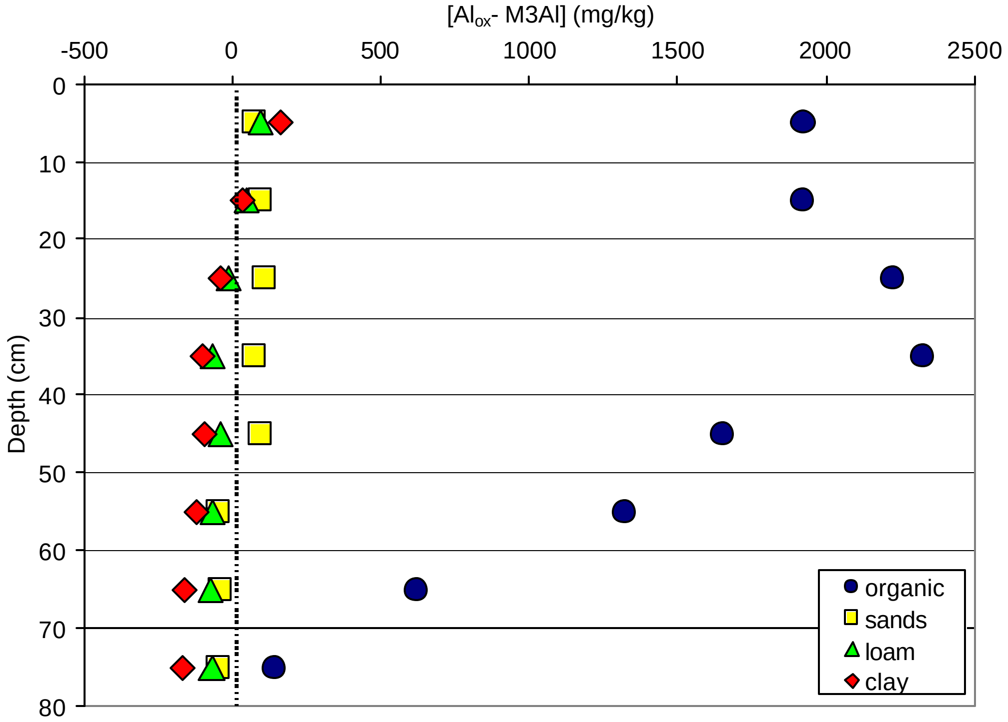

Figure 10. Differences in extractable Al between oxalate and Mehlich-3, segregated by soil group.

The difference between extractants in organic soil types was greatest at the upper soil layers (0-35 cm) and decreased with increasing depth. As mentioned above these soils were not deep organics and this finding seems to mimic the decrease in organic material and the subsequent increase of mineral matter that occurred at lower depths.

Similar amounts of Al were solubilized from sands at all depths by the two extractants. At greater soil depths (>55 cm) of sandy soils, the trend of oxalate extracting greater amounts of Al was reversed as M3Al was higher than Al_{ox}, although not significantly. This result also occurred in loams and clays but at much shallower depths (~15 cm). In the subsurface the effect of organic matter is minimal and the content of clay begins to increase. The majority of the sample soils in the loam subgroup and all those in the clay subgroup are Udults, tending to have highly weathered subsoils (Buol et al., 1997). Saunders (1964) found that Al_{ox} increased in soils that had a higher degree of weathering except in highly weathered soils, in which Al_{ox} decreased as compared to more moderately weathered soils. Mehlich-3 Al, on the other hand, increases with depth and seems to mirror the increase in clay content. It seems possible that Mehlich-3 extracted some Al from more crystalline oxides in highly weathered subsoils, while oxalate was selective for non-crystalline Al oxides. Mehlich-3 extracts Al by forming soluble complexes with fluoride, which has log K_{Al}'s of 6.98, 12.60, and 16.65 for metal-ligand ratios of 1:1, 1:2, and 1:3 respectively (Lindsay, 1979), while oxalate has log K_{Al}'s of 6.10, 11.09, and 15.12 (Martell and Simth, 1977; Lindsay, 1979). Flouride has a very strong affinity for Al^{3+} and has been found able to remove the nonbridged OH^- groups at clay edges as well as breaking the interior Al-OH-Al linkages, thereby destroying the $Al(OH)_3$ structure (Hsu, 1989). Our finding that Mehlich-3 extracts greater quantities of Al than oxalate, especially in finer-textured soils, suggests that Mehlich-3 is, to some extent, extracting crystalline Al-oxides. The other possible alternative is that oxalate is leaving behind non-crystalline Al that is more protected in nodules or concretions while Mehlich-3 is better able to penetrate these areas (Borggaard, 1992; Guo and Yost, 1999).

Iron

Overall, less iron was extracted from the study soils as compared with Al, the effect being more pronounced in Mehlich-3-extracted soils. Oxalate was able to extract up to four times more iron than was Mehlich-3, up to four times more (Table 7). While stability constants for aqueous complexation of Al^{3+} with flouride and oxalate are relatively similar, the Fe solubility constants for flourine, ($\log_{Fe}K$'s equal to 6.00, 9.20 and 11.70 for ML, ML_2 and ML_3 respectively) are lower than that of oxalate (7.53, 13.64, 18.49) (Martell and Smith, 1977; Lindsay, 1979). The disparity between extractable soil Fe and Al was significantly greater ($\boldsymbol{P}<0.001$) when Mehlich-3 was the extractant as opposed to oxalate, as demonstrated by Al:Fe molar ratios (Table 8). With the exception of the organic soil group, particle size distribution affected Mehlich-3- and oxalate-extractable Fe in opposite manners; Mehlich-3 extracted relatively more Fe as particle size increased, and oxalate extracted more Fe in finer-textured soils (Figures 11 and 12). The fact that Mehlich-3-extractable Fe increased as soil texture became coarser is counter-intuitive to the notion of smaller sized particles having more surface area and thus more poorly-crystalline metal oxides (Shaw, 2001). This result was most likely due to the mechanism by which Mehlich-3 extracts Fe. Because it was developed for use in acidic soils high in Al^{3+}, the designers were more interested in the ability to extract Al. acid that is a component of the Mehlich-3 extractant. The neutralization of the extracting solution makes iron less soluble, leaving EDTA to be a weaker extractor of Fe (Kamprath, E.J., personal communication).

Table 8. Mean Al:Fe molar ratios of threshold group soils. Numerical values with the same letter are not significantly different from each other at the 0.05 probability level.

Threshold soil group	**Mehlich-3 Al:Fe (mmol kg^{-1})**	**Oxalate Al:Fe (mmol kg^{-1})**
Overall average	**31.4:1**	**4.9:1**
Sands	**19.4:1 A**	**5.3:1 A**
Loams	**29.4:1 B**	**3.7:1 A**
Clays	**44.6:1 C**	**2.8:1 A**
Organics	**29.2:1 B**	**14.9:1 B**

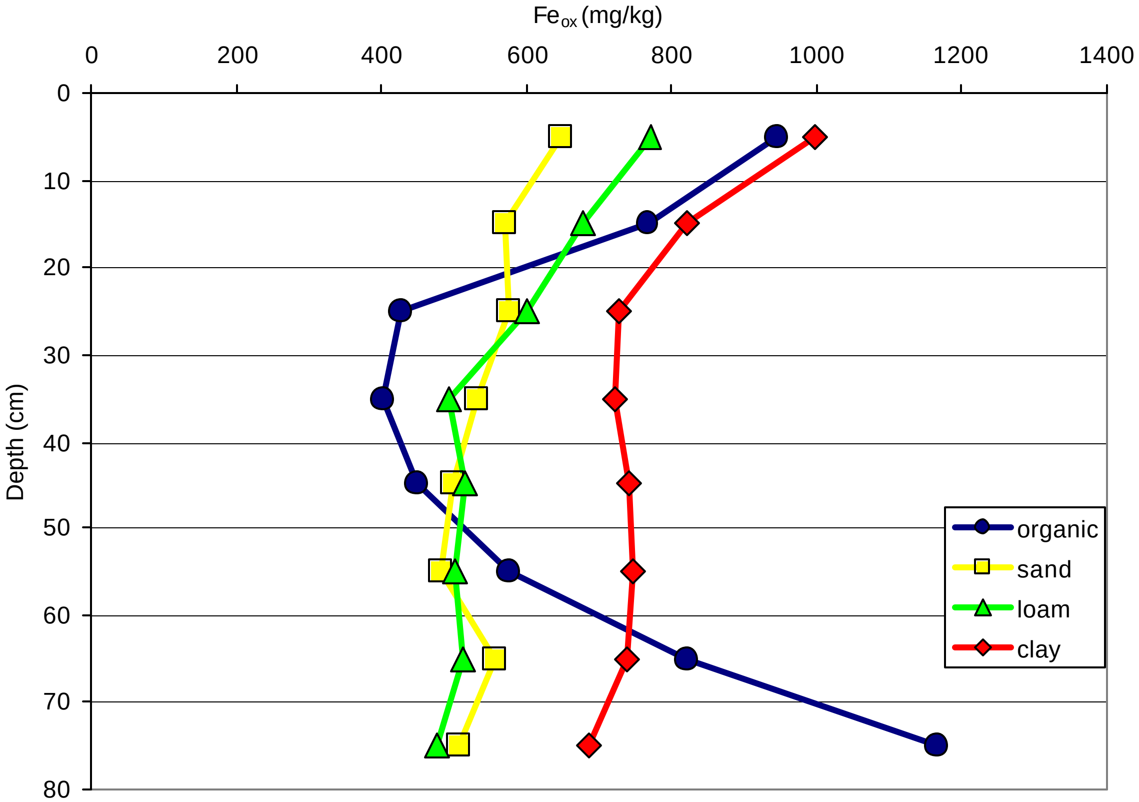

Figure 11. Oxalate-extractable Fe levels with depth, segregated by soil group.

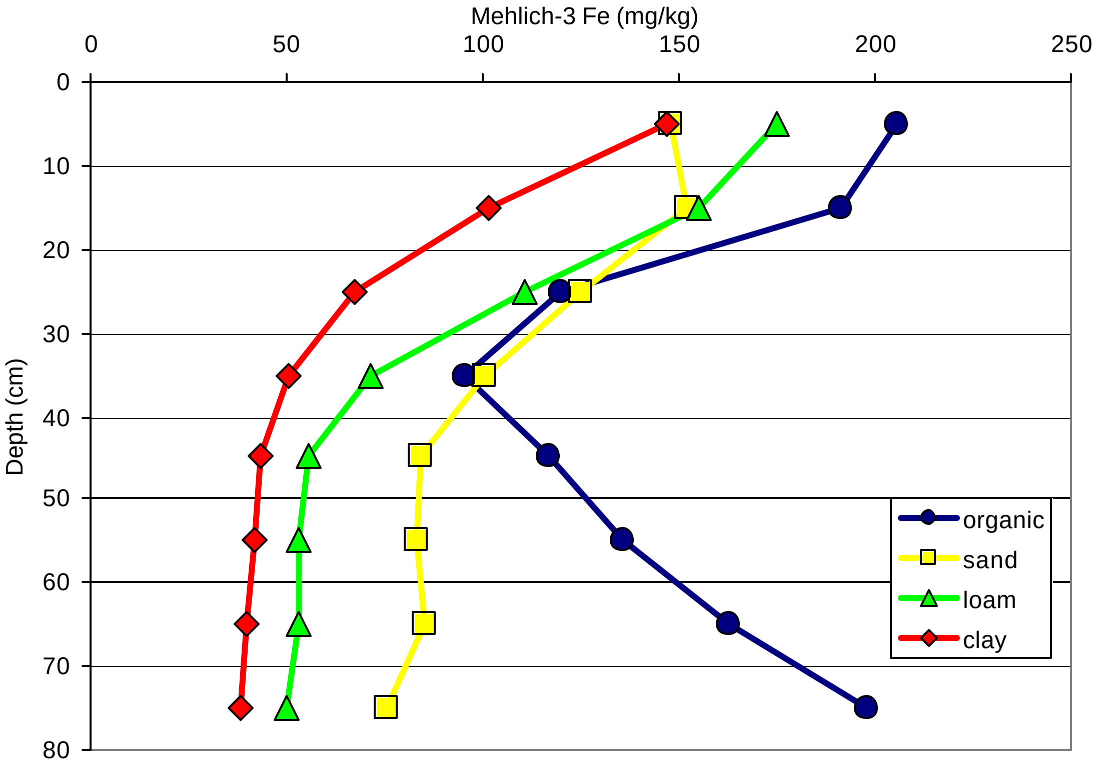

Figure 12. Mehlich-3-extractable Fe levels with depth, segregated by soil group.

Consequently, Mehlich-3 extracted a greater amount of Fe from sands than from loams (***P***<0.01) or clays (***P***<0.001), and more Fe from loams than from clays (***P***<0.001).

Acid ammonium oxalate extracted significantly more (***P***<0.001) Fe from clay soils than the other two mineral soil groups (Figure 11). All soil groups, with the exception of organics, significantly decreased in both forms of extractable Fe (***P***<0.001 for Mehlich and ***P***<0.01 for oxalate) between surface soil depths (0-10 and 10-20 cm) to the deepest six soil depths (>20 cm). Lookman et al. (1995) also found that Fe_{ox} decreased with soil depth. Significantly greater amounts of Fe were extracted in the A horizon than the B horizon. This pattern was especially noticeable in the clay soil group, which decreased by 400 mg kg^{-1} from topsoil to subsoil when extracted with oxalate. Rather than the amount of iron oxides decreasing with depth, it is likely that the reactivity of the iron oxides decreased, i.e. the ratio of non-crystalline iron oxides to crystalline iron oxides or organically-bound Fe decreased. This is presumably why there was no correlation between clay content and extractable Fe (or any other components measured in this study).

The amount of extractable-Fe in organic soils was much different for both extractants when compared to the mineral soils (Figures 11 and 12). The reason for the relatively high Fe_{ox} at the soil surface of these soils is unknown, although in a study conducted on organic soils from this region of NC, Dolman and Buol (1968) found that shallow organics had a "mull" top horizon upon cultivation, i.e. a layer of mixed organic matter and mineral soil. In addition, a "chemical ripening" is known to occur in these soils following improved drainage, which, once the soils become aerated, allows formation of new organic and organo-mineral compounds (Dolman and Buol, 1968), presumably increasing the reactivity of Fe-oxides and the amount of organic matter-bound Fe. The decrease in extractable Fe in the middle portion of these soils' profiles may be explained by the presence of less organic matter and more oxidizing conditions as the organic matter content gradually decreases and mineral matter increases. The mean level of extractable Fe in these soils increased sharply toward the bottom of the organic soil profile (>50 cm), which may suggest more reducing conditions at these depths. The exact mechanism producing the observed trends in oxalate suggested; ii) before these soils were drained, reductive dissolution of Fe (III) minerals occurred and soluble Fe^{2+} was removed by leaching; or (iii) there exists a population of Fe-organo complexes that are insoluble even in oxalate. It is also possible that a combination of (i) and (ii) occurred, but it is obvious that more study is needed to fully understand this system. However, Figures 11 and 12 seem to support the theory of drainage controlling Fe and imply a loss of soluble Fe through leaching or greater Fe-oxide crystallinity in the middle portion of the soil profile.

Due to the inability of Mehlich-3 to extract Fe, the differences between the two extractants (Figure 13) No correlation between drainage (poorly-drained versus well-drained soils) and extractable Fe was found in our samples. However, four soil types, all of which occurred as floodplain soils in the Mountains region contained the greatest levels of $Fe_{ox}$of all samples. These soils most likely consisted of soil materialthat has been eroded from side slopes and so contain a fair amount of mineral iron. Once in the alluvial areas, they are subject to redox processes keeping them in non-equilibrium. Fluctuating cycles of reduction and oxidation promote the solubilization of ferrous iron and subsequent reoxidation into new poorly-crystalline minerals containing ferric iron (Young and Ross, 2001). Other studies have found higher Fe_{ox} in followed the pattern of oxalate-extractable Fe in Figure 11. The difference between extractants was greatest in clays, significantly more so than loams or sands (***P***<0.001). Differences between the extractants for Fe in loams,

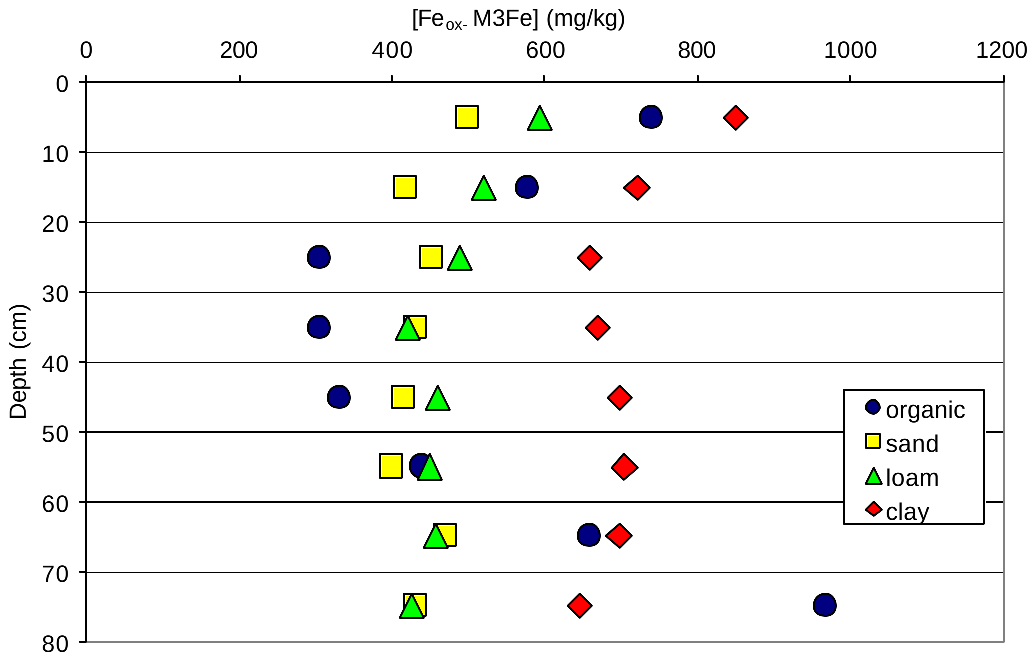

Figure 13. Differences in extractable Fe between oxalate and Mehlich-3, segregated by soil group.

sand and organics were not significantly different from each other. Mehlich-3-extractable Fe decreased while Fe$_{ox}$ increased with smaller particle size, and the magnitude of difference between the extractants increased with decreasing particle size. Oxalate-extractable Fe probably represents a more accurate indication of what is actually occurring in respects to amorphous Fe oxides in this collection of soils. This result is important because sandy Coastal Plain soils in NC generally have lower relative iron oxide contents as opposed to Piedmont and Mountain soils (clays) that have been weathered in-place. This is believed to be due to silicate weathering prior to erosion from the piedmont and soluble Fe^{2+} removal at the time sediments were deposited from depositional areas (J.P. Lilly, http://agronomy.agr.state.nc.us/sssnc/13jplilly.htp).

No correlation between drainage (poorly drained soils) and extractable Fe was found in the present samples. However, four soil types, all of which occurred as floodplain soils in the Mountains region were the highest in Fe$_{ox}$ of all soil samples. These soils had no B horizon and would therefore not be expected to contain large amounts of Fe-oxides. However, they most likely consist of soil material that has been eroded from side slopes and so contain a fair amount of mineral iron. Once in the alluvial areas, they are subject to redox processes keeping them in non-equilibrium. Fluctuating cycles of reduction and oxidation promote the solubilization of ferrous iron and subsequent reoxidation into new poorly-crystalline minerals containing ferric iron (Young and Ross, 2001). Other studies have found higher Fe_{ox} in alluvial areas as opposed to other landscape positions and attributed this result to the complex redox processes (Lookman et al., 1995; Lookman et al., 1996; Darke and Walbridge, 2000). Much is still unknown about how these conditions affect P retention.

Phosphorus

Overall, oxalate extracted more P than Mehlich-3 ($\boldsymbol{P}<0.001$), between 1.6 and 3.3 times as much. The extractable P values reported in Table 7 represent topsoil samples (0-20 cm), while average P levels for the entire soil profile (0-80 cm) are indicated in parentheses. Results of multiple mean comparisons shown are for extractable P in soil surfaces (0-20 cm) and reveal that no significant differences between soil types occurred for either M3P or P_{ox}. Mehlich-3 extracted significantly more P from subsoils of sands than from all other soil types ($P<0.001$) while P_{ox} was greater ($p<0.001$) in sands than both loams and clays but not organics (Figure 2a and Figure 14). This occurrence of higher subsoil extractable P in coarser-textured soils again illustrates the lower P sorption capacity of these soils, as surface extractable P values were lower than that in other soil groups. Once the finite capacity to retain P in the surface is reached, downward movement of soluble P occurs. In the case of both extractants, subsoils had significantly less extractable P ($\boldsymbol{P}<0.001$) than topsoils.

The difference between the ability of Mehlich-3 and oxalate to extract P is shown in Figure 15. The magnitude of difference between extractants was significantly higher ($\boldsymbol{P}<0.001$) in the topsoils (0-20 cm) than in the subsoils (20-80 cm). General trends for topsoils reveal that as particle size decreased the disparity between the two extractants increased. Overall, Mehlich-3 P extracted 60% of P_{ox} in sands, 54% in loams, 41% in clays and only 22% in organic soils. The magnitude of extractable P differences in clays was significantly larger ($\boldsymbol{P}<0.01$) than that of loams or sands in the surface (0-20 cm). The difference between P_{ox} and M3P was significantly greater ($\boldsymbol{P}<0.001$) in organic soils than in all other soils types. This suggests that oxalate is more effective than Mehlich-3 in solubilizing organically complexed Al and Fe.

Fox and Kamprath (1971) found in a column leaching study that most of the applied aqueous P readily

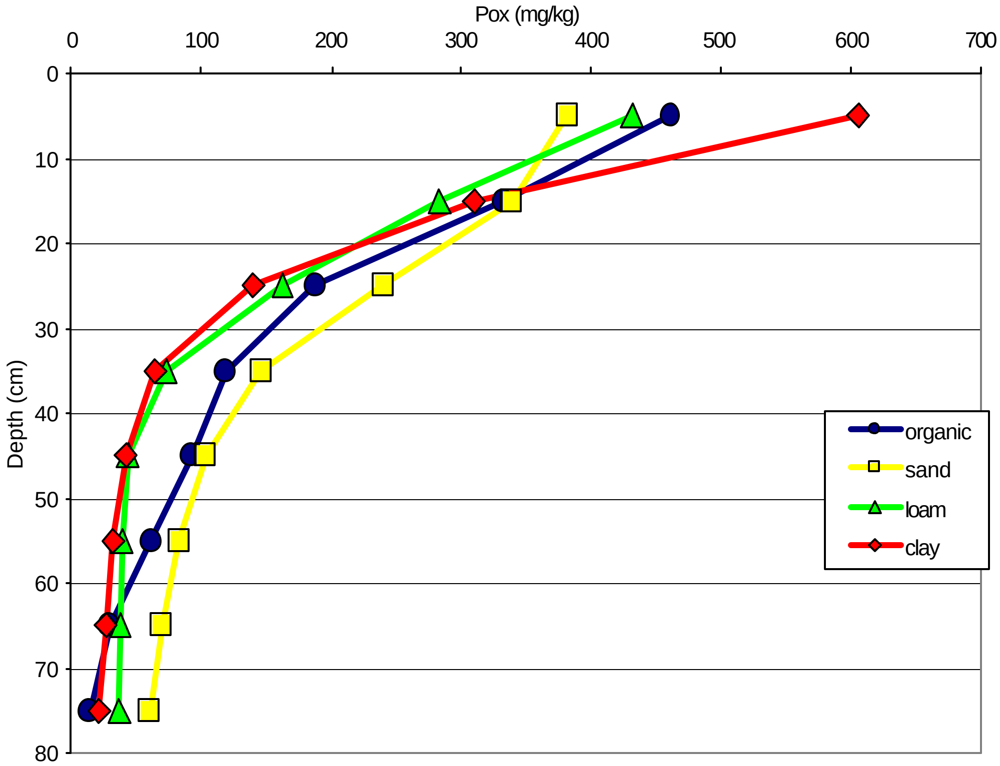

Figure 14. Oxalate-extractable P levels with depth, segregated by soil group.

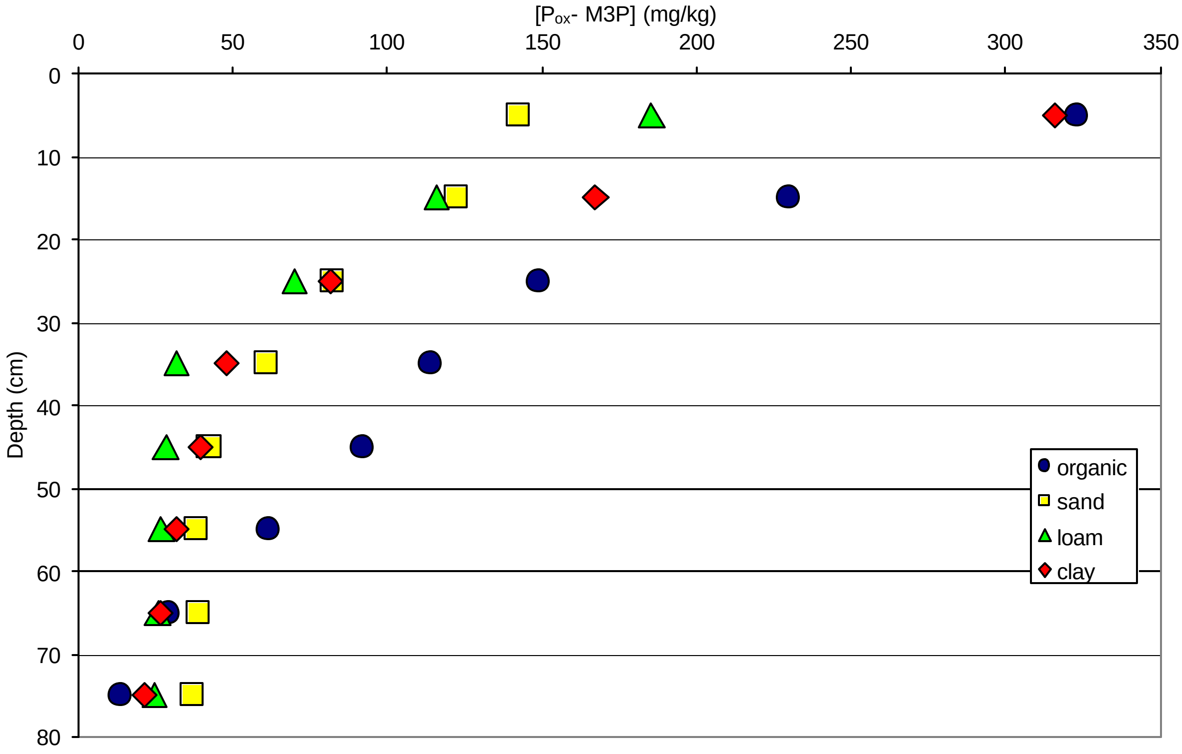

Figure 15. Differences in extractable P between oxalate and Mehlich-3, segregated by soil group.

leached from an acid organic soil, suggesting that P is held weakly in these soils. Although some studies have indicated an increase in P sorption in the presence of organic ligands (Kwong and Huang, 1979; Gerke and Hermann, 1992; Darke and Walbridge, 2000), Fox and Kamprath's (1971) study suggests that NC organic soils probably experience ligand-exchange reactions between organic acids and added phosphate on Fe and Al surfaces (Stumm, 1986; Violante et al., 1991; Bhatti et al., 1998; He et al., 1999; Liu et al., 2001). The trend found in the current study suggests that these soils are capable of retaining extractable P, just not in Mehlich-3 extractable form. Whether this finding indicates that P associated with organic matter in these soils becomes somehow protected and what this means for the availability of P in these soils remains unknown. A higher level of (Fe_{ox} + Al_{ox}), implies that these soils should be able to sorb large quantities of P. Based on Fox and Kamprath's (1971) study, however, organic soils of NC have a relatively low ability to retain plant available P. Clearly, P saturation as measured by the [P_{ox} / (Al_{ox} + Fe_{ox})] underestimates P saturation in organic soils and a different and more effective approach is needed for these unique soils. The apparent inability of these soils to sorb P contributed to designating them with a 50 mg kg^{-1} M3P threshold.

Typical farming operations for these soils involves applying large amounts of soluble P fertilizer in an attempt to counteract the lack of P retention that is generally seen in the field. Considering that these soils occur in the eastern most part of the state along economically important yet environmentally threatened estuaries, the need to accurately predict P loss from these soils is important. The exact mechanisms of P retention/release are not fully understood and require more in-depth study. In Delaware, Maguire and Sims (2002b) found that a soil with higher organic matter content behaved differently than mineral study soils, and no general trends could be identified. At the present time no studies have examined the composition of organic acids and their interactions with metal oxides and P in these soils of NC.

Measuring Phosphorus Saturation using Mehlich-3

One of the objectives of this study was to determine if the saturation status of soils in North Carolina could be estimated from Mehlich-3-extracted elements as accurately as PSR_{ox}. Since PSR_{ox} has been related to potential P loss (Pote et al., 1996; Hooda, et al., 2000; Paulter and Sims, 2000), it has been suggested that if Mehlich-3 can extract comparable amounts of Fe, Al and P, then states that already

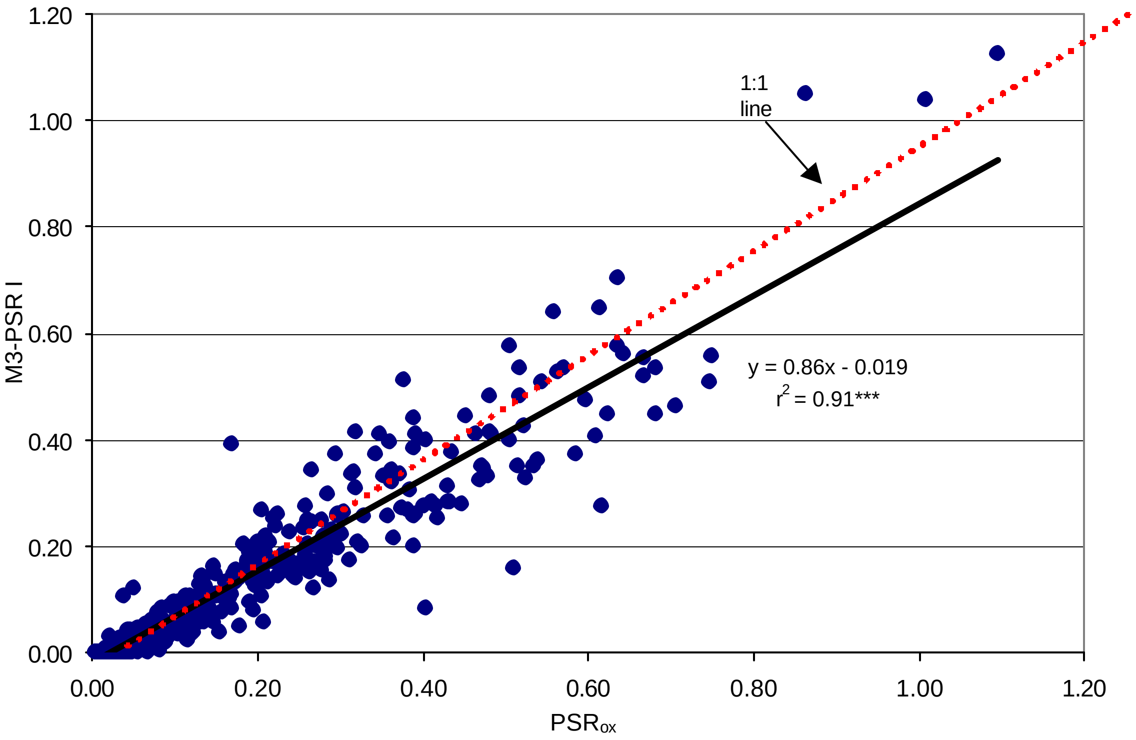

Figure 16. Relationship between the oxalate P sorption ratio, $[P_{ox}/Al_{ox} + Fe_{ox})]$, and the Mehlich-3 P sorption ratio, [M3P/(M3Al + M3Fe)].

determine STP by M3P can easily add M3Al and M3Fe to obtain an estimate of P saturation (Khiari et al., 2000; Maguire and Sims, 2002a; Sims et al., 2002). Mehlich-3-extracted Fe comprised only ~10% of M3(Al + Fe) in our study soils, while Fe_{ox} represented ~40% of $(Al + Fe)_{ox}$. This observation suggests that much of the Fe is in a form not accessible to extraction by Mehlich-3. Despite the disparity between the two extractants in estimating individual elements, especially Fe, the correlation between PSR_{ox} and M3[P/(Al+Fe)] was quite good ($r^2 = 0.95$; Figure 16). Other authors have also found significant relationships between these two measures of P saturation (Khiari et al., 2000; Kleinman and Sharpley, Maguire and Sims, 20002a; Sims et al., 2002). This result in the present study could be due to the combination of higher P_{ox} and Fe_{ox} when compared to M3P and M3Fe, thereby raising both the numerator and denominator values of equation [2]. The molar ratio of Fe_{ox} to M3Fe is 1.5 times higher than the ratio of P_{ox} to M3P, giving PSR_{ox} a slightly greater value than M3-PSR I. Therefore, as M3-PSR I increases, PSR_{ox} increases by approximately the same amount due to an increase in both P_{ox} and Fe_{ox}. Oxalate-extractable Fe always increased slightly more than P_{ox}, thus keeping the overall rate of change (M3-PSR I / PSR_{ox}) the same. The slope (M3-PSR I / PSR_{ox}) of the linear regression line shown in Figure

16 indicates that PSR_{ox} increases approximately 1.16 times more than that of M3-PSR I, indicating that PSR_{ox} is the more sensitive of the two measurements. The slope of the regression line is significantly different (***P***<0.001) than the slope of the 1:1 line, meaning that the M3-PSR II predicts a smaller P saturation when compared to PSR_{ox}. When segregating by soil type, the slope increases in the pattern clay>loam>sand>organic which indicates that, with the exception of organic soils, as the soil texture becomes coarser, PSR_{ox} changes faster than M3-PSR I (data not shown). This is because the disparity between Fe_{ox} content and P_{ox} was greater in clay soils relative to sand soils, thereby lowering PSR_{ox} more in the clay soils.

Iron was a much more important component of a soil's estimated P sorption capacity when using oxalate-extraction rather than Mehlich-3 extraction (Table 7). For example, the amount of Fe_{ox} in clays was almost 12 times greater than the amount of Fe extracted by Mehlich-3. Given this, the assertion that M3-PSR I offers a relatively easy and affordable option for states already using M3P may not be such a good one for North Carolina. An additional difficulty is presented by the changing behavior of Fe in alluvial soils of the Mountains. Lab studies indicate that reduction of Fe-oxides with sorbed P in a Coastal Plain soil causes a seven-fold increase in dissolved P (Hutchison and Hesterberg, in press). By ignoring the contribution of Fe in this situation, a severe underestimation of P buildup and potential dissolution could result. Again, additional study in this area is warranted before fully accepting the Mehlich-3 P saturation measure. However, it is clear that there is a significant relationship between PSR_{ox} and M3-PSR I. Table 9 compares current agronomic M3P categories for the state of NC with the degree of P saturation measured by oxalate and Mehlich-3 for our study soils. Although these results reinforce the finding that M3-PSR underestimates the level of P saturation when compared to PSR_{ox}, they both increase as M3P increases.

Table 9. Mean values of selected parameters for soil samples grouped into each of NC's four agronomic soil test categories.

	Soil test category			
Parameter	**Low** (0–27 mg P kg^{-1})	**Medium** (27-53 mg P kg^{-1})	**High** (53-107 mg P kg^{-1})	**Very high** (>107 mg P kg^{-1})
n	401	56	58	170
PSR_{ox}	0.02	0.10	0.15	0.34
M3-PSR	0.003	0.04	0.09	0.29

CONCLUSIONS

Phosphorus saturation or PSR_{ox}, defined as the quantity of oxalate-extractable P / the sum of oxalate extractable Al and Fe, and related to a soil's capacity to sorb P, was determined for of 685 samples from North Carolina. Soil test (Mehlich-3-extractable) P was also measured. Soil test P alone does not give complete information on the likelihood of a soil to release soluble P because it does not give a quantitative indication of the soil's potential to retain P, as indicated by PSR_{ox}. Soil saturation indices are preferred to more accurately predict a soil's potential to retain and release soluble P. The mean PSR_{ox} of all samples analyzed was 0.12, and the mean value for surface and subsurface samples was 0.29 and 0.06 respectively. Samples in the sand soil group were significantly more P-saturated (PSR_{ox}) than other soil groups, with a mean value of 0.36 in surface samples. Sandy soils clearly represent the greatest threat for movement of soluble P, either as surface runoff or as subsurface drainage when compared to the other soil groups. It was apparent from this study that sample soils in the sand group had exceeded their capacity to strongly retain P at the surface and were exhibiting downward movement of extractable P.

Comparing different animal waste source types indicated that P loading rate was a more important factor determining surface buildup and subsurface changes in extractable P than was the percent solids of the waste. Extractable P in sites receiving poultry litter exceeded that of sites receiving other types of animal waste across all soil types and at all soil depths.

Because the state soil testing lab (NCDACS, Soil Testing Division) currently uses the Mehlich-3 extraction and Mehlich-3 STP is being employed in the state's P indexing tool (PLAT), we were interested in whether Mehlich-3 STP can be used as a proxy for P saturation as represented by PSR_{ox}. Additionally, it was desirable to know if an estimate of soil P saturation based on the Mehlich-3 extraction can sufficiently estimate a soil's degree of P saturation as compared to the oxalate method. Because the state soil testing lab already runs Mehlich-3 extractions, testing for extractable Al and Fe would not substantially increase the workload. Based on the data reported in this study it is clear that Mehlich-3 STP was significantly correlated (r^2=0.76) with PSR_{ox}, especially when soils were grouped according to P threshold groups. Estimates of P saturation by Mehlich-3 and oxalate extractions were also highly correlated (r^2=0.91). However, the substitution M3-PSR for PSR_{ox} should be done with caution due to our finding that Mehlich-3 was a poor extractor of Fe in all soil groups as compared to the oxalate extractant. Using M3-PSR to estimate the level of P saturation in soils with substantial extractable iron contents such as clays of the Piedmont and Mountain regions of NC should be avoided. The dynamic behavior of Fe and its interactions with P is not fully understood, especially in drainage areas experiencing cycles of reduction and oxidation. It is clear that Mehlich-3 did not sufficiently account for iron and its potential importance on P retention.

Organic soils of the state represent a unique situation for which P saturation measurements do not appear to be valid, most notably the assumption that amorphous iron and aluminum are responsible for the majority of P retention. The organic soils studied had significantly greater P sorption capacities, as estimated by the sum of extractable Al and Fe, than other soil groups but did not exceed them in the level of extractable P. Using the PSR_{ox} measurement on these soils would underestimate the risk for P loss, and it is concluded that more research is required to fully understand organic soils of NC.

REFERENCES

Beauchemin, S., R.R. Simard, and D.Cluis. 1996. Phosphorus sorption-desorption kinetics of soil under contrasting land uses. J. Environ. Qual. 25:1317-1325.

Beauchemin, S., and R.R. Simard. 1999. Soil phosphorus saturation degree: Review of some indices and their suitability for P management in Quebec, Canada. Canadian J. of Soil Sci. 79:615-625.

Bhatti, J.S., N.B. Comerford, and C.T. Johnston. 1998. Influence of oxalate and soil organic matter on sorption and desorption of phosphate onto a spodic horizon. Soil Sci. Soc. Am. J. 62: 1089-1095.

Borggaard, O.K., S.S. Jorgensen, J.P. Moberg, and B. Raben-Lange. 1990. Influence of organic matter on phosphate adsorption by aluminum and iron oxides in sandy soils. J. Soil Sci. 41:443-449.

Borggaard, O.K. 1992. Dissolution of poorly crystalline iron oxides in soils by EDTA and oxalate. Z. Pflanzenernahr. Bodenkunden. 155:431-436.

Breeuwsma, A., J.G.A. Reijerink, and O.F. Schoumans. 1995. Impact of manure on accumulation and leaching of phosphate in areas of intensive livestock farming. p.239-249. *In:* K. Steele (ed.) Animal waste and the land-water interface. Lewis Publ.-CRC Press, New York.

Buol, S.W., F.D. Hole, R.J. McCracken, and R.J. Southard. 1997. Soil Genesis and Classification. 4th ed. Iowa State Univ. Press. Ames, Iowa.

Cox, F.R. 1994. Predicting increases in extractable phosphorus from fertilizing soils of varying clay content. Soil Sci. Soc. Am. J. 58:1249-1253.

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

D'Angelo, E.M., M.V. Vandiviere, W.O. Thom, and F. Sikora. 2003. Estimating soil phosphorus requirements and limits from oxalate extract data. J. Environ. Qual. 32:1082-1088.

Darke, A.K., and M.R. Walbridge. 2000. Al and Fe biogeochemistry in a floodplain forest: Implications for P retention. Biogeochemistry 51: 1-32.

Daughtrey, Z.W., J.W. Gilliam, and E.J. Kamprath. 1972. Phosphorus supply characteristics of acid organic soils as measured by desorption and mineralization. Soil Sci. 115:18-24.

Dolman, J.D., and S.W. Buol. 1968. Organic soils on the lower coastal plain of North Carolina. Soil Sci. Soc. Am. Proc. 32:414-418.

Fox, R.L. and E.J. Kamprath. 1970. Phosphate sorption isotherms for evaluating phosphate requirements of soils. Soil Sci. Soc. Am. Proc. 34:902-907.

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

Freese, D., S.E.A.T.M. van der Zee, and W.H. van Riemsdijk. 1992. Comparison of different models for phosphate sorption as a function of the iron and aluminum oxides of soil. J. Soil Sci. 43:729-738.

Gerke, J. and R. Hermann. 1992. Adsorption of orthophosphate to humic-Fe-complex and to amorphous Fe-oxide. Z. Pflanzenernahr. Bodenk. 155: 233-236.

Guo, F., and R.S. Yost. 1999. Quantifying the available soil phosphorus pool with the acid ammonium oxalate method. Soil Sci. Soc. Am. J. 63:651-656.

Havlin, J.L., S. Hodges, D. Osmond, A. Johnson, D. Crouse, W. Skaggs, R. Evans, J. Parsons, P. Westerman, L. Price, R. Reich. 2001. Assessing the risk of phosphorus delivery to North Carolina waters. Proceedings International Symposium: *Addressing Animal Production and Environmental Issues.* Oct. 3-5 2001, Raleigh, NC.

He, J.Z., A. De Cristofaro, and A. Violante. 1999. Comparison of adsorption of phosphate, tartrate, and oxalate on hydroxy aluminum montmorillonite complexes. Clays Clay Mineral. 47: 226-233.

Hooda, P.S., V.W. Truesdale, A.C. Edwards, P.J.A. Withers, M.N. Aitken, A. Miller, and A.R. Rendell. 2001. Manuring and fertilization effects on phosphorus accumulation in soils and potential environmental implications. Advances in Environ. Research. 5:13-21.

Hooda, P.S., A.R. Rendell, A.C. Edwards, P.J.A. Withers, M.N. Aitken, and V.W. Truesdale. 2000. Relating soil phosphorus indices to potential phosphorus release to water. J. Environ. Qual. 29:1161- 1171.

Hsu, P.H. 1989. Aluminum hydroxides and oxyhydroxides. In: Dixon, J.B. and S.B. Weed (eds.) Minerals in Soil Environments (2nd ed.). SSSA. No. 1. Madison, WI.

Hutchinson, K.J., and D. Hesterberg. Dissolution of phosphate in a phosphorus-enriched Ultisol as affected by microbial reduction. (in review).

Khiari, L., L.E. Parent, A. Pellerin, A.R.A. Alimi, C. Tremblay, R.R. Simard, and J. Fortin. 2000. An agri-environmental phosphorus saturation index for acid coarse-textured soils. J. Environ. Qual. 29:1561-1567.

Kilmer, V.J., and L.T. Alexander. 1949. Methods of making mechanical analyses of soils. Soil Sci. 68:15-24.

Kleinman, P.J.A., and R.B. Bryant, W.S. Reid, A.N. Sharpley, and D. Pimentel. 2000. Using soil phosphorus behavior to identify environmental thresholds. Soil Sci. 165:943-950.

Kleinman, P.J.A., and A.N. Sharpley. 2002. Estimating soil phosphorus sorption saturation from Mehlich-3 data. Commun. Soil Sci. Plant Anal. 33: 1825-1839.

Kleinman, P.J.A., B.A. Needelman, A.N. Sharpley, and R.W. McDowell. 2003. Using soil phosphorus profile data to assess phosphorus leaching potential in manured soils. Soil Sci. Soc. Am. J. 67:215-224.

Kwong, K.F. Ng Kee, and P.M. Huang. 1979. Surface reactivity of aluminum hydroxides precipitated in the presence of low molecular weight organic acids. Soil Sci. Soc. Am. J. 43: 1107-1113.

Kwong, K.F. Ng Kee, and P.M. Huang. 1977. Influence of citric acid on the hydrolytic reactions of aluminum. Soil Sci. Soc. Am. J. 41: 692-697.

Lan, M., N.B. Comerford, and T.R. Fox. 1995. Organic anions' effect on phosphorus release from Spodic horizons. Soil Sci. Soc. Am. J. 59:1745-1749.

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

Liu, F., A. De Cristofaro, and A. Violante. 2001. Effect of pH, phosphate and oxalate on the adsorption/desorption of arsenate on/from goethite. Soil Sci. 166: 197-208.

Lookman, R.N., Vandeweert, R. Merckx, and K.Vlassak. 1995. Geostatistical assessment of the regional distribution of phosphate sorption capacity parameters (Fe_{ox} and Al_{ox}) in northern Belgium. Geoderma. 66:285-296.

Lookman, R.N., K.Jansen, R. Merckz, and K. Vlassak. 1996. Relationship between soil properties and phosphate sorption parameters: A transect study in northern Belgium. Geoderma. 69:265-274.

Maguire, R.O., J.T. Sims, and F.J. Coale. 2000a. Phosphorus solubility in biosolids-amended farm soils in the Mid-Atlantic region of the USA. J. Environ. Qual. 29:1225-1233.

Maguire, R.O., J.T. Sims, and F.J. Coale. 2000b. Phosphorus fractionation in biosolids-amended soils: Relationship to soluble and desorbable phosphorus. Soil Sci. Soc. Am. J. 64:2018-2024.

Maguire, R.O., and J.T. Sims. 2002a. Measuring agronomic and environmental soil phosphorus saturation and predicting phosphorus leaching with Mehlich-3. Soil Sci. Soc. Am. J. 66: 2033-2039.

Maguire, R.O., and J.T. Sims. 2002b. Soil testing to predict phosphorus leaching. J. Environ. Qual. 31:1601-1609.

Martell, A.E. and R.M. Smith. Critical Stability Constants. Volume 3: Other Organic Ligands. Plenum Press, New York. 1977.

McDowell,R.W., and L. Condron. 2001. Influence of soil constituents on soil phosphorus sorption and desorption. Commun. Soil Sci. Plant Anal. 32:2531-2547.

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.

McDowell, R.W., A. Sharpley, P. Brookes, and P. Poulton. 2000. Relationship between soil test phosphorus and phosphorus release to solution. Soil Sci. 166:137-149.

McKeague, J.A., and J.H. Day. 1966. Dithionite and oxalate-extractable Fe and Al as aids in differentiating various classes of soils. Can. J. Soil Sci. 46:13-22.

Mehlich, A. 1984. Mehlich-3 soil test extractant: A modification of Mehlich-2 extractant. Commun. Soil Sci. Plant Anal. 15:1409-1416.

Nair, V.D., and D.A. Graetz. 2002. Phosphorus saturation in Spodosols impacted by manure. J. Environ. Qual. 31:1279-1285.

North Carolina Dept. of Agric. and Consumer Services. 2002 Agronomic Division Annual Soil Test Summary.

Novozamsky, I., R. van Eck, V.J.G. Houba and J.J. van der Lee. 1986. Use of inductively coupled plasma atomic emission spectrometry for determination of iron, aluminum and phosphorus in Tamm's soil extracts. Netherlands J. Agric. Sci. 34:185-191.

Olsen, S.R., and L.E. Sommers. 1982. Phosphorus. In: Page, A.L., R.H. Miller, and D.R. Keeney (eds.) Methods of Soil Analysis. Part 2. Chemical and Microbiological Properties. Agron. Monograph no. 9. ASA-SSSA. Madison, WI.

Parfitt, R.L. 1989. Optimum conditions for extraction of Al, Fe, and Si from soils with acid oxalate. Commun. Soil Sci. Plant Anal. 20:801-816.

Paulter, M.C., and J.T. Sims. 2000. Relationship between soil test phosphorus, soluble phosphorus, and phosphorus saturation in Delaware soils. Soil. Sci. Soc. Am. J. 64:765-773.

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.

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 concentration in runoff. J. Environ. Qual. 28:170-175.

SAS Institute. 1998. SAS user's guide: Statistics. Version 8.0 ed. SAS Inst., Cary, NC.

Saunders, W.M.H. 1964. Phosphate retention by New Zealand soils and its relationship to free sesquioxides, organic matter and other soil properties. N. Z. J. Agric. Res. 8:30-57.

Schoumans, O.F. 2000. Determination of the degree of phosphate saturation in non-calcareous soils. p. 31-34. *In:* G.M. Pierzynski (ed.) Methods of phosphorus analysis for soils, sediments, residuals, and waters. So. Coop. Series Bull. No. 396. Kansas St. Univ., Manhattan, KS.

Schwertmann, U., H. Kodama, and W.R. Fischer. 1986. Mutual interactions between organics and iron oxides. *In:* Huang, P.M. and M. Schnitzer (eds.) Interactions of Soil Minerals with Natural Organics and Microbes. pp. 223-250. Soil Sci. Soc. Am., Madison, WI.

SERA-IEG-6 (Southern Extension Research Activity-Information Exchange Group-6). 2001. Procedures used by state soil testing laboratories in the southern region of the United States. Bull. 190-C, SW Florida Res. Educ. Center, Immokalee, FL.

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

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

Sharpley, A.N., J.L. Weld, D.B. Beegle, P.J.A. Kleinman, W.J. Gburek, P.A. Moore, Jr., and G. Mullins. 2003. Development of phosphorus indices for nutrient management planning strategies in the United States. J. Soil Water Conserv. 58:137-152.

Shaw, J.N. 2001. Iron and aluminum oxide characterization for weathered Alabama Ultisols. Comm. Soil Sci. Plant Analy. 32: 49-64.

Sims, J.T., A.C. Edwards, O.F. Schoumans, and R.R. Simard. 2000. Integrating soil phosphorus testing into environmentally based agricultural management practices. J. Environ. Qual. 29:60-71.

Sims, J.T., and F.J. Coale. 2002. Solutions to nutrient management problems in the Chesapeake Bay Watershed, USA. p. 345-371. *In:* Haygarth, P.M. and S.C. Jarvis (eds.) Agriculture, Hydrology and Water Quality. CAB International, Oxfordshire, UK.

Sims, J.T., R.O. Maguire, A.B. Leytem, K.L. Gartley, and M.C. Paulter. 2002. Evaluation of Mehlich 3 as an agri-environmental soil phosphorus test for the Mid-Atlantic United States of America. Soil Sci. Soc. Am. J. 66:2016-2032.

Stumm, W. 1986. Coordinative interactions between soil solids and water – an aquatic chemist's point of view. Geoderma 38: 19-30.

Tarkalson, D. 2001. Phosphorus loss in surface runoff from Piedmont soils receiving animal manure and fertilizer additions. Ph.D. Thesis, N.C. State University.

U.S. Environmental Protection Agency (USEPA) 1996. Environmental indicators of water quality in the United States. EPA 841-R-96-002. U.S. EPA. Office of Water Regulations and Standards (4503F), U.S. Government Printing Office, Washington, D.C.

van der Zee, S.E.A.T.M, and W.H. van Riemsdijk. 1988. Model for long-term phosphate reaction kinetics in soil. J. Environ. Qual. 17:35-41.

Violante, A., C. Colombo, and A. Buondonno. 1991. Competitive adsorption of phosphate and oxalate by aluminum oxides. Soil Sci. Soc. Am. J. 55: 65-70.

Young, W.O. and D.S. Ross. 2001. Phosphate release from seasonally flooded soils: A laboratory microcosm study. J. Environ. Qual. 30:91-101.

Yuan, G., and L.M. Lavkulich. 1994. Phosphate sorption in relation to extractable iron and aluminum in spodosols. Soil Sci. Soc. Am. J. 58: 343-346.

Appendix

Table A1. Characteristics of all 685 samples.

County	Depth (in)	Threshold/ Series	M3P (mg/kg)	P_{ox} (mg/kg)	M3Al (mg/kg)	Al_{ox} (mg/kg)	M3Fe (mg/kg)	Fe_{ox} (mg/kg)	Texture
Duplin	0-4	100/ Leon	162.5	212	505.5	560	148.9	283	S
Duplin	4-8		112.3	136	594.6	588	142.8	219	S
Duplin	8-12		22.2	44	994.4	952	105.1	120	S
Duplin	12-16		5.4	45	1116.6	1688	85.1	251	S
Duplin	16-20		0.0	49	1073.5	1616	64.8	254	S
Duplin	20-24		1.3	24	1091.7	1460	47.4	219	S
Duplin	24-28		1.3	23	1044.7	1348	46.0	187	S
Duplin	28-32		2.0	25	964.7	1372	30.1	124	S
Duplin	0-4	200/ Craven	422.5	628	747.2	724	249.4	676	LS
Duplin	4-8		390.7	576	709.2	744	254.9	700	LS
Duplin	8-12		214.8	271	728.3	632	173.1	386	SL
Duplin	12-16		56.0	74	828.5	728	67.4	400	SL
Duplin	16-20		0.0	15	1026.9	1088	41.5	480	L
Duplin	20-24		0.0	13	1076.9	1084	38.7	384	SCL
Duplin	24-28		0.0	11	1079.7	1040	35.4	394	SL
Duplin	28-32		0.0	17	1104.1	1176	39.8	664	SCL
Duplin	0-4	200/ Gritney	155.9	351	729.7	764	182.5	764	L
Duplin	4-8		60.5	164	900.0	908	99.4	384	SCL
Duplin	8-12		1.0	58	1195.0	1344	40.8	368	L
Duplin	12-16		0.0	25	1322.8	1656	30.1	336	L
Duplin	16-20		0.0	18	1405.2	1740	28.0	432	CL
Duplin	20-24		0.0	9	1382.8	1680	28.2	452	CL
Duplin	24-28		0.0	12	1398.2	1628	29.2	440	SCL
Duplin	28-32		0.0	11	1300.0	1384	27.3	472	SCL
Duplin	0-4	200/ Norfolk	240.5	333	430.9	1176	212.4	1100	LS
Duplin	4-8		149.2	259	523.6	552	192.9	536	LS
Duplin	8-12		68.8	116	731.0	608	115.5	480	SL
Duplin	12-16		5.9	22	830.5	844	48.9	568	SL
Duplin	16-20		0.0	17	878.0	888	42.9	504	SL
Duplin	20-24		0.0	8	883.1	804	44.1	375	SL
Duplin	24-28		0.0	7	894.6	836	40.3	345	SL
Duplin	28-32		0.0	7	806.5	708	37.1	299	SL
Duplin	0-4	200/ Norfolk	27.5	60	665.6	560	252.4	540	S
Duplin	4-8		7.1	24	632.5	656	199.9	588	S
Duplin	8-12		0.0	7	623.9	456	93.9	342	S
Duplin	12-16		0.0	7	720.0	452	51.9	292	S
Duplin	16-20		0.0	6	770.5	516	35.8	252	LS
Duplin	20-24		0.0	8	838.5	604	32.7	281	LS
Duplin	24-28		0.0	7	793.6	484	32.8	259	S
Duplin	28-32		0.0	11	703.4	344	41.6	262	S
Duplin	0-4	200/ Norfolk	603.5	986	809.4	944	269.3	812	LS
Duplin	4-8		622.4	852	864.7	1004	310.6	896	LS
Duplin	8-12		384.5	534	847.4	1180	272.1	1184	SL
Duplin	12-16		53.5	116	1093.0	1664	107.8	1556	LS

Duplin	16-20		3.6	17	664.2	456	57.7	292	SL
Duplin	20-24		3.8	30	649.7	448	50.4	264	SL
Duplin	24-28		0.0	13	640.7	436	43.3	237	SL
Duplin	28-32		0.5	14	613.2	399	44.2	279	SL
Duplin	0-4	100/	100.9	149	254.2	118	137.2	194	S
Duplin	4-8	Pactolus	90.2	169	295.7	238	145.2	347	S
Duplin	8-12		173.8	232	595.2	764	88.2	306	S
Duplin	12-16		162.5	220	657.5	568	88.7	214	S
Duplin	16-20		121.8	151	537.7	512	80.4	296	S
Duplin	20-24		61.1	99	456.0	380	108.3	404	S
Duplin	24-28		43.0	81	526.6	608	229.1	780	S
Duplin	28-32		41.4	70	398.1	364	173.1	444	S
Duplin	0-4	200/	377.3	578	537.7	488	224.5	384	LS
Duplin	4-8	Marvyn	388.7	502	535.7	548	207.2	452	S
Duplin	8-12		272.0	311	525.3	404	144.7	274	S
Duplin	12-16		33.2	44	991.9	1208	94.0	1160	SL
Duplin	16-20		153.5	217	780.9	596	73.3	254	SL
Duplin	20-24		25.9	51	1005.5	980	44.8	380	SCL
Duplin	24-28		67.2	105	1031.1	912	70.8	448	SCL
Duplin	28-32		121.7	52	982.0	1040	40.1	386	SCL
Duplin	0-4	100/	365.0	736	864.6	1240	191.2	1208	S
Duplin	4-8	Pactolus	303.7	536	881.2	1280	166.8	1100	S
Duplin	8-12		193.0	365	1123.4	1820	104.8	1184	S
Duplin	12-16		80.2	154	881.1	1236	109.2	952	S
Duplin	16-20		94.8	82	632.0	1192	131.8	428	S
Duplin	20-24		57.6	66	555.7	258	133.5	180	S
Duplin	24-28		55.6	73	464.8	412	117.9	492	S
Duplin	28-32		33.0	85	463.1	496	119.3	1120	S
Duplin	0-4	100/	423.5	816	852.8	1388	164.1	1204	S
Duplin	4-8	Pactolus	449.3	740	871.6	1484	158.3	1124	LS
Duplin	8-12		290.1	582	891.7	1776	115.8	1428	S
Duplin	12-16		152.9	264	772.6	1084	115.1	1024	S
Duplin	16-20		135.1	222	743.2	840	118.9	956	S
Duplin	20-24		104.2	194	545.9	672	130.0	984	S
Duplin	24-28		44.0	75	367.3	328	148.3	748	S
Duplin	28-32		27.8	61	292.7	237	137.3	548	S
Duplin	0-4	200/	4.7	57	885.5	1244	121.4	1496	SL
Duplin	4-8	Johns	266.4	666	891.7	1384	296.2	2188	SL
Duplin	8-12		65.2	268	879.8	1520	248.8	2424	SL
Duplin	12-16		281.2	303	541.8	448	101.9	207	S
Duplin	16-20		4.2	45	952.4	1256	91.3	1236	SL
Duplin	20-24		3.9	49	859.1	976	116.7	1300	SL
Duplin	24-28		9.4	51	686.2	812	176.8	1640	S
Duplin	28-32		12.7	32	586.0	500	138.4	772	S
Duplin	0-4	200/	181.5	409	817.2	1212	195.3	1428	SL
Duplin	4-8	Johns	138.1	343	816.4	1136	184.4	1200	LS
Duplin	8-12		56.2	224	865.2	1356	120.7	1468	SL
Duplin	12-16		57.5	97	896.1	560	115.2	540	SL
Duplin	16-20		5.4	90	1185.8	1924	68.0	1460	SL
Duplin	20-24		7.8	78	1208.9	1724	70.2	1476	SL

Duplin	24-28		13.8	85	1168.0	1716	74.0	1284	SL
Duplin	28-32		25.5	104	1112.5	1680	74.9	836	SL
Duplin	0-4	200/	199.9	257	648.9	672	117.9	392	LS
Duplin	4-8	Norfolk	76.9	129	578.0	592	107.8	376	SL
Duplin	8-12		1.5	12	625.1	432	66.2	200	SL
Duplin	12-16		0.0	8	839.8	768	41.6	253	SCL
Duplin	16-20		0.0	9	1005.6	1052	26.7	284	SCL
Duplin	20-24		0.0	9	944.8	980	21.6	264	SCL
Duplin	24-28		0.0	8	834.6	832	27.7	246	SCL
Duplin	28-32		0.0	8	803.7	704	22.4	195	SCL
Duplin	0-4	200/	109.9	208	607.4	544	144.5	412	SL
Duplin	4-8	Norfolk	73.2	189	626.4	640	130.2	472	SL
Duplin	8-12		29.4	98	810.0	736	98.8	492	SL
Duplin	12-16		6.8	52	850.0	808	79.6	504	SL
Duplin	16-20		0.0	16	931.5	1000	39.7	468	SCL
Duplin	20-24		0.0	11	872.2	896	24.3	320	SCL
Duplin	24-28		0.0	8	944.2	760	76.9	340	SCL
Duplin	28-32		0.0	5	931.2	696	30.0	206	SL
Duplin	0-4	200/	32.5	57	436.8	520	115.2	350	LS
Duplin	4-8	Torhunta	19.3	25	349.7	355	100.3	152	S
Duplin	8-12		16.7	33	815.3	920	179.4	336	LS
Duplin	12-16		0.0	30	1124.6	1448	134.8	532	LS
Duplin	16-20		0.0	31	1204.4	1812	141.7	1072	LS
Duplin	20-24		3.2	26	972.4	1200	127.8	576	LS
Duplin	24-28		3.1	25	981.1	1128	107.6	476	S
Duplin	28-32		4.9	28	1095.0	1232	83.9	366	LS
Duplin	0-4	200/	139.5	189	607.5	496	123.4	290	S
Duplin	4-8	Goldsboro	122.4	188	554.2	528	134.1	328	S
Duplin	8-12		29.0	61	724.7	688	103.8	299	LS
Duplin	12-16		5.9	22	621.8	512	71.6	317	S
Duplin	16-20		1.8	18	582.8	394	70.7	344	LS
Duplin	20-24		0.0	19	642.9	420	68.4	408	LS
Duplin	24-28		0.0	16	704.7	440	63.7	456	LS
Duplin	28-32		0.0	31	877.9	684	74.9	812	LS
Duplin	0-4	200/	186.8	278	363.5	384	144.4	362	S
Duplin	4-8	Lumbee	105.6	159	352.9	366	139.5	268	S
Duplin	8-12		93.5	124	479.4	532	112.9	185	S
Duplin	12-16		51.2	75	998.6	1100	110.9	223	S
Duplin	16-20		11.5	29	941.0	1040	76.2	176	S
Duplin	20-24		149.8	204	1009.9	1176	115.0	260	S
Duplin	24-28		184.1	223	1120.0	1332	59.6	215	S
Duplin	28-32		105.0	209	1260.0	2204	41.6	240	LS
Duplin	0-4	100/	436.0	432	610.8	370	144.7	274	S
Duplin	4-8	Autryville	299.7	456	480.7	500	142.5	343	S
Duplin	8-12		326.4	392	532.4	520	120.2	295	S
Duplin	12-16		173.0	255	493.9	428	87.2	234	S
Duplin	16-20		149.5	176	472.7	355	77.4	232	S
Duplin	20-24		124.4	132	478.3	284	85.7	234	S
Duplin	24-28		148.7	193	617.6	416	90.5	376	S
Duplin	28-32		160.4	203	775.9	464	77.0	209	SL

Duplin	0-4	100/	382.1	542	605.9	704	177.9	580	S
Duplin	4-8	Pactolus	333.4	624	739.6	1056	223.4	924	S
Duplin	8-12		249.8	323	766.4	900	191.6	804	S
Duplin	12-16		104.2	165	762.5	828	230.0	892	S
Duplin	16-20		58.0	100	684.9	648	108.6	420	S
Duplin	20-24		29.8	64	609.0	476	100.2	496	S
Duplin	24-28		28.1	71	583.3	596	119.1	892	S
Duplin	28-32		23.9	61	531.0	628	127.3	892	S
Duplin	0-4	100/	592.1	816	807.6	1004	179.2	840	S
Duplin	4-8	Pactolus	537.7	574	829.2	976	180.6	720	S
Duplin	8-12		409.1	456	896.4	1056	119.7	616	S
Duplin	12-16		218.2	328	904.2	1140	98.5	812	S
Duplin	16-20		165.1	232	763.9	664	95.9	472	S
Duplin	20-24		97.4	140	604.6	408	95.6	452	S
Duplin	24-28		66.1	117	649.2	428	113.7	612	S
Duplin	28-32		29.5	68	514.1	361	102.1	740	S
Duplin	0-4	100/	140.9	331	845.7	1304	189.0	1000	S
Duplin	4-8	Pactolus	123.5	283	956.0	1284	210.3	976	S
Duplin	8-12		135.6	310	878.5	1204	197.3	964	LS
Duplin	12-16		53.1	125	886.1	1032	136.1	768	LS
Duplin	16-20		8.7	46	964.3	1248	89.7	680	LS
Duplin	20-24		10.5	40	914.7	912	80.4	480	LS
Duplin	24-28		10.8	35	686.2	608	62.7	383	S
Duplin	28-32		15.6	37	684.5	656	62.0	283	S
Duplin	0-4	100/	28.4	58	309.6	264	102.4	242	S
Duplin	4-8	Leon	13.6	17	195.3	185	70.7	111	S
Duplin	8-12		29.5	32	584.5	536	131.8	198	S
Duplin	12-16		5.3	58	1341.9	2176	80.0	354	S
Duplin	16-20		0.6	62	1559.1	3060	81.7	824	S
Duplin	20-24		13.9	57	1291.9	2032	87.6	552	S
Duplin	24-28		36.4	74	1146.8	1676	90.4	548	S
Duplin	28-32		50.9	92	1215.9	1612	76.8	381	S
Duplin	0-4	100/	312.7	512	941.5	1088	239.6	1848	S
Duplin	4-8	Blanton	249.8	448	935.8	1312	254.8	1040	LS
Duplin	8-12		13.2	49	1007.5	1356	149.3	964	LS
Duplin	12-16		0.0	10	815.7	620	83.7	420	LS
Duplin	16-20		0.0	8	809.3	564	60.5	287	LS
Duplin	20-24		0.0	7	850.4	564	36.3	239	LS
Duplin	24-28		0.0	5	889.9	492	35.0	177	LS
Duplin	28-32		0.0	6	804.7	452	42.2	190	SL
Duplin	0-4	200/	358.9	488	853.2	1144	149.3	424	LS
Duplin	4-8	Woodington	239.8	287	954.2	1176	171.9	387	SL
Duplin	8-12		50.6	174	937.9	884	126.1	856	LS
Duplin	12-16		7.4	25	952.5	1080	68.3	243	SL
Duplin	16-20		0.0	12	769.7	696	71.5	327	SL
Duplin	20-24		0.0	10	820.7	676	63.7	201	SL
Duplin	24-28		0.0	8	794.7	636	64.2	255	SL
Duplin	28-32		0.0	6	744.3	580	59.0	236	SL
Haywood	0-4	500/	372.6	668	865.1	980	210.4	1220	L
Haywood	4-8	Evard	106.5	235	787.1	836	121.0	792	L

Haywood	8-12		0.4	36	920.7	812	48.8	564	L
Haywood	12-16		0.0	23	996.4	876	39.0	664	CL
Haywood	16-20		0.0	20	920.8	1000	33.8	800	CL
Haywood	20-24		0.0	29	955.0	1036	35.5	928	CL
Haywood	24-28		0.0	23	1060.4	1180	40.8	976	C
Haywood	28-32		0.0	26	1001.8	1160	38.7	1004	CL
Haywood	0-4	500/	176.9	456	877.6	1044	238.2	804	L
Haywood	4-8	Hayesville	35.0	112	940.1	1032	98.2	764	L
Haywood	8-12		8.3	81	1028.9	1108	68.2	784	L
Haywood	12-16		0.0	85	1063.8	1172	68.2	936	L
Haywood	16-20		7.5	79	1007.5	1008	66.3	740	L
Haywood	20-24		8.7	66	1025.8	900	70.5	600	SL
Haywood	24-28		0.6	38	885.5	736	57.2	524	SL
Haywood	28-32		0.3	33	931.8	688	60.3	524	SL
Haywood	0-4	500/	227.4	932	999.1	2176	156.2	2744	CL
Haywood	4-8	Dillsboro	172.1	834	983.8	2156	147.9	2884	CL
Haywood	8-12		47.2	438	1051.4	2084	122.0	2936	CL
Haywood	12-16		0.0	183	1078.0	2316	80.7	2808	C
Haywood	16-20		0.0	181	1115.6	2084	81.8	2536	C
Haywood	20-24		0.0	128	1094.3	1932	73.4	2596	C
Haywood	24-28		0.0	125	1158.3	1884	76.1	2548	C
Haywood	28-32		0.0	101	1080.4	1772	63.5	2116	C
Haywood	0-4	200/	171.3	496	829.1	1216	185.2	2172	SL
Haywood	4-8	Rosman	181.0	446	841.1	1104	215.1	1992	SL
Haywood	8-12		155.5	498	830.5	1256	207.8	2340	SL
Haywood	12-16		107.9	397	772.6	1220	166.5	2268	SL
Haywood	16-20		32.3	252	851.6	1332	143.6	2376	SL
Haywood	20-24		9.2	254	971.2	1772	126.9	2964	SL
Haywood	24-28		8.9	260	954.4	1844	127.6	3240	SL
Haywood	28-32		10.0	237	923.3	1548	127.0	2704	LS
Haywood	0-4	500/	78.1	330	1104.6	1376	182.3	1572	SCL
Haywood	4-8	Dillsboro	48.4	300	1064.5	1372	157.8	1564	SCL
Haywood	8-12		24.6	180	1040.2	1192	123.0	1228	SCL
Haywood	12-16		0.0	37	1037.9	816	69.7	548	SCL
Haywood	16-20		0.0	23	980.2	776	52.7	496	SCL
Haywood	20-24		0.0	19	980.4	752	54.8	500	SL
Haywood	24-28		0.0	19	985.3	692	57.1	524	SCL
Haywood	28-32		0.0	17	897.9	716	52.9	544	SCL
Iredell	0-4	500/	12.7	106	767.6	764	68.7	808	SCL
Iredell	4-8	Cecil	1.2	64	845.4	832	62.9	816	SCL
Iredell	8-12		0.0	26	957.7	872	59.7	824	CL
Iredell	12-16		0.0	18	1021.0	1012	38.5	888	C
Iredell	16-20		0.0	17	1062.5	1052	38.7	1008	C
Iredell	20-24		0.0	15	1108.8	1072	40.6	972	C
Iredell	24-28		0.0	19	1095.2	1060	39.3	1028	CL
Iredell	28-32		0.0	16	1142.0	1108	40.9	964	CL
Iredell	0-4	500/	126.8	238	795.2	696	92.2	584	SL
Iredell	4-8	Cecil	14.4	72	791.3	704	57.8	472	SL
Iredell	8-12		0.0	21	944.6	764	44.5	532	SCL
Iredell	12-16		0.0	19	1001.7	1044	41.4	596	SCL

Iredell	16-20		0.0	16	1092.9	1112	41.8	660	C
Iredell	20-24		0.0	13	1128.0	1152	43.3	648	L
Iredell	24-28		0.0	12	1189.8	1040	37.3	568	CL
Iredell	28-32		0.0	13	1089.3	964	30.9	520	CL
Iredell	0-4	500/	159.9	470	922.5	1108	150.3	1144	SL
Iredell	4-8	Cecil	68.7	200	882.8	944	113.9	1144	SCL
Iredell	8-12		6.3	60	946.4	984	78.0	1052	SC
Iredell	12-16		0.0	21	1108.3	956	47.0	940	C
Iredell	16-20		0.0	28	1149.5	1060	45.9	1148	C
Iredell	20-24		0.0	20	1139.2	848	41.1	720	C
Iredell	24-28		0.0	15	1124.7	660	42.3	576	CL
Iredell	28-32		0.0	13	1069.0	572	33.7	476	SCL
Iredell	0-4	500/	24.1	135	947.0	1228	135.8	1256	SCL
Iredell	4-8	Cecil	0.0	29	949.5	800	104.0	1136	C
Iredell	8-12		0.0	14	1021.3	584	77.1	680	L
Iredell	12-16		0.0	17	1013.5	648	63.5	720	SCL
Iredell	16-20		0.0	11	1085.8	644	50.7	504	SCL
Iredell	20-24		0.0	11	1058.9	620	55.9	476	SL
Iredell	24-28		0.0	15	1107.6	616	72.2	508	SCL
Iredell	28-32		0.0	14	1097.2	632	75.9	480	SC
Iredell	0-4	500/	44.8	162	936.4	892	92.3	948	C
Iredell	4-8	Cecil	2.7	49	944.8	760	63.4	780	C
Iredell	8-12		0.0	25	1002.0	812	49.2	856	CL
Iredell	12-16		0.0	23	1044.9	896	49.5	1048	C
Iredell	16-20		0.0	18	1157.3	944	50.2	1180	C
Iredell	20-24		0.0	15	1151.6	840	46.3	984	L
Iredell	24-28		0.0	21	1125.0	788	52.3	968	SCL
Iredell	28-32		0.0	21	1158.7	804	53.5	900	SCL
Iredell	0-4	500/	31.9	125	760.8	608	61.2	624	SL
Iredell	4-8	Lloyd	5.3	58	818.5	596	59.1	620	SC
Iredell	8-12		0.0	32	920.5	892	48.2	868	C
Iredell	12-16		0.0	21	972.0	812	41.7	808	C
Iredell	16-20		0.0	22	975.2	840	36.1	848	C
Iredell	20-24		0.0	18	1091.1	832	44.7	816	C
Iredell	24-28		0.0	16	1012.6	704	40.9	716	CL
Iredell	28-32		0.0	16	1003.8	696	38.4	708	C
Nash	0-4	500/	224.5	336	682.3	656	212.8	832	SiL
Nash	4-8	Nason	49.4	96	645.7	532	123.4	484	SiL
Nash	8-12		9.5	42	803.9	552	61.5	230	SiL
Nash	12-16		1.0	27	942.4	592	42.1	164	SiL
Nash	16-20		0.0	17	926.3	728	32.5	242	SiL
Nash	20-24		0.0	8	996.4	804	31.4	273	CL
Nash	24-28		0.0	6	1031.2	848	28.3	232	CL
Nash	28-32		0.0	6	1164.4	916	31.3	260	CL
Nash	0-4	200/	100.8	203	530.7	532	110.3	390	LS
Nash	4-8	Norfolk	65.7	126	556.8	516	86.8	331	SL
Nash	8-12		2.4	29	731.7	580	37.9	230	SL
Nash	12-16		0.0	14	922.0	696	41.9	273	CL
Nash	16-20		0.0	13	1003.7	928	36.0	404	C
Nash	20-24		0.0	10	1067.0	1004	38.8	416	C

Nash	24-28		0.0	8	1055.7	1064	28.9	408	C
Nash	28-32		0.0	8	1063.8	1084	29.3	404	C
Nash	0-4	200/	283.2	436	325.7	357	114.3	540	SL
Nash	4-8	Norfolk	145.6	192	438.9	377	116.8	392	SL
Nash	8-12		60.0	103	580.1	398	76.2	308	SL
Nash	12-16		18.9	45	651.1	348	59.4	186	SL
Nash	16-20		2.0	23	769.1	408	40.1	165	L
Nash	20-24		0.0	24	808.1	464	32.4	177	SCL
Nash	24-28		0.0	7	900.8	532	30.4	189	SCL
Nash	28-32		0.0	6	922.6	592	24.2	170	CL
Nash	0-4	200/	153.1	236	425.0	355	96.3	253	SL
Nash	4-8	Norfolk	109.0	159	439.4	375	86.2	288	SL
Nash	8-12		45.3	82	566.5	380	68.7	250	SL
Nash	12-16		15.7	43	691.8	365	67.5	170	SL
Nash	16-20		0.0	21	714.1	436	64.9	243	SL
Nash	20-24		0.0	10	757.7	500	56.3	210	SCL
Nash	24-28		0.0	9	787.0	568	38.2	334	SCL
Nash	28-32		0.0	7	825.6	516	32.0	212	SCL
Nash	0-4	100/	307.4	494	493.3	452	221.1	492	S
Nash	4-8	Blanton	365.3	626	576.2	532	279.3	556	S
Nash	8-12		438.7	562	592.6	488	250.8	476	S
Nash	12-16		316.9	376	704.2	532	189.7	333	LS
Nash	16-20		262.5	307	658.3	480	147.8	295	LS
Nash	20-24		282.1	319	865.9	560	166.6	307	SL
Nash	24-28		108.5	166	843.4	572	95.7	275	SL
Nash	28-32		35.9	75	772.6	488	60.8	217	SL
Nash	0-4	200/	193.5	356	424.4	382	188.9	432	S
Nash	4-8	Norfolk	305.4	472	600.3	496	284.7	552	S
Nash	8-12		321.4	408	632.6	420	251.7	386	S
Nash	12-16		187.2	219	580.8	335	148.3	253	S
Nash	16-20		62.6	108	572.2	576	56.7	800	LS
Nash	20-24		31.5	47	643.0	274	42.6	171	SL
Nash	24-28		17.2	35	855.0	283	49.3	123	SL
Nash	28-32		5.7	28	800.0	328	37.8	135	LS
Nash	0-4	200/	39.2	106	534.5	476	76.4	472	SL
Nash	4-8	Wickham	30.7	113	563.5	584	83.0	524	SL
Nash	8-12		6.7	72	715.5	848	66.3	656	SCL
Nash	12-16		0.0	52	808.5	1108	53.1	904	SCL
Nash	16-20		0.0	48	884.2	1132	62.0	976	SCL
Nash	20-24		0.0	45	805.2	1064	57.5	960	SCL
Nash	24-28		0.0	47	801.8	940	58.5	876	SCL
Nash	28-32		0.0	44	745.6	824	52.1	784	SCL
Nash	0-4	200/	162.9	262	475.7	440	89.0	224	SL
Nash	4-8	Norfolk	125.7	189	486.9	416	82.1	200	SL
Nash	8-12		112.0	170	549.9	492	74.7	218	SL
Nash	12-16		26.3	63	664.6	500	49.1	204	SCL
Nash	16-20		0.0	18	712.0	488	29.4	198	SCL
Nash	20-24		0.0	13	727.7	516	26.6	200	L
Nash	24-28		0.0	11	763.9	592	22.7	215	SCL
Nash	28-32		0.0	11	785.6	676	22.2	257	CL

Nash	0-4	200/	139.3	294	653.4	672	149.1	580	SL
Nash	4-8	Altavista	73.8	173	674.9	652	114.5	548	SL
Nash	8-12		45.5	131	738.3	692	85.6	436	SL
Nash	12-16		40.7	85	764.1	704	66.0	420	SCL
Nash	16-20		3.8	41	793.2	784	47.7	436	SCL
Nash	20-24		0.0	14	872.1	796	42.5	370	SCL
Nash	24-28		0.0	12	787.4	728	33.5	332	SCL
Nash	28-32		0.0	9	794.8	656	39.0	293	SCL
Nash	0-4	200/	32.4	180	648.6	732	146.3	1196	SL
Nash	4-8	Wickham	6.4	104	628.2	808	96.1	1100	SL
Nash	8-12		0.0	26	709.1	764	81.6	796	SCL
Nash	12-16		0.0	23	726.9	748	80.0	788	SCL
Nash	16-20		0.0	18	724.9	748	70.9	812	SCL
Nash	20-24		0.0	22	696.5	724	60.0	856	SCL
Nash	24-28		0.0	25	704.4	732	63.4	1136	SCL
Nash	28-32		0.0	26	684.9	668	65.1	1096	SCL
Nash	0-4	200/	447.3	708	277.9	338	141.1	468	SL
Nash	4-8	Goldsboro	99.6	102	477.1	287	75.7	246	SL
Nash	8-12		157.4	258	433.9	360	120.7	296	SL
Nash	12-16		5.5	28	583.7	348	28.3	164	L
Nash	16-20		0.0	9	588.5	354	25.3	168	L
Nash	20-24		0.0	11	524.0	334	28.1	234	SL
Nash	24-28		0.0	11	527.4	354	29.6	294	SL
Nash	28-32		0.0	7	584.9	348	26.2	238	SL
Nash	0-4	200/	156.6	212	386.8	287	71.2	222	LS
Nash	4-8	Norfolk	143.5	181	596.4	400	72.3	177	SL
Nash	8-12		21.9	53	745.7	588	37.8	236	SCL
Nash	12-16		0.0	17	780.0	692	27.9	290	SCL
Nash	16-20		0.0	13	776.6	676	25.1	322	SCL
Nash	20-24		0.0	10	776.9	708	22.6	311	SCL
Nash	24-28		0.0	10	365.8	704	8.9	270	SL
Nash	28-32		0.0	10	866.7	756	24.6	280	SCL
Pitt	0-4	200/	329.9	380	635.6	588	176.7	484	S
Pitt	4-8	Goldsboro	256.8	350	640.0	684	179.9	560	LS
Pitt	8-12		337.5	428	803.5	896	152.7	604	LS
Pitt	12-16		165.4	190	725.0	640	84.3	321	SL
Pitt	16-20		114.1	130	883.1	704	73.2	348	SL
Pitt	20-24		130.5	206	872.2	728	71.7	384	SL
Pitt	24-28		69.6	134	780.9	580	62.9	340	SL
Pitt	28-32		66.1	101	704.3	386	51.7	142	LS
Pitt	0-4	100/	232.5	293	449.0	330	105.9	184	S
Pitt	4-8	Wagram	204.8	235	488.2	345	109.0	171	S
Pitt	8-12		109.7	128	549.5	385	71.0	136	S
Pitt	12-16		54.1	73	541.9	353	59.1	204	S
Pitt	16-20		31.5	46	485.9	190	45.9	10	S
Pitt	20-24		24.1	35	522.4	195	48.8	3	LS
Pitt	24-28		10.7	31	706.0	408	45.6	157	LS
Pitt	28-32		0.0	16	727.8	484	34.1	215	SL
Pitt	0-4	100/	195.5	280	437.9	367	102.7	219	LS
Pitt	4-8	Wagram	186.9	235	421.0	347	93.4	182	LS

Pitt	8-12		105.7	121	436.4	240	63.1	64	LS
Pitt	12-16		79.3	91	438.9	265	46.7	171	LS
Pitt	16-20		52.3	88	533.4	181	40.9	18	SL
Pitt	20-24		8.0	23	677.1	341	37.3	127	SL
Pitt	24-28		0.0	17	759.3	540	30.8	212	SL
Pitt	28-32		0.0	14	809.6	584	29.3	219	SL
Pitt	0-4	100/	153.7	214	476.6	374	105.9	222	LS
Pitt	4-8	Wagram	165.2	214	481.2	377	114.7	228	LS
Pitt	8-12		81.2	102	446.7	333	78.1	194	SCL
Pitt	12-16		16.8	31	523.1	274	52.6	152	SL
Pitt	16-20		0.0	11	716.9	432	42.3	197	SL
Pitt	20-24		0.0	10	833.8	600	36.2	246	SCL
Pitt	24-28		0.0	13	895.3	700	33.0	286	SCL
Pitt	28-32		0.0	13	884.4	732	33.7	304	SCL
Pitt	0-4	200/	80.2	203	1036.4	1496	262.6	1248	SL
Pitt	4-8	Lynchburg	8.6	46	1145.9	1480	210.0	1148	SL
Pitt	8-12		0.0	17	978.1	980	103.1	612	SL
Pitt	12-16		0.0	8	1032.8	928	53.8	384	SCL
Pitt	16-20		0.0	7	1022.4	960	29.9	264	SCL
Pitt	20-24		0.0	6	970.9	880	27.8	262	SCL
Pitt	24-28		0.0	6	996.7	916	33.4	279	SCL
Pitt	28-32		0.0	7	1033.3	856	40.9	296	SCL
Pitt	0-4	100/	190.1	262	538.7	436	97.4	196	S
Pitt	4-8	Alaga	179.5	231	529.8	420	91.3	169	LS
Pitt	8-12		108.9	124	602.6	366	64.7	93	LS
Pitt	12-16		38.9	58	698.6	381	44.4	60	SL
Pitt	16-20		18.5	35	685.9	424	35.1	147	SL
Pitt	20-24		7.1	27	648.7	377	35.1	138	SL
Pitt	24-28		0.0	13	831.3	572	32.9	180	SL
Pitt	28-32		0.0	11	872.0	660	26.5	219	SL
Pitt	0-4	200	169.4	325	770.2	876	257.6	968	SL
Pitt	4-8	Rains	87.3	150	700.9	732	243.2	788	SL
Pitt	8-12		20.1	43	722.8	588	165.0	548	SL
Pitt	12-16		2.0	18	736.8	560	108.6	496	SL
Pitt	16-20		0.0	19	831.0	680	105.3	472	SL
Pitt	20-24		0.0	7	770.8	556	73.8	452	SL
Pitt	24-28		0.0	4	707.4	524	46.7	340	SL
Pitt	28-32		0.0	4	747.9	520	40.9	291	SL
Pitt	0-4	100/	115.5	185	515.8	428	89.4	300	LS
Pitt	4-8	Alaga	109.1	161	502.5	408	92.1	181	LS
Pitt	8-12		37.3	57	540.6	288	64.4	104	SL
Pitt	12-16		2.7	16	617.5	394	35.3	170	SL
Pitt	16-20		0.0	11	712.1	560	36.0	269	SL
Pitt	20-24		0.0	10	831.7	696	34.9	323	SCL
Pitt	24-28		0.0	10	815.5	668	33.8	294	SCL
Pitt	28-32		0.0	10	702.5	500	33.7	224	SL
Pitt	0-4	100/	208.9	371	652.4	672	118.0	476	S
Pitt	4-8	Wagram	188.4	283	557.6	520	102.8	289	S
Pitt	8-12		114.5	158	600.8	556	68.7	326	LS
Pitt	12-16		52.2	88	674.1	480	63.4	282	LS

Pitt	16-20		25.6	54	616.3	408	54.4	235	LS
Pitt	20-24		18.3	47	661.2	388	53.8	230	LS
Pitt	24-28		15.2	41	680.6	397	57.4	244	LS
Pitt	28-32		14.4	39	698.6	361	51.1	200	LS
Pitt	0-4	200/	165.5	260	658.6	624	121.6	344	LS
Pitt	4-8	Wickham	149.0	236	675.1	688	116.2	388	LS
Pitt	8-12		101.6	195	820.1	812	100.7	432	SL
Pitt	12-16		45.6	88	875.9	780	75.7	337	LS
Pitt	16-20		11.5	32	712.5	484	51.9	229	LS
Pitt	20-24		3.7	20	641.3	356	48.5	198	LS
Pitt	24-28		1.7	16	576.5	308	39.5	179	LS
Pitt	28-32		2.9	15	534.3	272	37.9	158	S
Pitt	0-4	200/	149.9	265	835.3	940	122.3	548	LS
Pitt	4-8	Ocilla	138.8	232	820.7	880	118.2	516	LS
Pitt	8-12		43.3	80	811.1	808	87.3	492	LS
Pitt	12-16		5.4	33	866.9	796	74.1	520	SL
Pitt	16-20		0.0	35	1003.3	1052	61.1	668	SL
Pitt	20-24		0.0	34	984.6	968	61.0	612	SL
Pitt	24-28		0.0	27	917.8	808	62.6	560	SL
Pitt	28-32		0.0	25	945.2	868	49.4	564	SL
Union	0-4	500/	445.4	878	677.9	996	119.1	752	SiL
Union	4-8	Tatum	189.6	279	769.9	936	54.7	572	SiCL
Union	8-12		51.4	103	903.2	904	29.6	488	SiCL
Union	12-16		7.5	64	991.2	872	30.9	560	SiCL
Union	16-20		0.0	28	931.5	944	23.1	620	SiC
Union	20-24		0.0	39	986.7	956	25.1	716	SiCL
Union	24-28		0.0	18	952.7	976	25.0	852	SiCL
Union	28-32		0.0	8	988.4	1012	21.0	752	SiC
Union	0-4	200/	313.5	596	577.8	904	189.4	1212	SiL
Union	4-8	Cid	44.2	118	687.5	672	98.1	724	SiL
Union	8-12		2.4	31	911.3	588	57.3	420	SiL
Union	12-16		0.0	9	894.0	664	36.6	354	SiL
Union	16-20		0.0	7	964.2	832	32.5	384	SiCL
Union	20-24		0.0	6	1046.0	796	36.5	334	SiCL
Union	24-28		0.0	11	1276.4	1272	31.3	372	SiL
Union	28-32		0.0	5	1257.0	1328	28.4	364	SiL
Union	0-4	200/	190.2	458	659.9	780	192.4	1112	Si
Union	4-8	Secrest	47.5	161	710.8	728	117.1	808	SiL
Union	8-12		3.7	44	736.0	676	43.1	480	SiCL
Union	12-16		0.0	24	758.4	636	34.5	420	SiCL
Union	16-20		0.0	13	785.2	692	28.5	480	SiCL
Union	20-24		0.0	10	858.2	788	28.3	608	SiCL
Union	24-28		0.0	12	938.7	792	33.6	520	SiL
Union	28-32		0.0	9	902.8	764	31.8	476	SiL
Union	0-4	500/	183.0	410	935.2	976	171.0	944	SiL
Union	4-8	Tatum	37.7	132	965.5	800	94.5	664	SiL
Union	8-12		0.0	15	1025.7	680	37.0	299	SiL
Union	12-16		0.0	13	1156.6	804	33.8	354	SiL
Union	16-20		0.0	9	1029.8	748	32.1	412	SiL
Union	20-24		0.0	8	1035.9	764	27.3	460	SiL

Union	24-28		0.0	7	1070.4	808	25.3	480	SiL
Union	28-32		0.0	7	1058.4	844	26.1	496	SiL
Union	0-4	500/	200.7	383	695.1	884	125.9	876	SiL
Union	4-8	Badin	19.0	89	925.7	776	73.4	568	SiL
Union	8-12		0.0	22	942.0	832	42.9	520	SiL
Union	12-16		0.0	25	783.5	616	50.5	496	Si
Union	16-20		0.0	12	829.6	668	49.2	596	SiL
Union	20-24		ns[*]	ns	ns	ns	ns	ns	ns
Union	24-28		ns	ns	ns	ns	ns	ns	ns
Union	28-32		ns	ns	ns	ns	ns	ns	ns
Union	0-4	500/	134.4	268	663.9	644	106.4	656	SiL
Union	4-8	Badin	11.4	56	806.4	680	44.3	390	SiCL
Union	8-12		0.0	22	934.5	788	29.0	476	SiC
Union	12-16		0.0	18	989.7	808	27.9	576	SiC
Union	16-20		0.0	9	1052.0	976	22.5	640	SiCL
Union	20-24		0.0	9	1157.7	1040	23.4	708	SiC
Union	24-28		0.0	26	1129.5	1140	21.0	704	SiC
Union	28-32		0.0	9	1035.4	1076	20.6	780	SiCL
Union	0-4	200/	380.3	944	548.8	584	343.3	1204	SiL
Union	4-8	Cid	85.3	212	535.1	384	163.5	632	SiL
Union	8-12		0.0	27	680.5	460	55.0	293	SiL
Union	12-16		0.0	12	734.2	508	45.3	349	SiL
Union	16-20		0.0	7	866.8	724	38.2	397	SiL
Union	20-24		0.0	7	925.0	900	40.0	464	SiL
Union	24-28		0.0	5	957.9	688	42.5	324	SiL
Union	28-32		0.0	6	963.2	1000	55.0	524	SiL
Union	0-4	200/	147.2	370	522.3	564	236.9	1384	SiL
Union	4-8	Cid	30.2	143	485.8	420	97.9	572	SiL
Union	8-12		0.0	19	587.4	424	63.8	396	SiL
Union	12-16		0.0	6	756.0	488	41.9	268	SiL
Union	16-20		0.0	6	844.0	648	37.5	298	SiL
Union	20-24		0.0	20	904.4	716	40.1	308	SiL
Union	24-28		0.0	6	863.6	712	41.8	316	SiL
Union	28-32		0.0	7	897.8	704	52.2	396	SiL
Union	0-4	500	152.9	404	711.1	896	127.1	952	SiL
Union	4-8	Tatum	50.2	181	715.4	860	87.7	800	SiL
Union	8-12		0.0	19	796.9	664	45.4	416	SiL
Union	12-16		0.0	14	911.6	664	29.4	364	SiL
Union	16-20		0.0	10	955.7	644	28.4	432	SiCL
Union	20-24		0.0	10	982.2	712	23.5	516	SiCL
Union	24-28		0.0	8	1053.1	772	24.2	660	SiCL
Union	28-32		0.0	11	962.0	732	21.5	632	SiCL
Union	0-4	500/	436.6	850	767.6	1140	204.9	1588	SiL
Union	4-8	Badin	161.8	436	820.3	420	135.5	672	SiL
Union	8-12		53.0	151	882.9	984	134.7	960	SiL
Union	12-16		15.6	60	853.5	772	92.3	904	SiL
Union	16-20		1.5	38	897.9	640	51.6	428	SiL
Union	20-24		0.0	16	891.0	680	46.9	560	SiL
Union	24-28		0.0	9	949.7	748	32.6	444	SiL
Union	28-32		0.0	7	926.3	760	29.6	444	SiL

Washington	0-4	50/	127.9	718	1813.1	5560	173.7	1040	SCL
Washington	4-8	Belhaven	71.3	466	2063.8	5680	182.3	952	SL
Washington	8-12		19.2	162	1757.1	3984	82.3	374	SL
Washington	12-16		0.0	189	2028.9	5720	39.5	232	SCL
Washington	16-20		0.0	168	2020.4	5000	37.1	202	SCL
Washington	20-24		0.0	142	1894.3	4400	56.0	251	SCL
Washington	24-28		0.0	63	1562.3	2992	99.1	385	SL
Washington	28-32		0.0	24	1068.9	980	148.3	856	SL
Washington	0-4	200/	169.0	704	1791.0	5320	206.4	1120	SL
Washington	4-8	Wasda	107.4	405	2187.0	5640	156.8	612	SCL
Washington	8-12		34.0	301	2526.0	7520	66.4	247	SCL
Washington	12-16		0.0	221	2738.8	7840	52.6	312	SCL
Washington	16-20		0.0	194	2383.5	6200	44.0	224	SL
Washington	20-24		0.0	137	2143.9	5520	60.1	279	SL
Washington	24-28		0.0	45	1414.9	2092	107.0	322	SL
Washington	28-32		0.0	13	945.9	1052	148.1	544	SL
Washington	0-4	200/	56.6	359	1453.5	4000	122.7	868	CL
Washington	4-8	Cape Fear	27.5	275	1582.5	4080	116.7	644	CL
Washington	8-12		4.6	143	1786.9	4160	96.7	484	CL
Washington	12-16		0.0	127	1818.4	4160	112.8	536	CL
Washington	16-20		0.0	100	1801.0	4160	108.8	399	CL
Washington	20-24		0.0	68	1686.1	3728	108.3	432	CL
Washington	24-28		0.0	57	1623.4	3416	118.6	644	CL
Washington	28-32		0.0	24	1500.9	2556	173.6	1460	CL
Washington	0-4	200/	163.8	388	907.8	1152	213.7	716	SL
Washington	4-8	Portsmouth	129.2	294	976.6	1272	244.4	788	SL
Washington	8-12		69.0	147	1075.2	1196	263.0	776	SL
Washington	12-16		8.2	39	826.0	956	189.3	980	SL
Washington	16-20		0.0	21	864.1	832	204.4	924	SL
Washington	20-24		0.0	15	758.8	796	195.8	1096	L
Washington	24-28		0.0	9	722.2	704	199.8	1312	SCL
Washington	28-32		0.0	11	616.6	568	211.6	1740	SCL
Washington	0-4	200/	174.9	336	1468.4	2356	272.5	932	SL
Washington	4-8	Wasda	148.0	283	1417.2	2436	241.4	964	SL
Washington	8-12		91.9	226	1801.1	3436	129.5	408	SiL
Washington	12-16		15.9	80	1515.7	2472	115.5	305	SL
Washington	16-20		0.0	22	1212.4	1456	214.5	772	SL
Washington	20-24		0.0	8	1050.8	1312	215.7	840	SL
Washington	24-28		0.0	9	866.7	924	209.9	1104	SL
Washington	28-32		0.0	9	798.4	840	273.8	1292	SCL
Washington	0-4	200/	208.4	556	1612.2	3548	251.3	1224	L
Washington	4-8	Wasda	210.0	556	1787.5	3840	242.4	1088	L
Washington	8-12		57.1	239	2056.0	4520	94.5	334	SiL
Washington	12-16		16.4	92	1741.1	3048	51.0	120	L
Washington	16-20		7.1	29	1026.9	1260	127.3	398	SL
Washington	20-24		0.7	21	1065.4	1280	203.9	788	SCL
Washington	24-28		0.0	14	970.7	1008	261.3	1336	SCL
Washington	28-32		0.0	11	760.7	740	251.2	1380	SCL
Washington	0-4	50/	102.2	312	1185.4	2244	196.8	840	SL
Washington	4-8	Belhaven	56.4	177	1322.8	2472	162.1	504	SL

Washington	8-12		18.0	141	1511.7	3288	103.0	334	SL
Washington	12-16		0.5	97	1779.6	4600	91.1	390	SCL
Washington	16-20		0.0	126	1811.5	3912	75.4	286	SCL
Washington	20-24		0.0	66	1564.3	2964	85.4	305	SL
Washington	24-28		0.0	23	1290.1	1980	128.3	820	SL
Washington	28-32		0.0	12	981.7	1132	159.2	1124	SCL
Washington	0-4	50/	102.5	320	1345.3	2804	209.1	836	SCL
Washington	4-8	Belhaven	72.0	204	1375.5	2664	184.5	596	SL
Washington	8-12		19.1	143	1661.9	3884	123.4	448	SCL
Washington	12-16		0.0	110	1750.5	3996	110.4	336	SCL
Washington	16-20		0.0	81	1533.9	3052	120.5	379	SL
Washington	20-24		0.0	37	1388.0	2140	159.9	608	SL
Washington	24-28		0.0	14	1261.2	1548	180.0	648	SL
Washington	28-32		0.0	5	864.3	828	217.9	928	SL
Wilkes	0-4	500/	591.5	1168	831.7	1048	201.7	988	SL
Wilkes	4-8	Pacelot	342.9	508	868.3	992	127.3	484	SL
Wilkes	8-12		193.8	298	833.1	716	69.2	328	SCL
Wilkes	12-16		63.7	143	916.3	752	53.0	420	CL
Wilkes	16-20		3.1	69	969.0	868	36.7	608	C
Wilkes	20-24		0.0	36	956.4	724	38.0	568	SCL
Wilkes	24-28		0.0	21	958.9	728	37.3	608	SCL
Wilkes	28-32		0.0	15	903.7	760	32.5	628	SCL
Wilkes	0-4	500/	534.1	1002	828.9	1164	186.9	912	SL
Wilkes	4-8	Pacelot	414.6	714	786.2	868	197.5	868	SL
Wilkes	8-12		152.2	296	773.4	752	123.4	552	SCL
Wilkes	12-16		23.4	104	878.6	716	78.5	712	SCL
Wilkes	16-20		0.0	52	869.9	796	31.1	640	C
Wilkes	20-24		0.0	29	941.7	912	30.3	736	C
Wilkes	24-28		0.0	14	1007.8	864	30.0	780	C
Wilkes	28-32		0.0	14	1022.5	780	30.8	676	SCL
Wilkes	0-4	500/	598.1	1082	928.9	1080	157.1	692	SC
Wilkes	4-8	Pacelot	695.9	1258	973.8	1300	161.7	880	SCL
Wilkes	8-12		352.5	562	971.8	1004	88.2	568	SCL
Wilkes	12-16		75.2	178	1060.6	824	70.2	672	C
Wilkes	16-20		27.2	168	1033.0	836	88.6	888	C
Wilkes	20-24		5.4	98	1064.9	688	72.1	768	C
Wilkes	24-28		0.0	44	1111.7	664	52.0	784	CL
Wilkes	28-32		0.0	38	1055.4	644	41.6	588	CL
Wilkes	0-4	500/	436.4	666	1042.3	864	83.5	464	CL
Wilkes	4-8	Pacelot	33.6	50	1081.5	660	29.3	358	C
Wilkes	8-12		0.0	21	1153.3	776	23.7	536	C
Wilkes	12-16		0.0	20	1184.0	800	25.6	540	C
Wilkes	16-20		0.0	15	1163.5	716	25.5	536	SC
Wilkes	20-24		0.0	22	1162.7	788	26.9	624	SCL
Wilkes	24-28		0.0	16	1283.3	756	29.1	584	SCL
Wilkes	28-32		0.0	15	1160.4	656	27.6	464	SC
Wilkes	0-4	500/	132.4	239	801.7	544	97.6	376	SCL
Wilkes	4-8	Pacelot	26.6	82	885.3	636	62.0	400	SL
Wilkes	8-12		0.0	21	1016.5	768	34.5	460	SCL
Wilkes	12-16		0.0	18	978.3	824	27.3	548	CL

Wilkes	16-20		0.0	15	1009.4	744	29.9	508	CL
Wilkes	20-24		0.0	13	989.0	676	26.2	460	SCL
Wilkes	24-28		0.0	7	1016.0	608	25.8	460	SCL
Wilkes	28-32		0.0	17	1050.9	676	28.2	440	SCL
Wilkes	0-4	200/	460.8	852	840.5	944	288.9	1968	SL
Wilkes	4-8	Toccoa	234.3	492	821.8	1036	241.9	2108	SL
Wilkes	8-12		124.9	273	774.9	956	165.3	1808	SL
Wilkes	12-16		31.9	137	872.4	972	123.7	1784	SL
Wilkes	16-20		10.6	136	974.1	1252	116.3	2056	SL
Wilkes	20-24		14.4	156	972.9	1152	122.4	2144	SL
Wilkes	24-28		16.0	143	869.3	1012	118.5	1960	LS
Wilkes	28-32		49.0	179	736.9	744	130.3	1408	LS
Wilkes	0-4	500/	394.0	614	688.5	748	169.7	904	SL
Wilkes	4-8	Pacelot	142.2	247	712.3	784	109.9	676	SL
Wilkes	8-12		15.4	57	843.0	820	65.5	492	L
Wilkes	12-16		2.2	46	924.4	936	50.8	568	SCL
Wilkes	16-20		0.0	31	995.1	1092	46.8	712	CL
Wilkes	20-24		0.0	36	1020.6	1384	46.2	916	CL
Wilkes	24-28		0.0	23	1025.0	1320	37.2	836	CL
Wilkes	28-32		0.0	17	1092.8	1116	42.5	796	CL
Wilkes	0-4	200/	231.7	442	691.3	696	162.1	1172	LS
Wilkes	4-8	Buncombe	261.9	428	683.7	728	168.2	1204	LS
Wilkes	8-12		172.3	310	729.3	824	136.7	1244	LS
Wilkes	12-16		64.6	156	570.2	576	102.5	988	S
Wilkes	16-20		22.4	105	584.0	584	96.8	888	S
Wilkes	20-24		20.2	101	575.3	640	101.0	960	S
Wilkes	24-28		25.2	129	684.0	868	110.1	1192	S
Wilkes	28-32		28.0	170	840.0	1080	127.2	1608	LS
Wilkes	0-4	200/	1294.4	2210	976.4	1544	199.4	1428	SL
Wilkes	4-8	Toccoa	768.7	1196	972.3	1348	158.9	1080	LS
Wilkes	8-12		420.6	656	926.0	1228	119.9	904	LS
Wilkes	12-16		284.1	420	899.2	888	105.2	720	SL
Wilkes	16-20		129.6	209	878.8	660	87.0	516	SL
Wilkes	20-24		27.7	83	796.1	616	69.7	568	SL
Wilkes	24-28		11.8	94	871.9	696	76.4	772	SL
Wilkes	28-32		1.7	84	797.9	764	71.6	900	SL
Wilkes	0-4	500/	1228.8	2612	943.9	1632	182.6	1296	L
Wilkes	4-8	Pacelot	803.2	1368	919.7	1416	148.9	948	SL
Wilkes	8-12		490.1	822	1067.0	1452	103.8	800	L
Wilkes	12-16		219.4	382	991.7	976	59.4	448	CL
Wilkes	16-20		28.0	125	985.2	1036	39.6	600	CL
Wilkes	20-24		1.0	87	1048.6	1088	35.8	620	CL
Wilkes	24-28		0.0	110	1081.7	1132	32.9	592	C
Wilkes	28-32		0.0	58	1252.5	1184	34.8	544	C

*no sample

Chapter 2

Predicted Impact and Evaluation of North Carolina's Phosphorus Indexing Tool

INTRODUCTION

The Problem

North Carolina is a major animal producing state, being the largest producer of turkeys in the United States with 45.5 million annually ($429 million), the second largest producer of swine with 9.7 million ($1.4 billion) and the fourth largest producer of broilers with 735 million ($1.4 billion) (N.C. Dept. Agric. Consumer Services, 2002). Thus, large amounts of animal manures are generated, which has led to the annual accumulation of more manure nitrogen (N) and phosphorus (P) in some areas than can be used by growing crops (Figure 1). Based on the median soil test results for 2003 (NCDA, 2003) the majority of counties in the state exceeded the Mehlich-3 soil test phosphorus (STP) value of 53 mg kg^{-1}, which is considered to be optimum for crop production (Figure 2). It is evident from both Figures 1 and 2 that certain areas of the state present a greater risk of P loss to the environment. This buildup of soil P in agricultural soils suggests that P is susceptible to loss off-field and has been cited as a major factor contributing to decreased water quality in

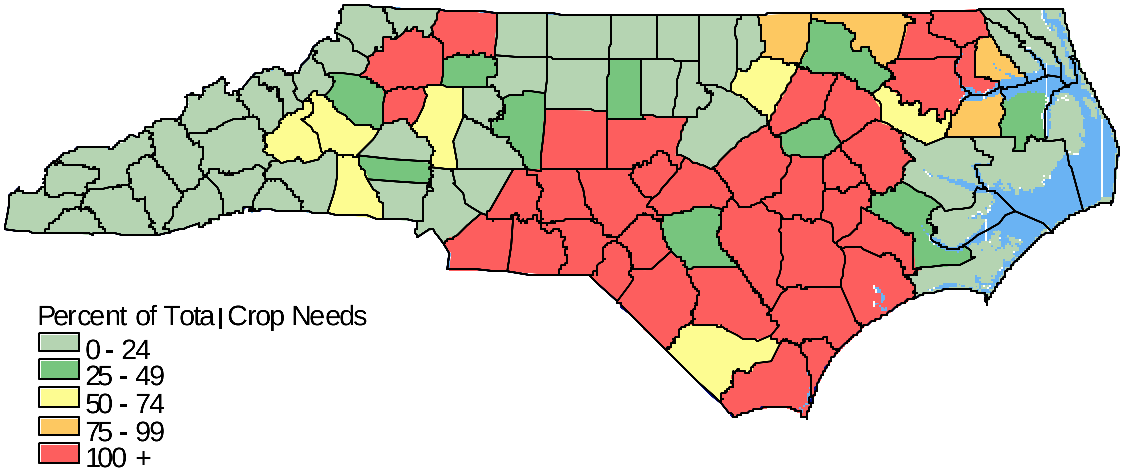

Figure 1. Percent of agronomic and forage crop phosphorus needs supplied by recoverable plant available manure nutrients. From Crouse (2000).

the state (NCDENR, 2003). For instance, it is estimated that non-point source pollution from agriculture and livestock contributes 55% of the P in the Tar-Pamlico River Basin, one of the state's largest river basins (NCDENR, 1994). This situation has severe economic and environmental ramifications as the Albemarle and Pamlico Sounds, into which most of the state's surface waters empty, make up the nation's second largest estuary (Association of National Estuary Programs).

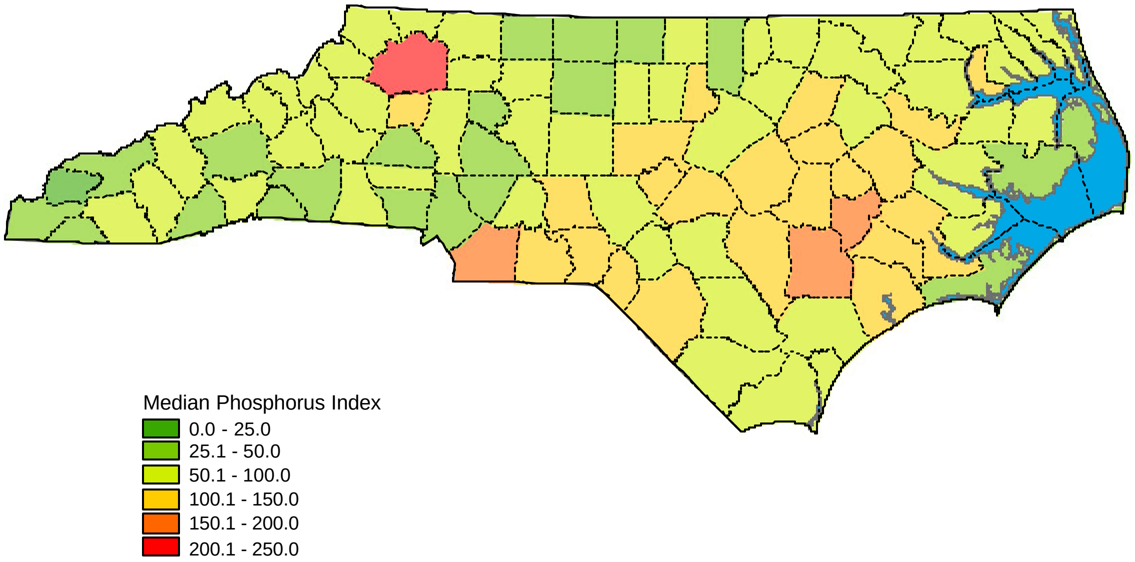

Figure 2. Median soil P index values for all crops by county (2003). Data is from soils submitted to the NC Dept. of Agriculture and Consumer Services, Soil Testing Division, 2003. Data compiled by R. Austin and D. Osmond, North Carolina State University, Department of Soil Science.

A number of state initiatives have been passed in response to media coverage of lagoon breaches and fish kills in the Tar-Pamlico and Neuse rivers and the discovery of Pfiesteria, a toxic dinoflaggelate thought to be fatal to fish (NCDENR, 2000). In 1993, under pressure from a rapidly expanding animal industry, e.g. an increase of 260% in swine facilities in the Neuse River Basin between 1990 and 1998, changes were made to the state non-discharge rules, the so called .0200 rules (NCDENR, 2002). Under the revisions to the .0200 rules, facilities above certain animal threshold populations were required to have animal waste management plans in place. As the minimum criteria for developing waste plans, the state adopted USDA

Natural Resources Conservation Service (NRCS) standards related to animal waste management (Standard 590) (USDA-NRCS, 2001).

In 1996, the North Carolina General Assembly enacted Senate Bill 1217, which placed further regulation on animal waste facilities, requiring, among other things, dry litter poultry operations with over 30,000 birds to develop a modified waste management plan. In 1997, State Bill 515 created a moratorium on new and existing swine farms which has been extended numerous times and continues to be in effect at the present time.

Additionally, the North Carolina Environmental Management Commission (EMC) has designated the Neuse River and Tar-Pamlico River basins as Nutrient Sensitive Waters (NSW), both of which have experienced extensive fish kills in recent years (71 events in the Neuse River Basin between 1990 and 2000, 70 events in the Tar-Pamlico between 1996 and 2002). As an effect of the NSW Basinwide Water Quality Plans that have been implemented, operations within these watersheds that apply animal waste must have a nutrient management plan in place by 1998. Again, the state has deferred to NRCS standards for the requirements of animal waste plans. Pending federal rules designed to prevent animal waste from entering surface waters will require confined animal feeding operations (CAFO's) above specified threshold populations to obtain a permit. Table 1 shows the threshold populations above which the implementation of a nutrient management plan is, or will be, required for the various state and federal regulations.

The revision of NRCS Standard 590 required that sites receiving organic by-products or fertilized fields in sensitive watersheds be assessed for P loss potential. Based on this revision, and the referral to NRCS standards in the state rules, all operations in North Carolina that apply liquid animal waste, receive federal or state cost-share monies, or meet federal CAFO animal thresholds will be required to assess site P loss potential using the Phosphorus Loss Assessment Tool (PLAT), as well as those operations that meet federal CAFO standards. Much confusion and debate still exists in North Carolina as to exactly which operations are covered under which regulations.

Given the importance of the P loss assessment, it was essential that North Carolina develop a tool that was more comprehensive in its risk assessment of potential P loss than only soil test P, yet still relatively easy to use on a large scale. Therefore, in 1999 an interagency committee was formed to develop an indexing tool to help identify "problem" fields based on the processes that contribute to P delivery to surface water. The development of PLAT involved scientists, engineers and conservation planners from NC State University, the NC Department of Agriculture and Consumer Service, USDA-NRCS and the NC Department of Environment and Natural Resources, Division of Soil and Water Conservation (Havlin et al., 2001).

Table 1. Comparison of threshold animal populations set in state non-discharge regulations (.0200 rules), NSW Basinwide Water Quality Plans for the Neuse and Tar-Pamlico River Basins and federal CAFO regulations.

Animal type	**NC .0200 Rules/SB 1217†**	**Neuse‡ and Tar-Pamlico§ River Basins**	**CAFO middle tier operations¶**	**CAFO large tier operations**
Swine	250	150	2,500	750
Cattle	100	20	1,000	300
Poultry:				
Chicken	30,000#	3,500††	9,000 (25,000) 37,500‡‡	30,000 (82,000) 125,000‡‡
Turkey		650	16,500	55,000
Sheep	1,000	120	10,000	150
Horses	75	20	500	150

† Environmental Management Commission. Waste Not Discharged to Surface Waters, .0200 Rules. http://h2o.enr.state.nc.us/admin/rules/#redbook.

‡ NCDENR, 2002.

§ NCDENR, 2003.

¶ USEPA. Concentrated Animal Feeding Operations Clean Water Act Requirements. Information Series Pamphlet.

Liquid waste systems. Per Senate Bill 1217, dry litter operations must develop a waste management plan but are not required to have a complete .0200 waste management plan.

††Neuse Rules only apply to poultry operations using liquid manure handling systems; rules in Tar-Pamlico River Basin apply to all poultry operations, liquid or dry.

‡‡ Operations using liquid manure handling systems; (Laying hens, operations using other than liquid manure handling systems); Chickens other than laying hens and/or operations using other than liquid manure handling systems.

The specific objectives of this study were (i) to estimate the percentage and types of farms that will need to use P-based nutrient management as a result of new regulations and (ii) to evaluate the performance of North Carolina's P indexing tool, PLAT, on diverse land conditions. Although we had no actual field data measuring the specific components of PLAT, we can compare PLAT's predictions to specific soils/regions/managements in the state in which we know STP has accumulated, and that we suspect of contributing to off-field P loss.

Phosphorus Loss Site-Assessment in North Carolina

North Carolina's PLAT identifies four P loss pathways: i) loss via erosion (sediment P), (ii) loss via surface runoff (soluble P), (iii) loss via subsurface drainage (soluble P), and (iv) loss via applied P source (particulate and soluble P from sources carried by runoff). Source and transport factors are evaluated within each loss pathway in a multiplicative manner and results for the four parts are summed to estimate a relative amount of overall P being delivered off the site. Although the tool provides comparative estimates of P loss in lbs P ac^{-1} yr^{-1}, it is in no way intended to be an approximation of actual P load leaving a field, but, rather, a relative indication of potential P loss. This allows the user to make comparative estimates of risk between different management practices and field conditions. In PLAT, estimated values of P loss are converted to index values by multiplying lbs P ac^{-1} by '25'. This value was chosen in order to scale index ratings similar to the state soil-test indexing system. Index values are then used to rate the potential for edge-of-field P loss by classifying them into one of four risk categories (Table 2).

Table 2. PLAT risk categories and their generalized interpretation.

Predicted Loss (lbs P ac^{-1} yr^{-1})	**Index Value**	**Rating**	**Consequence**
0 – 1	**0 – 25**	**Low**	N-based application
1 – 2	**25 – 50**	**Medium**	N-based application
2 – 4	**50 – 100**	**High**	P applied at crop removal rates
>4	**> 100**	**Very High**	No additional P applications

One of the underlying features of PLAT is the use of what we have termed 'P threshold groups'. This concept essentially relates the amount of soluble P that can potentially be dissolved from a soil given a certain soil test phosphorus (STP) value. Four threshold soil groups are defined based on their relative ability to retain P and are shown with their respective P threshold values in Table 3. The P threshold value is defined as the M3P (mg kg^{-1}) level at which 1 mg L^{-1} of soluble P is expected to be lost to the soil solution (i.e. runoff) for a given group and has been determined experimentally for North Carolina soils. In general, these groups are based on soil texture, drainage and depth to the Bt horizon. For simplicity, the groups are referred to by their texture, as shown in Table 3. This method of differentiating soils' abilities to hold P essentially serves to approximate degree of P saturation.

Table 3. North Carolina phosphorus threshold groups used in PLAT.

Generalized soil texture	**P threshold (mg kg^{-1})**	**Relative landscape location**
Organic	**50**	Poorly drained Coastal Plain sites
Sand	**100**	Stream and river terraces
Loam	**200**	Stable upland Coastal Plain soils
Clay	**500**	Stable upland residual soils

The following is only a brief overview of the PLAT indexing system and the reader is referred to the following website for a more detailed explanation of the algorithms used in PLAT: http://www.soil.ncsu.edu/nmp/ncnmwg/ncanat/plat/PLAT_Science_behind_the_tool.pdf.

Table 4 illustrates how PLAT functions using two example sites; the first site has natural drainage and the second site requires artificial drainage. Part 1 P loss, an estimate of the particulate P that is lost through erosion, is calculated the same for both drainage conditions. Certain parameters of Parts 2 and 3, those components estimating transport of soluble P either through surface runoff or subsurface drainage, require different methods of estimation depending on whether the site is naturally or artificially drained. Therefore, these components of PLAT have been presented separately in Table 4 as two distinct examples. Additionally, an estimate of runoff volume is required for calculating Part 4, and because the two different drainage conditions accomplish this by different methods, this has been shown in Table 4. However, because the majority of Part 4 is developed the same regardless of drainage condition, only one example table has been provided. For ease of interpretation, all other factors have been held constant in the two examples; with the exception of surface runoff and subsurface drainage volumes mentioned above, all other factors are calculated identically in the two different drainage situations. Although predicted P loads are presented here, only the resultant index values and overall ratings of a site's susceptibility to P loss are presented in a typical PLAT output.

The loss of sediment-bound P is a function of the amount of soil leaving the field through erosion, the total amount of P that is attached to those soil particles, and any sediment re-deposition that occurs within the field. As has been done in most states' P index, PLAT predicts erosion using RUSLE (Renard et al., 991). The total P content of the eroded soil material is estimated by employing an exponential relationship between the clay content of the soil and the amount of Mehlich-3 phosphorus (M3P) at the soil surface (0-20 cm) to develop a factor for estimating total P (Cox, 1994). Multiplying the erosion rate by the total P factor yields an estimate of the particulate P that could potentially leave the field. Depositional areas in the field, referred to here as receiving slopes, are capable of reducing the movement of particulate matter and are accounted for in the tool. Additionally, credits, or reductions in the amount of eroded material leaving the field, can be achieved through the use of a vegetative buffer, water control structure, farm-pond or sediment basin.

Table 4. Example calculation of the North Carolina Phosphorus Loss Assessment Tool. All input factors that apply to the example situation are shown in italics. Where the user is offered a list of options from which to choose a factor value, such as sediment trapping practices in Part 1, the items used in the current example have been shaded. Two different methods exist for calculating runoff volume and subsurface drainage volume in Parts 2 and 3: one is for situations involving natural drainage and the other for situations in which artificial drainage is required. Therefore, the two different methods for calculating Parts 2 and 3 have been presented separately. Part 1 is calculated identically for both drainage situations. Part 4 is calculated using the same method for both situations with the exception of one factor, the 'Runoff Fraction'. This parameter relies on an estimate of runoff volume, which is calculated differently for the two drainage conditions.

Part 1. Phosphorus loss potential due to surface erosion.							
Parameter							Value
Soil Erosion	Calculated from the Revised Universal Soil Loss Equation						*1.5* tons ac^{-1}
Total P factor†	Based on clay percentage and Mehlich-3 STP (0-20 cm) *(Lumbee sl=11%; threshold M3P = 200 mg kg^{-1} (223 mg kg^{-1})*					×	*0.613* lbs P ton^{-1}
Particulate P loss before BMP's						=	*0.92* lbs P ac^{-1}
Reduction factors for sediment trapping practices‡	Vegetative buffer§ *(10 ft)* **0.6**	Farm pond **0.5**	Water control structure **0.65**	Sediment basin **0.90**	Receiving slope¶ *(10-19 ft)* **0.91**	×	*0.546*
Final P loss via erosion						=	***0.50*** **lbs P ac^{-1}#**

†Total P factor = 0.002 * ***Threshold M3P*** * [1.579-(0.0784 **** % clay***) + (0.0058 * (***% clay***)^2] + [0.02 * (***M3P - Threshold M3P)***].

‡Reduction factors for individual sediment trapping practices are multiplied together to obtain the final reduction factor value.

§Reduction factor due to presence of a buffer is a function of buffer width and is assigned based on empirical relationships. The current example of a ***10 ft buffer width*** yields a reduction factor value of ***0.6***.

¶ Reduction factor due to receiving slope is a function of depositional slope distance and was determined using numerous RUSLE (version 1.06) model runs. The current example of ***10-19 ft slope distance*** yields a reduction factor value of ***0.91***.

Yields an index value of **12.5 (*x lbs P/ac*** * 25).

SURFACE AND SUBSURFACE LOSS OF SOLUBLE P FOR A NATURALLY DRAINED SITUATION.			
Parameter			Value
Part 2. Phosphorus loss potential due to surface runoff.			
Runoff concentration†	Based on Mehlich-3 STP (0-20 cm) and soil threshold group *(223 mg kg^{-1})* *(Lumbee sl, threshold = 200 mg kg^{-1})*		*1.12* mg L^{-1}
Runoff volume‡	Based on curve number and county *(pasture, fair condition; CN =69)* *(Duplin County)*	×	*1.87* ac-in
Conversion factor		×	*0.2265*
Final P loss via runoff		=	***0.47*** **lbs P ac^{-1}§**
Part 3. Phosphorus loss potential due to subsurface drainage.			
Drainage concentration¶	Based on subsoil Mehlich-3 STP (70-80 cm)# *(45 mg kg^{-1})*		*0.225* mg L^{-1}
Drainage volume ††	Based on county rainfall, ET and runoff volume (as calculated in Part 2) *(51.83 in)* *(36 in)* *(1.87 in)*	×	*13.96* ac-in
Conversion factor		×	*0.2265*
Final P loss via subsurface drainage		=	***0.71*** **lbs P ac^{-1}‡‡**

† Empirical relationships between STP and solution P were determined by threshold group. The current example, involving the loam threshold group, was calculated by*:* Solution P (mg L^{-1}) = 0.005 * ***M3P (mg kg^{-1}).***

‡ Runoff (in) = ß1 * exp$^{(ß0 * CN)}$. The coefficients *ß1* and *ß0* are determined based on long-term weather data for each county.

§ Yields an index value of **12**.

¶ Solution P (mg L^{-1}) = 0.005 * ***M3P (mg kg^{-1})***.

\# If required based on surface M3P and soil type.

††Subsurface drainage (in) = Annual rainfall (in) – Runoff (in) – ET (in). Uses runoff volume that is calculated in Part 2; Annual rainfall and ET are county dependent.

‡‡Yields an index value of **18**.

SURFACE AND SUBSURFACE LOSS OF SOLUBLE P FOR AN ARTIFICIAL DRAINED SITUATION.			
Part 2. Phosphorus loss potential due to surface runoff.			
Parameter			Value
Runoff concentration†	Based on Mehlich-3 STP (0-20cm) and soil threshold group *(223 mg kg^{-1})* *(Craven fsl, threshold = 200 mg kg^{-1})*		*1.12* mg L
Runoff volume‡	Based on county rainfall, crop cover, and drainage intensity *(51.83 in)* *(pasture)* *(1.86 ft hr^{-1})*	×	*1.56* ac-in
Conversion factor		×	*0.2265*
Final P loss via runoff		=	***0.40* lbs P ac^{-1}§**
Part 3. Phosphorus loss potential due to subsurface drainage.			
Drainage concentration¶	Based on subsoil Mehlich-3 STP (70-80 cm)# *(45 mg kg)*		*0.225* mg L^{-1}
Drainage volume††	Based on county rainfall, crop cover, and drainage intensity *(51.83 in)* *(pasture)* *(1.86 ft hr^{-1})*	×	*17.10* ac-in
Conversion factor		×	*0.2265*
Transmissivity factor‡‡	Specific to soil	×	*0.14*
Final P loss via subsurface drainage		=	***0.12* lbs P ac^{-1}§§**

† Calculated as for the naturally drained situation above.

‡ Runoff (in) = e / (drainage intensity + f)^{g} where drainage intensity = [soil transmissivity * (***drain depth*** – 1)] / [***drain spacing*** / 100]^{2} and the coefficients e, f and g are dependant on the amount of rainfall and cover type (pasture or row crop). See Skaggs et al., (2004) for more detail. In the current example transmissivity = ***3.72 ft^{2} hr^{-1}***, drain depth = ***3.0 ft*** and drain spacing = ***200 ft*** giving a drainage intensity of ***22.32 ft hr^{-1}***.

§ Yields an index value of **10**.

¶ Calculated as for the naturally drained situation above.

If required based on surface M3P and soil type.

†† Subsurface drainage (in) = (a + b * (***drainage intensity***)^{c} / (d + ***drainage intensity***)^{c} where drainage intensity is defined as above and the coefficients a, b, c, and d are dependant on rainfall and cover type.

‡‡ Transmissivity factor = transmissivity of the 30-80 cm (12-30 in) soil depth / transmissivity of the entire soil profile.

§§ Yields an index value of **3**.

Part 4. Phosphorus loss potential due to applied P source.†

Parameter							Value
Source P type (*unit* ac^{-1})	*Broiler house litter (tons ac^{-1})*						
	Soluble P (specific to manure type)						*8.635* lbs P ton^{-1}
	Soluble P attenuation factor	Litter/scraped **0.4**	Sludge/slurry **0.3**	Liquid/Inorganic **0.1**		×	*0.4*
	Deliverable soluble P					=	*3.454* lbs P ton^{-1}
	Non-soluble P (specific to manure type)						*25.905* lbs P ton^{-1}
	Non-soluble P attenuation factor	Litter/scraped **0.1**	Sludge/slurry **0.1**	Liquid/Inorganic **0.1**		×	*0.1*
	Buffer retention factor§ (same factor as used in Part 1)					×	*0.6*
	Deliverable non-soluble P					=	*1.554* lbs P ton^{-1}
Naturally drained site							
Deliverable source P‡	Deliverable soluble P + Deliverable non-soluble P						*5.008* lbs P ton^{-1}
P application rate	Expressed in lbs P *unit*$^{-1}$					×	*3.5* tons ac^{-1}
Application method/timing factor	Injected **0.01**	Incorporated within 48 hours **0.05**	Incorporated 48 hours to 4 weeks **0.1**	Incorporated >4 weeks to 3 months **0.5**	All other surface applications **1.0**	×	*1.0*
Runoff fraction for natural drainage	Runoff volume (in) (as calculated in Part 2) / Rainfall amount (in) *(1.87 in)* *(51.83 in)*					×	*0.0361*
Final P loss via applied P source for naturally drained example						=	***0.63* lbs P ac^{-1}¶**
Artificially drained site							
Same as above: Deliverable source P * P application rate * application method/timing factor							*17.523* lbs P ton^{-1}
Runoff fraction for artificial drainage	Runoff volume (in) (as calculated in Part 2) / Rainfall amount (in) *(1.56 in)* *(51.83 in)*					×	*0.0300*
Final P loss via applied P source for artificially drained example						=	***0.53* lbs P ac^{-1} #**

† Both natural drainage and artificial drainage situations use the same method to calculate Part 4 with the exception of the 'runoff fraction'. This parameter, as well as the final P loss from applied P source, has been presented separately in this table for the two different drainage conditions.

‡ Deliverable Source P = [(Soluble P * Soluble P attenuation factor) + (Non-soluble P * Non-soluble P attenuation factor * buffer retention factor)].

§ Same value as used in Part 1.

¶ Yields an index value of **16**.

Yields an index value of **13**.

Results of PLAT for naturally drained example.

Loss pathway		Value
Particulate P loss	+	***0.50*** **lbs P ac**$^{-1}$
Runoff P loss	+	***0.47*** **lbs P ac**$^{-1}$
Subsurface drainage P loss	+	***0.71*** **lbs P ac**$^{-1}$
Applied P source loss	+	***0.63*** **lbs P ac**$^{-1}$
Total P loss	=	**2.31 lbs P ac**$^{-1}$ **yr**$^{-1}$ †

† Yields an index value of **58**. The potential for P loss in this example field is 'high' and further P applications should be restricted to crop removal rates.

Results of PLAT for artificially drained example.

Loss pathway		Value
Particulate P loss	+	***0.50*** **lbs P ac**$^{-1}$
Runoff P loss	+	***0.40*** **lbs P ac**$^{-1}$
Subsurface drainage P loss	+	***0.12*** **lbs P ac**$^{-1}$
Applied P source loss	+	***0.53*** **lbs P ac**$^{-1}$
Total P loss	=	**1.55 lbs P ac**$^{-1}$ **yr**$^{-1}$ †

† Yields an index value of **39**. The potential for P loss in this example field is 'medium' and further animal waste applications can be applied based on nitrogen. However, with continuous additions of animal waste or high rates of fertilizer the potential exists for this index value to increase in the future.

The concentration of P in both runoff and subsurface drainage is estimated the same way for naturally and artificially drained soils. The amount of soluble P in surface runoff is determined by measuring the STP of a surface (0-20 cm) soil sample and by assigning the given soil to one of the four P threshold groups. A unique equation has been developed for each threshold group in order to estimate the soluble P at a given STP for each threshold group.

Determining the concentration of soluble P that will be present in subsurface drainage is done in only certain specific situations. There are two conditions that must be met in order for a subsoil sample (70-80 cm) to be required. First, the STP at the soil surface must exceed the P threshold (see Table 3) for the particular soil. If this does not occur, it is assumed that the surface soil is still able to retain a significant amount of P and very little, if any, soluble P would reach the subsoil even given a large volume of subsurface drainage. Second, the soil must be designated as being of a type in which soluble P could potentially move downward through the soil profile. For certain types of soil with highly developed Bt horizons, it is assumed that even if the surface is P saturated, soluble P will not move below this soil depth due to the relatively high sorption capacity of the fine-textured Bt horizon. In this case a subsurface sample is not required. If, however, the soil surface is P saturated, as indicated by the P threshold, and the soil type is not a well-drained soil with a stable Bt horizon, then a subsoil sample is taken from the 70-80 cm depth and STP determined. The subsoil soluble P concentration is determined by assuming a subsoil texture of loam and treating it as if it were a surface soil, i.e. estimating soluble P by using the relationship between M3P and soluble P. It is appreciated that all subsurface soils will not behave like a loam, releasing 1 mg L^{-1} soluble P at a M3P level of 200 mg kg^{-1}, but there is currently insufficient data to fully understand the dynamics of soluble P movement through subsurface soils. This is an area that is presently being studied on some of the more problematic coarse-textured or poorly drained soils in North Carolina and will undoubtedly allow the tool to be updated in the future to improve prediction of P loss via this pathway. Additionally, if the subsoil M3P is = 50 mg kg^{-1} then the site is automatically given a rating of ‘high’ as it is assumed that the subsoil has been saturated to a level of P that poses a serious problem with respect to the loss of soluble P laterally to drainage ditches or streams. It is possible for the index rating of such a site to increase above a ‘high’ if factors in the any of the other loss pathways contribute to an overall higher rating.

While the concentrations of soluble P in Parts 2 and 3 are calculated identically for naturally and artificially drained sites, estimating the transport factors of these two loss pathways, runoff and subsurface drainage volume, is done using two different methods depending on drainage conditions, as described previously. Both techniques rely on long-term county weather data and the use of hydrologic model simulations. On well-drained soils requiring no artificial drainage, surface runoff is determined by using a modification of the SCS curve number (USDA-SCS, 1985) and long-term rainfall data from each county. Exponential equations relating average annual runoff to adjusted curve number were developed for each county. To estimate subsurface drainage volume on naturally drained soils, a mass balance approach is used whereby average subsurface drainage is considered to be the volume of water after precipitation, runoff (as calculated above) and evapotranspiration are accounted for. Evapo-transpiration was estimated for either of two situations, pasture or row crop, using simulations with the GLEAMS model (Leonard et al., 1987) and long-term weather data.

For soils that are poorly drained and require artificial drainage for crop production, the hydrologic model DRAINMOD, a water-balance model that partitions rainfall into the various hydrologic components of shallow water-table soils (Skaggs, 1976, 1978), was employed to predict the amount of surface runoff and subsurface drainage. Model simulations were performed using long-term rainfall data for various locations, soil types and two cover conditions, pasture or row crop. Equations were developed to estimate amounts of runoff and subsurface drainage for various drainage intensities. Drainage intensity is a factor that incorporates drain depth, drain spacing and soil transmissivity. An additional factor involved in estimating the volume of subsurface drainage is a transmissivity factor, which relates the transmissivity of the proportion of the soil profile that P-enriched water would theoretically move through, assumed to be the top 80 cm. More detailed information on the development of the algorithms used in this method can be found in Skaggs et al., (2004).

Part 4 of the index involves the phosphorus, both soluble and non-soluble, that will be lost directly from applied animal waste. In order to account for this, PLAT attempts to partition the total P of a given animal waste into liquid and solid fractions. Because there will be some level of attenuation for both soluble and non-soluble P moving across the field, separate P attenuation factors were developed for the different types of animal waste. Three categories of waste, each with similar properties of solubility, were identified: fresh manure/litter/scraped, sludge/slurry, and liquid/inorganic fertilizer. A P source such as lagoon liquid will have relatively little total P loss from the field due to its high solubility and tendency to infiltrate the soil rapidly, whereas a source such as dry poultry litter will be more likely to remain on the soil surface and provide a longer term source of P as it dissolves over time. Particulate matter of source types with a high proportion of solids may also be subjected to transport off field at varying rates by overland flow. However, insufficient data exists at this time to quantitatively account for this phenomenon, although its existence is recognized. Therefore, a single non-soluble P attenuation factor was given to all source types until further research is performed. An estimation of rainfall that is moving over the soil surface as runoff (‘Runoff fraction’) is made by using county rainfall averages and the runoff volume that is calculated in Part 2. The proportion of rainfall that does not infiltrate into the soil and becomes runoff is an important factor affecting the amount of applied source P lost off-site. Credits, or reductions in the overall applied source P leaving the field, are given for application methods that incorporate applied animal waste into the soil, leaving it less susceptible to overland loss. The existence of a vegetative buffer may also reduce particulate matter and is accounted for.

MATERIALS AND METHODS

To predict the percentage of NC farms that will be affected by the implementation of PLAT, input parameters from randomly selected agricultural fields in all one hundred counties of North Carolina were collected. North Carolina is characterized by three distinct physiographic regions (Mountains, Piedmont and Coastal Plain), each with differing soils and topography. Table 5 shows the percentage of land area and agricultural area in the different physiographic regions of the state. Additionally, specific animal types and management systems tend to be concentrated in certain areas of the state (Figures 3 and 4).

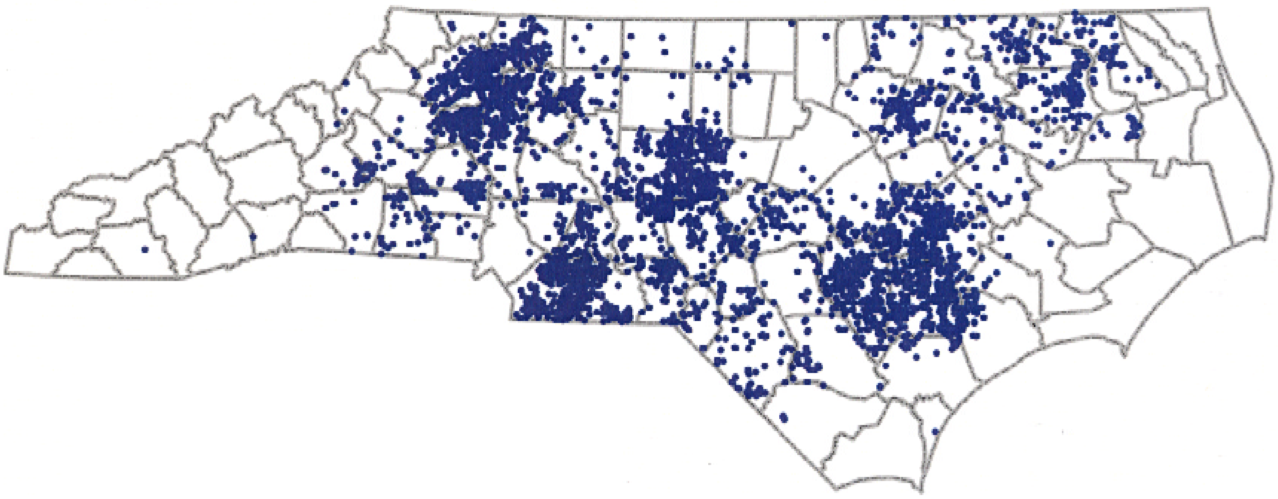

Figure 3. Poultry operations throughout the state of NC. Data obtained from North Carolina Department of Agriculture, Veterinary Division, 2004. Includes all poultry operations, regardless of waste management.

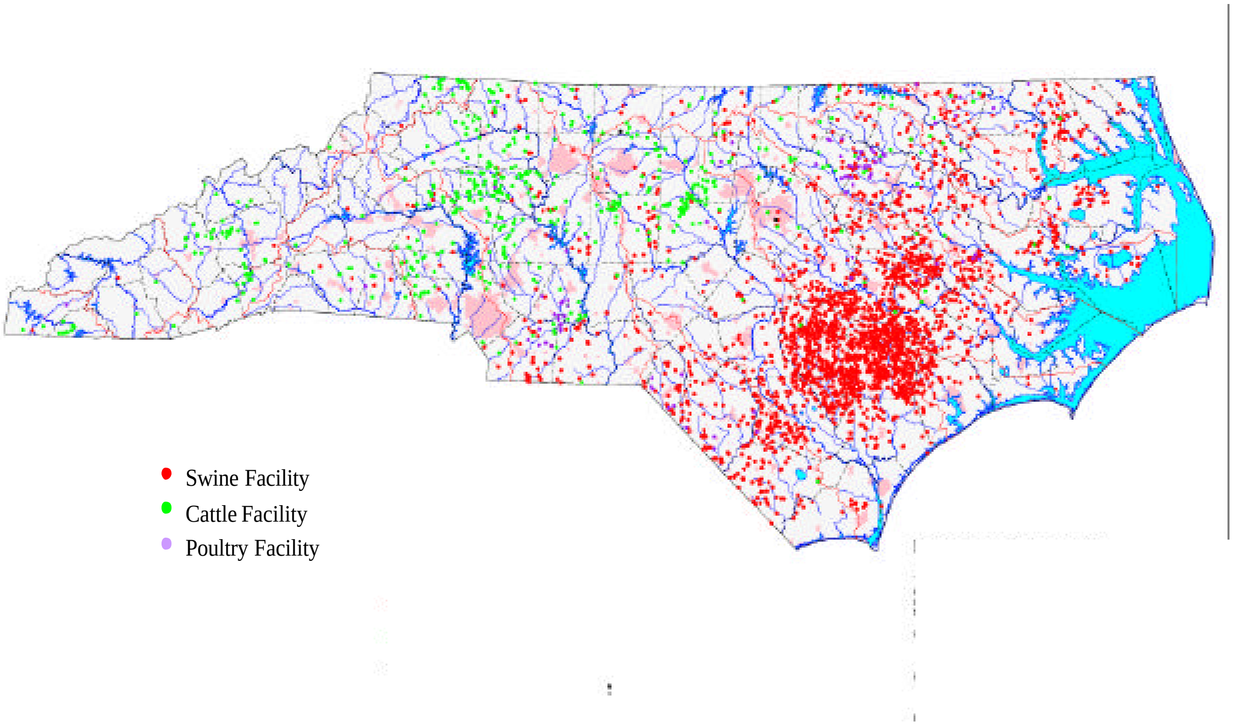

Figure 4. Confined animal feedlots registered with NC Department of Environment and Natural Resources (DENR) as required by NC .0200 Rules for Waste Not Discharged to Surface Waters. Animal operations with fewer than threshold populations (see Table 1) and poultry operations with dry litter waste systems (see Figure 4) are not included.

Table 5. Land area of North Carolina's physiographic regions.

REGION	% of state's land area	% of state's agricultural acreage
Coastal Plain	45	52
Piedmont	39	34
Mountain	16	14

To insure that samples were obtained without bias toward high or low P-saturated sites, the following sampling protocol was implemented: the number of fields sampled from an individual county was weighted to represent the agricultural acreage, both of row cropland and hay/pasture land, in that particular county. The aim was to be able to use our sampling results to predict, with the highest statistical probability possible, the results of running PLAT on every farm in the state. From these data, the number of producers that would be required to implement P-based management under the new regulations and thereby presumably suffer economic hardship could be ascertained.

The maximum number of total fields sampled from a given county was 36 and the minimum number of fields sampled was five, with the mean number of fields sampled being 16 fields. A total of 1379 sites in the state were sampled. A list of farms in each county was obtained from county extension personnel or from the Farm Service Agency from which twice the number of farms needed to fulfill the sampling requirement in each county were randomly selected. On a chosen farm, only one field was used unless the farm included both crop and pasture/hay fields; in this case, one field of each type was sampled. If the selection produced a landowner who was not farming or who was unwilling to participate, the next farm on the list was used. Once a site was chosen, the individual field was selected by obtaining the field on the farm that was closest to the geographic center of the county according to digital orthoquads.

Data collected for each field included: RUSLE inputs, (including vegetative cover type, cover condition, slope variables, and soil mapping unit), information on the presence of drainage tiles or ditches, as well as their depth and distance apart, water control structures, farm ponds or buffers including buffer width. Length of depositional area (receiving slope) from the RUSLE slope, tillage, types and amounts of phosphorus application, as well as timing and method of application, were also noted. In addition, a composite soil sample was collected at the 0-20 cm (0-8 in) soil depth and analyzed for Mehlich-3 P at the North Carolina Department of Agriculture and Consumer Services Soil Testing Lab. In cases where a subsoil sample was required, a composite sample was taken at approximately the 70-80 cm (28-32 in) soil depth. Soil mapping units were used to obtain information on clay percentage, curve number, and transmissivity of the sampled soil, as well as to determine classification in one of the four P threshold groups. Once collected, data for each site were entered into PLAT.

RESULTS AND DISCUSSION

Figure 5 shows the mean PLAT index ratings by county for the 1379 sites evaluated. Although most counties in North Carolina, on average, fall into the PLAT index categories that do not require modifications in nutrient management plans (‘low’ and ‘medium’), Figure 5 suggests that a few problem areas exist with respect to P management. These areas correspond to counties in which average soil test P levels are elevated (Figure 1), which have large P surpluses (Figure 2) and in which confined animal feeding operations are concentrated (Figures 3 and 4). The animal waste produced by these operations is land applied based on the nitrogen needs of the receiving crop and has resulted in a build up of P in the soil creating a greater risk of P loss to the environment. A few counties in the Lower Coastal Plain, in particular Duplin and Sampson counties, support one of the nation’s largest swine production industries.

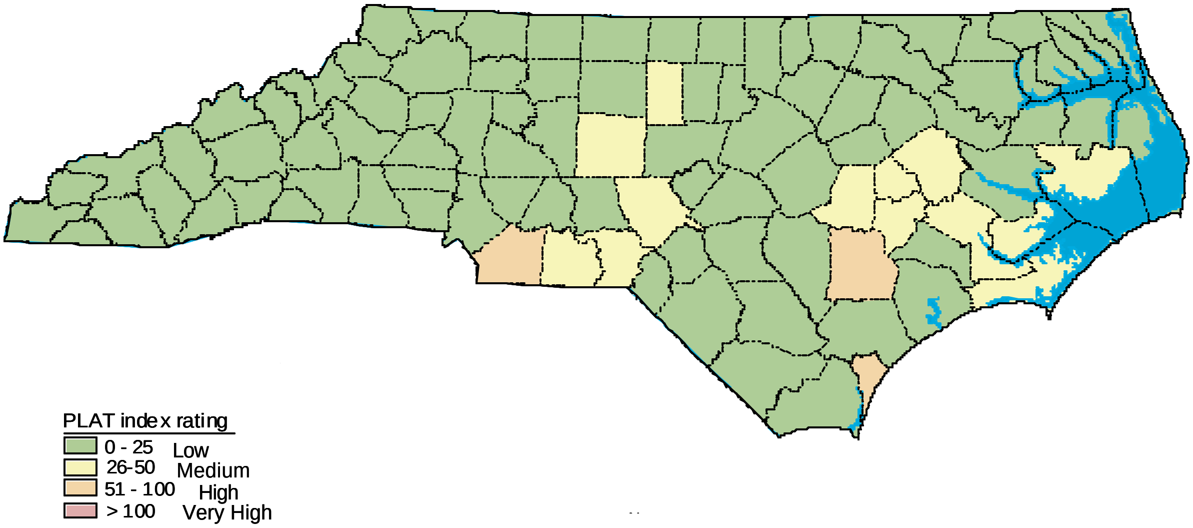

Figure 5. Mean PLAT index county ratings.

Sample sites in Duplin County, for example, had a mean STP of 232 mg kg^{-1} and PLAT results predicted that 26% of sites posed a ‘high’ risk and 23% a ‘very high’ risk. This result implies that if PLAT were applied to all fields in the county, roughly half of the fields receiving animal waste would be required to modify current management practices in order to address the potential for P loss, a much higher percentage than the state average.

Statewide, the application of PLAT resulted in 81% of the 1379 sample sites being categorized as ‘low’ with respect to potential P loss and 11% as ‘medium’ (Table 6). Sites rating ‘high’ and ‘very high’, and thereby requiring P-based animal waste applications, comprised 6% and 2% of the fields respectively. These results suggest that, potentially, 8% of farms in North Carolina will no longer be able to use N-based management plans. Considering only those sample sites on which animal waste was applied, PLAT predicted the percentage of producers expected to be affected to be substantially higher (Table 6). An average of 24% of the sites receiving animal waste were in the 'high' or 'very high' PLAT rating categories, whereas only 4% of sites applying inorganic fertilizer as a P source resulted in either of these ratings.

Table 6. Number and percentage of fields in PLAT rating categories according to P source type.

		PLAT Risk Category			
P Source Type	**n**	**Low**	**Medium**	**High**	**Very High**
All samples	1379	1123 (81%)	155 (11%)	76 (6%)	25 (2%)
Dairy	20	12 (60%)	4 (20%)	3 (15%)	1 (5%)
Poultry	148	89 (60%)	24 (16%)	26 (18%)	9 (6%)
Swine	43	22 (51%)	10 (23%)	7 (16%)	4 (9%)
Inorganic	1168†	1000 (86%)	117 (10%)	40 (3%)	11 (1%)

†Sites not using any P amendment (n=302) were included under the 'Inorganic' source type in this table.

The average amount of total P loss for the 1379 sites as estimated by PLAT was 0.64 lbs P ac^{-1} (Table 7), which ranks as 'low' when converted to an index value (multiplied by '25'). Again, it is important to note that these values do not accurately represent exact amounts of P loss in the field but rather provide a basis for making comparisons between different PLAT loss pathways, soils, regions or managements. From the four separate PLAT components an average of 0.10 lbs P ac^{-1} was predicted to be lost due to erosion, 0.36 lbs P ac^{-1} due to surface runoff, 0.05 lbs P ac^{-1} due to subsurface drainage and 0.16 lbs P ac^{-1} due to applied P source (Table 7). The mean STP for the 1379 sites (114 mg kg^{-1}) exceeded the 53 mg kg^{-1} recommended for optimum crop production by the state soil-testing lab.

Table 7. Summary statistics for PLAT index parameters from the 1379 sites sampled.

PLAT Parameter	Mean	Median	S.D.	Range
Part 1. Phosphorus loss via erosion				
Erosion (ton ac^{-1})	1.6	1.0	2.4	0 – 23.0
Surface STP (mg kg^{-1})	114	80	121	0 – 2057
Total P (lb P ton^{-1})	0.417	0.272	0.549	0 – 5.802
Reduction factor	0.52	0.46	0.35	0.01 – 1.00
Erosion P loss (lbs P ac^{-1})	0.10	0.01	0.33	0 – 5.62
Part 2. Phosphorus loss via surface runoff				
Runoff concentration (mg L^{-1})	0.54	0.29	0.75	0 – 10.28
Runoff volume (ac-in)	2.77	1.89	2.42	0.05 – 15.28
Surface runoff P loss (lbs P ac^{-1})	0.36	0.13	0.79	0 – 15.11
Estimated surface runoff volume and P loss via surface runoff from naturally drained sites†				
Runoff volume (ac-in)	2.45	1.80	2.00	0.05 – 13.21
Surface runoff P loss (lbs P ac^{-1})	0.31	0.08	0.70	0 – 15.11
Estimated surface runoff volume and P loss via surface runoff from artificially drained sites‡				
Runoff volume (ac-in)	3.82	2.86	3.24	0.06 – 15.28
Surface runoff P loss (lbs P ac^{-1})	0.53	0.28	0.68	0 – 4.17
Part 3. Phosphorus loss via subsurface drainage				
Subsurface STP (mg kg^{-1})§	26	2	56	0 – 296
Subsurface drainage concentration (mg L^{-1})	0.13	0.01	0.29	0 – 1.48
Subsurface drainage volume (ac-in)	12.58	11.30	5.21	3.10 – 29.30
Subsurface drainage P loss (lbs P ac^{-1})	0.05	0.00	0.35	0 – 5.00
Estimated subsurface drainage volume and P loss via subsurface drainage from naturally drained sites†				
Subsurface drainage volume (ac-in)	11.53	10.60	4.85	3.40 – 29.30
Subsurface drainage P loss (lbs P ac^{-1})	0.06	0.00	0.39	0 – 5.00
Estimated subsurface drainage volume and P loss via subsurface drainage from artificially drained sites‡				
Subsurface drainage volume (ac-in)	16.01	16.90	4.87	3.10 – 25.2
Subsurface drainage P loss (lbs P ac^{-1})	0.02	0.00	0.14	0 – 1.74

Part 4. Phosphorus loss via applied P source				
Available P (lbs P ac^{-1})¶	3.61	1.03	6.22	0.10 – 36.80
Application method/timing factor	0.99	1.00	0.09	0.05 – 1.00
Runoff fraction	0.0580	0.0420	0.0469	0.0009 – 0.2806
Applied source P loss (lbs P ac^{-1})	0.16	0.06	0.38	0 – 4.63
Total P loss (lbs P ac^{-1})	0.64	0.29	1.08	0 – 18.46

† Sample sites with natural drainage, n=1056.

‡ Sample sites with artificial drainage, n=323.

§ Subsurface (70-80 cm soil depth) sample required only if surface STP is above the P threshold value for the particular threshold group that the soil in question belongs to and the soil type is a designated ‘susceptible’ soil; 191 sites out of 1379 fit these criteria and required a subsurface sample.

¶ Available P = {application rate (*unit* ac^{-1})*[deliverable soluble P (lbs P *unit*$^{-1}$) + deliverable non-soluble P (lbs P *unit*$^{-1}$)]}; deliverable soluble P and non-soluble P are specific to waste-type. Both animal waste and inorganic fertilizer are included. Inorganic fertilizer is assumed to be 100% soluble. A total of 1077 sites recorded the use of some form of P amendment.

Runoff P concentration in PLAT is a function of two factors as described earlier: Soil test P and P threshold soil group, with the amount of soluble P released for a given STP depending on which threshold group a soil belongs to. This method of relating the STP of a particular site to runoff concentration is unique among other states’ P indexes as it accounts for differences in a soil’s capacity to hold P. It is appreciated that our method of grouping all state soils into only four threshold groups is an oversimplification of the P retention characteristics of all soils in the state, but it is thought to be a better approach than assuming that all soils behave identically with regards to P dissolution. The possibility for in-field attenuation of soluble P in runoff water is currently not addressed in PLAT.

Surface Runoff P Loss

Predicted phosphorus loss from surface runoff was the largest contributor, on average, to a site’s P loss risk, despite a relatively low mean runoff volume (2.77 ac-in). The maximum P loss from this pathway, 15 lbs P ac^{-1}, was almost three times the maximum amount of any of the other three PLAT P loss pathways. No BMP’s exist for surface runoff other than reducing the amount of runoff by improving soil cover conditions, thereby maximizing infiltration, or reducing the STP level. Reducing STP by discontinuing P applications and optimizing crop removal of residual soil P has been shown to require many years

(McCollum, 1991). The mean runoff P concentration as estimated by PLAT was more than 0.5 mg L^{-1}, a concentration at which eutrophication has been shown to occur (Sims et al., 1998).

Within Part 2 and Part 3 of PLAT, estimates of surface runoff and subsurface drainage volumes, as well as resultant P losses, were separated into sites having either natural drainage or those that are artificially drained. Sites on which artificial drainage was used experienced greater volumes of both surface runoff and subsurface drainage (Table 7). This result was due to the different methods used to estimate volumes of water moving over and through the soil. Both methods involved simulations of hydrologic models using local long-term rainfall data as described previously. One noteworthy aspect is that the water model DRAINMOD has been field validated at length for soils and conditions in eastern North Carolina where the majority of improved drainage occurs. The method used on naturally draining sites, however, encompasses all regions of the state, requiring a much more general approach. Which method is more correct in estimating quantities of the surface runoff and subsurface drainage is not known.

Artificially drained sites were estimated to have a greater volume of surface runoff than naturally drained sites, which given that the concentration of soluble P in runoff water was determined identically for both drainage situations, resulted in a greater predicted P loss via surface runoff (0.53 lbs P ac^{-1} vs. 0.31 lbs P ac^{-1} for artificial and natural drainage, respectively). Implementation of artificial drainage to a field increases the amount of subsurface drainage and would therefore be expected to concomitantly reduce surface runoff. When compared to naturally drained sites, however, this was not the case for our sample sites as described above.

Subsurface Drainage P Loss

A greater calculated subsurface drainage volume in the artificially drained sites did not result in a significantly higher estimated loss of P via this pathway when compared to sites having natural drainage (0.06 lbs P ac^{-1} vs. 0.02 lbs P ac^{-1} for artificially and naturally drained sites respectively). This result was due to the inclusion of a transmissivity factor (T_{30}/T_p) which was used in calculating P loss in Part 3; the transmissivity factor modifies the amount of soluble P moving through the soil profile. This factor was introduced because it is assumed that once soluble P moves to a soil depth of 30 in, it will be lost through subsurface drains rather than continue on downward through the soil profile. No such factor exists for the computation of P loss via subsurface drainage on naturally drained sites. Thus, the predicted P loss via subsurface drainage on artificially drained sites was reduced when compared to naturally drained sites, despite a greater drainage volume and equal drainage P concentrations.

Applied Source P Loss

Partitioning the mean P loss from applied source into the different types of P amendments used revealed that PLAT predicted P loss from inorganic fertilizer to be only a minor contributor to overall P loss risk as compared to animal waste applications (Table 8). The PLAT model estimated that sample sites receiving animal waste applications during the year that site information was collected would potentially lose 0.55 lbs P ac^{-1} to runoff from the applied P source, whereas sites amended with inorganic fertilizer would potentially lose 0.07 lbs P ac^{-1}. Comparing sites receiving different animal waste types revealed that loss assessments on sites receiving poultry waste had a higher risk of P loss from this component (0.62 lbs P ac^{-1}) than sites receiving either dairy or swine waste. Sites receiving swine waste had only half the calculated P loss of sites receiving the other animal waste types (0.29 lbs P ac^{-1}).

Table 8. Summary statistics for calculated P loss from applied source (Part 4) for different P amendments†.

P Source	n	Mean	Median	S.D.	Range
			lbs P ac^{-1}		
Inorganic fertilizer	866	0.07	0.04	0.08	0 – 0.79
Animal waste	211	0.55	0.28	0.72	0.01 – 4.63
• Dairy	20	0.58	0.27	0.67	0.01 – 2.02
• Poultry	148	0.62	0.32	0.79	0.01 – 4.63
• Swine	43	0.29	0.16	0.39	0.01 – 2.06

† P loss from applied P source = Available P (lbs P ac^{-1}) *Application method/timing factor * Runoff fraction; A total of 1077 sites recorded the use of some form of P amendment.

Overall P Losses

Table 9 shows the distribution of calculated PLAT parameters from the 1379 sites into the four risk categories. With a few exceptions, the mean values of most parameters increased as risk category increased from ‘low’ to ‘very high’. The rate of erosion did not follow this trend, indicating that erosion alone was not enough to create ‘high’ risk conditions, but that the eroded soil must also have a relatively high total P content. The presence of BMP’s that reduce the loss of eroded soil may also have affected this result. Mean subsurface drainage volume decreased with an increase in P loss risk category. This result was due to the mass balance approach that is used in PLAT for determining volumes of both surface runoff and subsurface drainage: as the amount of one increased the other must concomitantly decrease. The level of surface STP was an important factor affecting P loss as it is used to calculate the total P loss factor employed in predicting erosion losses of P. It is also used to estimate runoff concentration and, indirectly, drainage concentration, as P concentration with soil depth is related to surface buildup of P. The final

Table 9. Mean values of PLAT index parameters from the 1379 sites, categorized by risk potential.

PLAT Parameter	PLAT Risk Category			
	Low	**Medium**	**High**	**Very high**
	Part 1. Phosphorus loss via erosion			
Erosion (tons ac^{-1})	1.3	2.4	2.7	1.8
Surface STP (mg kg^{-1})	88	193	251	362
Total P factor (lbs P ton^{-1})	0.343	0.624	0.894	1.003
Reduction factor	0.50	0.59	0.56	0.71
Erosion P loss (lbs P ac^{-1})	0.05	0.22	0.45	0.72
Part 2. Phosphorus loss via surface runoff				
Runoff concentration (mg L^{-1})	0.36	1.10	1.42	2.46
Runoff volume (ac-in)	2.32	4.54	4.83	5.57
Surface runoff P loss (lbs P ac^{-1})	0.17	0.89	1.29	2.87
Part 3. Phosphorus loss via subsurface drainage				
Subsurface STP (mg kg^{-1})†	5	10	51	133
Subsurface drainage concentration (mg L^{-1})	0.04	0.05	0.25	0.62
Subsurface drainage volume (ac-in)	12.90	11.51	11.09	9.65
Subsurface drainage P loss (lbs P ac^{-1})	0.00	0.04	0.32	1.28
Part 4. Phosphorus loss via applied P source				
Available P (lbs P ac^{-1})‡	2.83	4.76	8.39	10.97
Runoff fraction	0.0470	0.0906	0.0961	0.1117
Applied source P loss (lbs P ac^{-1})	0.06	0.23	0.55	1.19
Total P loss (lbs P ac^{-1})	0.28	1.39	2.61	6.07

† Subsurface (70-80 cm soil depth) sample required only if surface STP is above the P threshold value for the particular threshold group that the soil in question belongs to and the soil type is a designated 'susceptible' soil; 191 out of 1379 sites fit these criteria and required a subsurface sample.

‡ Available P = {application rate (*unit* ac^{-1})*[deliverable soluble P (lbs P *unit*$^{-1}$) + deliverable non-soluble P (lbs P *unit*$^{-1}$)]}; deliverable soluble P and non-soluble P are specific to waste-type. Both animal waste and inorganic fertilizer are included. Inorganic fertilizer is assumed to be 100% soluble. A total of 1077 sites recorded the use of some form of P amendment.

estimated P loss for each of PLAT's four loss pathways was lowest in magnitude for sites in the 'low' risk category and highest for sites in the 'very high' category.

The distribution of sites having various levels of STP into the PLAT risk categories is shown in Table 10. In the range of STP values for which recommendations for additions of fertilizer or manure P applications would be made, almost all of the sites received a PLAT rating of 'low'. As STP increased, the percentage of fields that would fall into risk categories requiring P-based management expectedly increased. On sites with STP levels of 200 mg kg^{-1} or greater, 31% would be considered to have a high or very high potential to contribute P to surface waters. Still, 40% of sites with this level of STP are considered 'low' risk, indicating the importance of transport factors in PLAT. Despite a high source of P from either soil or manure P, if the mechanisms to transport it as either soluble or particulate P do not exist, it theoretically can not present a threat to water quality.

Table 10. Percentage of fields in PLAT rating categories according to STP level.

		PLAT Risk Category			
STP (mg kg^{-1})	**N**	**Low**	**Medium**	**High**	**Very High**
0 – 53	481	99%	1%	<1%	--
53 – 100	330	93%	5%	2%	--
100 – 200	350	73%	19%	7%	1%
>200	218	40%	29%	21%	10%

Comparison by P Source

Comparing sites receiving animal waste versus inorganic fertilizer revealed that the parameters of the PLAT index representing source factors (surface and subsurface STP and available P from applied amendments) were significantly higher for fields receiving animal waste (Table 11). A greater buildup of soil P in both surface and subsurface soil and greater concentrations of soluble P were evident on fields utilizing animal wastes. Sites utilizing inorganic fertilizer had higher mean values for the parameters representing potential transport of P off-field (erosion, runoff and subsurface drainage volume and runoff fraction) than did sites using animal waste. Overall P loss totals predicted by PLAT were 1.41 lbs P ac^{-1} versus 0.56 lbs P ac^{-1} for animal waste and inorganic fertilizer application respectively.

Table 11. Mean values of PLAT index parameters between inorganic and animal waste P sources.

PLAT Parameter	Inorganic fertilizer	Animal Waste
Erosion rate (tons ac^{-1})	1.7	1.2
Surface STP (mg kg^{-1})	97	220
Total P (lbs P ton^{-1})	0.333	0.925
Reduction factor	0.55	0.45
Erosion P loss (lbs P ac^{-1})	0.10	0.13
Runoff concentration (mg L^{-1})	0.48	1.06
Runoff volume (ac-in)	3.03	2.20
Surface runoff P loss (lbs P ac^{-1})	0.35	0.56
Subsurface STP (mg kg^{-1})†	21	41
Subsurface drainage concentration (mg/L)	0.11	0.21
Subsurface drainage volume (ac-in)	12.65	11.57
Subsurface drainage P loss (lbs P/ac)	0.03	0.17
Available P from applied source (lbs P/ac)‡	1.11	13.61
Application method/timing factor	1.00	0.95
Runoff fraction	0.0612	0.0490
Applied source P loss (lbs P/ac)	0.07	0.55
Total P loss (lbs P/ac)	0.56	1.41

† Subsurface (70-80 cm soil depth) sample required only if surface STP is above the P threshold value for the particular threshold group that the soil in question belongs to and the soil type is a designated 'susceptible' soil; 191 sites out of 1379 fit these criteria and required a subsurface sample.

‡ Available P = {application rate (*unit* ac^{-1})*[deliverable soluble P (lbs P *unit*$^{-1}$) + deliverable non-soluble P (lbs P *unit*$^{-1}$)]}; deliverable soluble P and non-soluble P are specific to waste-type. Both animal waste and inorganic fertilizer are included. Inorganic fertilizer is assumed to be 100% soluble. A total of 1077 sites recorded the use of some form of P amendment.

When sites were separated by the type of animal waste used, farms applying poultry waste were predicted to lose P predominantly through surface runoff and applied source (Table 12). The majority of P lost from sites receiving swine waste was lost as soluble P, through both surface and subsurface pathways. Swine waste is applied as lagoon effluent, which has less total P than either poultry or dairy waste but much more of it is in a soluble form. Sites receiving dairy waste had the greatest P loss due to erosion, most likely because many of these sites are on uplands prone to erosion and cropped to row crops rather than pasture or hay. Loss of P through surface runoff on dairy sites was low, presumably for the same reason; soils on these sites had a relatively high percentage of clay and so would be able to hold greater amounts of P before releasing it to runoff waters. In fact, these soils had a lower average STP (117 mg kg^{-1}) than soils on which either poultry (246 mg kg^{-1}) or swine waste (179 mg kg^{-1}) was applied.

Table 12. Mean calculated P loss values calculated for sites applying different P amendments.

	PLAT P Loss Pathway				
P Source	**Erosion**	**Surface runoff**	**Subsurface drainage**	**Applied source†**	**Total**
			lbs P ac^{-1}		
Inorganic fertilizer	0.10	0.35	0.03	0.07	0.55
Dairy	0.43	0.18	0.00	0.58	1.20
Poultry	0.11	0.61	0.12	0.62	1.45
Swine	0.05	0.59	0.43	0.29	1.36

†Included only those sample sites that recorded some form of P amendment (n=1077).

Comparison by Physiographic Region

Sample sites were additionally segregated by physiographic region as shown in Table 13. Sites located in the Piedmont region of North Carolina experienced, on average, greater estimated losses of P through erosion than other regions (Table 13). The Piedmont region consists of residual upland soils and was dominated by sites with clay soil types, which are more prone to erosive losses. Fields in the Mountain region, which would also be expected to have relatively higher erosion rates, had little P loss overall. This may be in part due to the prevalent cropping system of this region (94% of Mountain sites were in pasture).

Surface losses of soluble P decreased as elevation above sea level increased, (Coastal Plain>Piedmont>Mountain). Both the volume and the P concentration of surface runoff decreased in this pattern (3.74, 2.22 and 1.75 ac-in for runoff volume; and 0.84, 0.39 and 0.21 mg/L for concentration in Coastal Plain, Piedmont and Mountain sites respectively). Sites from the Mountain region had relatively low surface runoff volumes, which may be related to the dominance of pasture-based cropping systems as mentioned above. The greater runoff volume in Coastal Plain sites was most likely caused by the prevalence of artificially drained sites in this region. Ninety-four percent of artificially drained sites in the study occurred in the Coastal Plain, while 50% of the sites in this region were artificially drained. As discussed earlier, the method of estimating runoff and subsurface drainage volume in sites having artificial

Table 13. Mean calculated P loss values for sites from each of NC's three physiographic regions.

Physiographic Region	**N**	**Erosion**	**Surface runoff**	**Subsurface drainage**	**Applied source†**	**Total**
				lbs P ac^{-1}		
Mountains	341	0.06	0.09	0.00	0.12	0.23
Piedmont	433	0.18	0.24	0.01	0.23	0.62
Coastal Plain	605	0.07	0.60	0.10	0.14	0.88

(**PLAT P Loss Pathway** spans Erosion through Total.)

† Included only those sample sites that recorded some form of P amendment; (n=1077).

drainage tended to predict greater volumes of water when compared to the estimation method used on naturally drained sites. Soil test P also followed this trend, accounting for the greater runoff concentration seen as elevation above sea level decreased. As discussed previously, the Coastal Plain has a high density of animal units that has led to a build up of soil P. This region had a greater proportion of sandy (72% of sand samples), and loamy (81% of loam samples) sites, while 99% of clay sample sites occurred in the Piedmont and Mountains. Coarser textured soils would be expected to release P into solution at lower STP levels than finer-textured soils (Cox, 1994) and this in combination with the higher overall STP more than likely contributed to the greater mean runoff P concentrations in the Coastal Plain region.

Subsurface losses of P were negligible in the Mountain and Piedmont regions. In the Mountains, this result was due to the improbability of P leaching downward in the soils of this region. The majority of soils in the Mountain region are more clayey and due to the lack of intensive animal agriculture, the STP has not accumulated to a level that requires a subsurface sample to be taken. Sites in this region, which occur on floodplain landscapes, could potentially have some loss of soluble P via subsurface drainage, as these soils tend to be coarser-textured and would therefore have a lower P threshold level, thus requiring a subsample at a lower STP. Sample sites in the Piedmont had a moderate subsurface P concentration but combined with a low drainage volume had a low overall loss via subsurface drainage. In general, the Coastal Plain region has higher water tables and more poorly drained soils and so would provide an ideal environment for subsurface movement of P to occur. Approximately 95% of the swine farms in the sample set were located in the Coastal Plain, perhaps further contributing to the higher loss of soluble P via subsurface transport in this area of the state due to the soluble nature of this type of animal waste. For example, sample sites in Duplin County over half of which were in the 'high' or 'very high' risk category, had an average of 177 mg kg^{-1} STP and a predicted P concentration of 0.9 mg L^{-1}.

The Piedmont region had the greatest loss from applied P source as compared to the other two physiographic regions. Eighty percent of Piedmont sample sites receiving animal waste had poultry waste applied to them. Additionally, the majority of sites receiving dairy waste were located in the Piedmont. These two animal waste types have a higher overall P content than swine waste.

Overall P loss was controlled by surface runoff losses, with the region with the greatest runoff loss having the largest predicted P loss, again indicating the importance of this loss pathway in the PLAT index.

Comparison by Threshold Soil Group

Table 14 shows PLAT results for each P loss pathway differentiated by threshold soil group (as defined in PLAT). Overall particulate P loss was greatest in clay soils, which parallels the higher particulate P loss seen in the Piedmont region (Table 13). The rate of erosion was highest in clay soils, which again, is related to landscape position. However, these sites also have the lowest average reduction factor (0.44) decreasing the amount of particulate P moving off-site. Still, with their higher clay content, these sites had the highest mean value for Total P factor and P loss from this pathway, despite the lower STP. Since the Total P factor accounts for the relationship between clay content and STP status (Cox, 1994), this finding suggests that clay content had a greater effect on this factor than does the level of STP.

With the exception of organic soils, the calculated loss of P through surface runoff increased as soils became coarser-textured as, presumably, the P sorption capacity decreased. Soil test P was highest on sites having sandy-textured soils. And again, despite the variable runoff volume, PLAT values for the soluble P loss pathway followed trends in the source factor (STP) rather than the transport factor (runoff volume). This result was most likely caused by a history of animal waste applications as the majority of sandy sites occurred in the high swine density region of the Coastal Plain. As such, these sites arguably represent the worst problem in the state in terms of P loss to the environment.

Sites belonging to the sand threshold group had the highest subsurface drainage P loss, as expected. Disregarding one anomalous clayey site that had a disproportionately high subsurface STP, subsurface STP

Table 14. Mean values for PLAT index parameters by P threshold soil group†.

	P Threshold Soil Group			
PLAT Parameter	Clay	Loam	Sand	Organic
Erosion (tons ac^{-1})	1.7	1.4	1.4	0.0
Surface STP (mg kg^{-1})	90	126	170	101
Total P (lb ton^{-1})	0.473	0.363	0.405	0.257
Reduction factor	0.44	0.60	0.53	0.72
Erosion P loss (lbs P ac^{-1})	0.14	0.07	0.07	0.00
Runoff concentration. (mg L^{-1})	0.18	0.63	1.70	2.02
Runoff volume (ac-in)	1.83	3.95	1.86	2.95
Surface runoff P loss (lbs P ac^{-1})	0.08	0.56	0.72	1.18
Subsurface STP (mg kg^{-1})‡ (n=191)	43§	20	33	11
Subsurface drainage concentration (mg L^{-1})	0.21	0.10	0.17	0.06
Subsurface drainage volume (ac-in)	11.48	13.48	13.14	18.07
Subsurface drainage P loss (lbs P ac^{-1})	0.00	0.03	0.34	0.05
Available P (lbs P ac^{-1})¶	4.24	2.60	5.44	1.93
Application method/timing factor	1.00	0.99	0.98	1.00
Runoff fraction	0.0379	0.0786	0.0382	0.0563
Applied source P loss (lbs P ac^{-1})¶	0.10	0.14	0.18	0.09
Total P loss (lbs P ac^{-1})	0.32	0.80	1.31	1.32

† P threshold soil groups have been defined in the PLAT description section above and are referred to by their generalized soil texture names.

‡ Subsurface (70-80 cm soil depth) sample required only if surface STP is above the P threshold value for the particular threshold group that the soil in question belongs to and the soil type is a designated 'susceptible' soil; 191 out of 1379 sites met this criteria and required a subsurface sample.

§ Value skewed by disproportionately high subsurface STP at one site (254 mg kg^{-1}); median value in this soil category = 0.14 mg kg^{-1}.

¶ Available P = {application rate (*unit* ac^{-1}) * [deliverable soluble P (lbs P *unit*$^{-1}$) + deliverable non-soluble P (lbs P *unit*$^{-1}$)]}; deliverable soluble P and non-soluble P are specific to waste-type. Both animal waste and inorganic fertilizer are included. Inorganic fertilizer is assumed to be 100% soluble. A total of 1077 sites recorded the use of some form of P amendment.

levels followed the same trend between threshold groups as surface STP values. Coarser-textured soils have a lower P sorption capacity and generally occur on flat landscapes with shallow water tables that are more prone to leaching, which allows soluble P to move downward through the soil profile.

The organic soils had the greatest predicted loss of soluble P through surface runoff due to the high runoff P concentration and the moderately high runoff volume. Organic soils are treated very conservatively in PLAT, having a P threshold value of only 50 mg kg^{-1}. Therefore, PLAT predicts high P concentrations in runoff from these soils at relatively low STP levels. This level of STP was set as a threshold in these unique soils because of their pattern of P release. These 'shallow' organics have a substantial amount of mineral matter underneath a shallow organic surface (<66cm). When compared to poorly-drained mineral soils, these organics have been found to release relatively large amounts of P over successive extractions, as well as mineralizing a large amount of organic P, resulting in a much greater overall release of P when compared to mineral soils (Daughtrey et al., 1972). The low threshold of these soils in PLAT accounts for the higher predicted P concentration of runoff. The volume of runoff from these soils was intermediate, a finding that was likely due to the fact that all organic soils in our sample set were artificially drained which, in general, have higher predicted runoff, as described previously.

No clear trends for loss from applied P source were present despite the fact that the majority of sites with clay soils receiving animal waste had poultry waste applied to them. Sites with sandy soils had the largest mean value of available P from applied source, while sites with loam soils had the greatest proportion of rainfall occurring as runoff. It is evident that physiographic region, type of animal waste applications, and soil type are potentially being confounded in this situation.

Once again, surface runoff represented the largest component of overall P loss except on clay sites, which were not dominated by any one loss pathway. Subsurface drainage losses of P were also relatively important to overall P loss in sandy-textured soils.

Because site receiving animal waste are expected to be the areas of most concern in terms of preventing P loss to surface waters, the predicted impact of PLAT on each of the four threshold soil groups was examined (Table 15). Clay soils do not present as large of a P loss risk as do loam and sand soil groups.

Clays have a higher P threshold value and PLAT assumes they can accumulate to a greater level of STP before losing soluble P. On the other hand, loss through erosion can be significant on these soils. In the current sample set, however, this was not the case as the use of pasture systems on many of these sites seems to moderate this effect.

Table 15. Percentage of fields receiving animal waste in PLAT rating categories according to P threshold soil group.†

Threshold Group	n	PLAT Risk Category			
		Low	Medium	High	Very High
Clay	99	77%	14%	8%	1%
Loam	64	44%	22%	23%	11%
Sand	47	40%	21%	28%	11%

† P threshold soil groups have been defined in the PLAT description section above and are referred to by their generalized soil texture names.

Table 16. Mean values for PLAT index parameters according to P threshold soil group for sites receiving animal waste†.

		PLAT P Loss Pathway				
Threshold Group	**n**	**Erosion P Loss**	**Runoff P Loss**	**Drainage P Loss**	**Applied source P Loss**	**Total P Loss**
				lbs P ac^{-1}		
Clay	99	0.20	0.16	0	0.44	0.80
Loam	64	0.06	0.95	0.15	0.81	1.96
Sand	47	0.07	0.80	0.56	0.43	1.87
Organic	1	0	4.17‡	0	0.55	4.73

† P threshold soil groups have been defined in the PLAT description section above and are referred to by their generalized soil texture names.
‡Only one sample site in the organic soil group received animal waste.

CONCLUSIONS

Although no data were available to determine whether the PLAT index is accurately predicting quantities of P loss from individual fields, the tool was able to assign relative ratings of different sites' potential to contribute P to the environment. The results of 1379 fields showed that the tool predicted the areas in the state that are known to be disproportionately vulnerable to P loss due to histories of high P applications, high densities of animal units, or soil type and landscapes that are most susceptible to P loss. The tool also differentiated between the specific loss components that were prevalent in different soil types and under different field conditions and management, thus helping to target remediation strategies and future areas of research.

It is clear that, based on PLAT estimations, fields on which animal waste was applied represent a markedly higher risk of P loss from soil to water, and that animal producers are more likely to be forced to adjust their management practices in order to comply with federal and state regulations. Therefore, more attention and resources should be focused on these sites as they are more likely to contribute to degraded water quality. The application of poultry waste was predicted to result in greater overall losses of P when compared to other waste types, due to the higher P content of the waste. Dairy waste has an equally high total P content but the appropriate factors did not exist to allow for P transport off-field. Sites receiving swine waste tended to lose P as soluble P rather than source P, in both the soil surface and subsurface, because most of the P in this waste is in a soluble form.

Loss of P through surface runoff, and the parameters that are used to estimate it, presents an area of PLAT that should be targeted for future research efforts, as this loss pathway was relatively more important to P loss predictions, especially in sandy Coastal Plain sites that received animal waste. In soil groups with lower P threshold values, losses of soluble P through runoff became more significant, as it assumed that these soils have lower P sorption capacities. Additionally, sandy sites were more susceptible to subsurface drainage P losses. On average, PLAT appeared to identify suspected problem sites as well as the vulnerable P loss pathways of an individual site.

REFERENCES

Association of National Estuary Programs. National Estuary Programs Spotlight. http://www.anep-usa.org/nepspotlight/alb_pam/alb_pam.htm Verified June, 2004.

Cox, F.R. 1994. Predicting increases in extractable phosphorus from fertilizing soils of varying clay content. Soil Sci. Soc. Am. J. 58:1249-1253.

Crouse, D.R. 2000. North Carolina Nutrient Management Database Project. N.C. Cooperative Extension Service, N.C. State University. http://www.spatiallab.ncsu.edu/nutman/main.htm

Daughtrey, Z.W., J.W. Gilliam, and E.J. Kamprath. 1972. Phosphorus supply characteristics of acid organic soils as measured by desorption and mineralization. Soil Sci. 115:18-24.

Havlin, J.L., S. Hodges, D. Osmond, A. Johnson, D. Crouse, W. Skaggs, R. Evans, J. Parsons, P. Westerman, L. Price, R. Reich. 2001. Assessing the risk of phosphorus delivery to North Carolina waters. Proceedings International Symposium: *Addressing Animal Production and Environmental Issues*. Oct. 3-5 2001, Raleigh, NC.

Leonard, R.A., W.G. Knisel, D.A. Still. 1987. GLEAMS: Groundwater Loading Effects of Agricultural Management Systems. Trans. ASAE. 30:1403-1418.

McCollum, R.E. 1991. Buildup and decline in soil phosphorus: 30 year trends in Typic Umbraquult Agron. J. 83:77-85.

North Carolina Deptartment of Agriculture. and Consumer Services. Agricultural Statistics Division. 2002. http://www.ncagr. com/stats/index.htm.

North Carolina Department of Agriculture and Consumer Services. Agronomic Division. 2003. Annual Soil Test Summary.

North Carolina Department of Environment and Natural Resources (NCDENR). Division of Water Quality. Water Quality Section. 2002. Neuse River Basinwide Water Quality Plan. http://www.h2o.enr.state.nc.us/basinwide/.

North Carolina Department of Environment and Natural Resources (NCDENR). Division of Water Quality. Water Quality Section. 1994. Tar-Pamlico River Basinwide Water Quality Plan. http://www.h2o.enr.state.nc.us/basinwide/.

North Carolina Department of Environment and Natural Resources (NCDENR). Division of Water Quality. Water Quality Section. 2003. Tar-Pamlico River Basinwide Water Quality Plan. http://www.h2o.enr.state.nc.us/basinwide/.

North Carolina Department of Natural Resources (NCDENR). Division of Water Quality. Water Quality Section. Environmental Sciences Branch. 2000. Annual Report of Fish Kill Events. http://www.esb.enr.state.nc.us/Fishkill/2000killrep.pdf.

Renard, K.G., G.R. Foster, G.A. Weesies, and J.P. Porter. 1991. RUSLE: Revised Universal Soil Loss Equation. J. Soil Water Conserv. 46:30-33.

Skaggs, R.W., 1976. Evaluation of drainage-water table control systems using a water management model. IN: Drainage for Increased Crop Production and a Quality Environment. Proceedings of the Third National Drainage Symposium. ASAE. Chicago, IL. p61-68.

Skaggs, R.W. 1978. A water management model for shallow water table soils. Water Resources Research Institute, Univ. North Carolina. Tech. Rep. No. 134. Raleigh, N.C.

Skaggs, R.W., R.O. Evans, and G.M. Chescheir. 2004. Simple methods for predicting annual subsurface drainage and surface runoff from drained lands. Proceedings of the Eighth International Drainage Symposium, Sacramento, CA. ASAE, St. Joseph, MI. p367-376.

Sims, J.T., R.R. Simard, and B.C. Joern. 1998. Phosphorus loss in agricultural drainage: historical perspective and current research. J. Environ. Qual. 27:277-293.

U.S. Department of Agriculture – Soil Conservation Service. 1985. National Engineering Handbook, Section 4 – Hydrology. USDA-SCS. Washington, D.C.

U.S. Department of Agriculture – Natural Resources Conservation Service (USDA-NRCS). 2001. Field Office Technical Guide. USDA-NRCS. Washington, DC.

Chapter 3

The Evaluation of Uncertainty Associated with Predictions of Phosphorus Loss by North Carolina's Phosphorus Loss Index

INTRODUCTION

In areas with intensive livestock production, the over-application of animal waste has resulted in soil phosphorus (P) levels in excess of that needed for optimal crop growth. This situation has created conditions for an elevated risk of P movement off-site and its subsequent contribution to reduced water quality (Sharpley et al, 1994). Because of recent revised regulations concerning nutrient management, many states have implemented an index system to estimate the potential for site-specific P losses from fields receiving animal wastes. North Carolina is a major animal producing state and, therefore, generates large quantities of animal waste which, in situations where it has been land-applied for relatively long periods, has resulted in a buildup of soil test phosphorus (STP) throughout the state (N.C. Dept. of Agric. Consumer Services, Soil Testing Division, 2003). In response to federal regulations and public pressure, North Carolina has enacted state regulations that are either more restrictive than or point directly to NRCS's revised nutrient management standard (Standard 590), that considers P management (USDA-NRCS, 2001). In compliance with this standard, Standard 590, North Carolina has developed a site-indexing tool called the Phosphorus Loss Assessment Tool (PLAT) to estimate potential P loss from four loss pathways as well as total P loss: P loss through erosion, surface runoff, subsurface drainage and applied P source (Havlin et al., 2001).

Table 1 gives a brief overview of the structure of PLAT. More information on the algorithms involved in PLAT's estimation of P loss can be found at: **http://www.soil.ncsu.edu/nmp/ncnmwg/ncanat/plat/PLAT_Science_behind_the_tool.pdf**. The tool calculates individual loss pathways by multiplying source and transport factors pertaining to each pathway, in addition to any factors which may reduce the loss of P through that particular pathway. The four loss pathways are then summed to estimate a total edge-of-field P loss.

Empirical testing of the myriad of components that are part of the more complex P indexes is an arduous task at best. The cost- and time-intensive nature of such a proposal means that thorough validation of these tools with measured field data will not be accomplished in the near future. In the meantime, a knowledge of which input parameters affect the tool outcomes the most and carry the most uncertainty would be beneficial for allocating resources for future research aimed at improving state P indexes. This

Table 1. Overview of North Carolina's P indexing system, PLAT, and the inputs required for the calculation of each P loss pathway.

Total P loss (lbs P ac^{-1}) = Erosion P + Surface Runoff P + Subsurface Drainage P + Applied Source P	
Calculation of P Loss Pathway	**Parameters Involved**
Erosion P loss (lbs P ac^{-1}) = Erosion rate (tons ac^{-1}) * Total P (lbs P ton^{-1}) * Sediment trapping practices factor* Fe-P factor	STP (mg kg^{-1}) % clay Soil type Buffer width (ft) Depositional slope distance (ft)
Surface runoff P loss (lbs P ac^{-1}) = Runoff P concentration (mg L^{-1}) * Runoff volume (in)	STP (mg kg^{-1}) Soil type Curve Number Rainfall (in) Transmissivity (ft^2 hr^{-1}) Drain depth (ft) Drain spacing (ft)
Subsurface drainage P loss (lbs P ac^{-1}) = Drainage P concentration (mg L^{-1}) * drainage volume (in)	Subsurface STP (mg kg^{-1}) Rainfall (in) Evapotranspiration (in) Runoff volume (in) Transmissivity (ft^2 hr^{-1}) Drain depth (ft) Drain spacing (ft)
Applied Source P loss (lbs P ac^{-1}) = Available source P (lbs P ac^{-1}) * Application method/timing factor * Runoff factor	P source type Application rate (unit ac^{-1}) P content (lbs P unit^{-1}) P source solubility Rainfall (in) Runoff volume (in)

can be accomplished by applying statistical methods that are generally used in evaluating hydrologic models.

A common method used to evaluate hydrologic/water quality models is sensitivity analysis/uncertainty analysis (Haan et al., 1995). Performing a sensitivity analysis on model inputs identifies the parameters that have the greatest effect on model predictions, which then receive most of the attention in the remainder of the model evaluation. Sensitivity analysis gives information on how much change can be expected in a model output per unit of change from an input parameter. However, sensitivity analysis ignores the uncertainty in each input (Haan and Skaggs, 2003).

Uncertainty in model inputs needs to be understood in order to understand the uncertainty in model outputs. Uncertainty analysis considers the inherent uncertainty in model input parameters and examines how that affects the model outputs, thereby helping to assess uncertainty in output predictions. Output uncertainty can then be reduced by concentrating on the input parameters that are the largest contributors to uncertainty. Determining to what degree model inputs cause variability in model results is the goal of uncertainty analysis while sensitivity analysis indicates the impact of this variability on model predictions (Haan et al., 1995).

Vicens et al. (1975) defined three types of uncertainty in hydrologic models: inherent variability in natural processes, model uncertainty and parameter uncertainty. Model and parameter uncertainty fall under the general heading of knowledge uncertainty and refer to inadequate understanding or measuring of the system being modeled (Hession et al., 1996). Uncertainty in the natural process is due to random variability found in nature. In nature the same inputs will produce a whole range of outcomes, whereas a model will consistently produce the same output for a given set of inputs. Therefore, in order to be able to make management/policy decisions that more accurately reflect what occurs in nature, we need to assess the likelihood of a particular model output occurring (Helton, 1993). For example, one can determine if the uncertainty in a predicted quantity, such as the release of a toxic material from a disposal site, falls within a specified set of boundaries, i.e. below some regulatory limit. Uncertainty analysis is a method of quantitatively determining probabilities of possible outcomes (model outputs).

Morgan and Henrion (1990) state that uncertainty in a particular policy decision should be a concern when: peoples' (producers') attitudes toward risk in the analysis being performed (application of the PLAT index to field sites, in our case) is likely to be important; when uncertain information from different sources is being combined; and when it is necessary to determine the value of expending resources to acquire additional information. It is easy to see that these reasons for determining uncertainty can certainly apply to P indexes.

The overall goal of the current study was to evaluate the PLAT index and to determine which areas deserve most of our research efforts. Specifically, we wanted to determine which input parameters were contributing the most to the tool’s predictions. An additional objective was to assess the impact of uncertain input parameters on the uncertainty of the index’s outputs. An evaluation of the predictive ability of PLAT, where we had no observed data on the quantities being estimated, was desired. This is not to be confused with validation of PLAT, or evaluating the accuracy of the tool’s prediction, as this cannot be done without making comparisons to measured field data. Validation with measured data would be the last step in the overall evaluation of PLAT. As more research is done, various sections of PLAT will be able to be validated. A number of studies evaluating specific components of PLAT have recently been initiated.

METHODOLOGY

The method used to evaluate the PLAT index was that of Haan, et al. (1995) who suggested the following steps:

- conduct a sensitivity analysis on input parameters.
- generate probability distributions on input parameters.
- generate probability distributions on outputs.
- assess the model by using output probability distributions.

Input parameters of PLAT were collected from 1465 sites encompassing all 100 counties in the state of NC in order to account for the diversity of conditions for which PLAT will be used. The fact that we had field-derived values provided a unique situation for evaluating the PLAT index. In many cases, data on input parameters do not exist or are insufficient and must be obtained from reported values in the literature, or simply set using best scientific judgment (Haan et al., 1995).

Sensitivity Analysis

A sensitivity analysis evaluates the relationship between changes in an individual input parameter and changes produced in the output. The absolute (S) and relative sensitivities (S_r) were calculated using the following :

$$S = \frac{\Delta O}{\Delta I} \qquad\qquad S_r = \frac{\Delta O}{\Delta I}\,\frac{I_b}{O_b} \qquad\qquad (1)$$

where O represents a particular output, I represents a particular input, I_b is the base input value and O_b is the value of the output calculated at the base input. When evaluating a particular input, all other parameters were held constant at their base values. Base values were selected based on the mean value for each parameter from the 1465 sample sites and are shown in Table 2. Base values for the four loss pathways are highlighted and were estimated by holding all input parameters involved in that pathway at their base values. In most cases, only the sensitivity of the overall amount of P loss to various input parameters was estimated. In a few cases, the amount of P lost from individual loss pathways was used as the objective function.

Table 2. Base values for parameters used in the sensitivity analysis of PLAT predictions of P loss from four P loss pathways.†

PLAT Parameter	Base Value
Erosion rate (tons ac^{-1})	1.6
Soil test P (mg kg^{-1})	119
% clay	14
Sediment trapping practices factor	0.5
Fe-P factor	0.4
Erosion P loss (lbs P ac^{-1})‡	0.12
Runoff concentration (mg L^{-1})	0.6
Runoff volume (in)	2.75
Surface runoff P loss (lbs P ac^{-1})‡	0.37
Subsurface STP (mg kg^{-1})	26
Subsurface drainage volume (in)	12.65
Subsurface drainage P loss (lbs P ac^{-1})‡	0.37
Source available P (lbs P ac^{-1})§	3.94
Application method/timing factor	0.99
Runoff fraction	0.05566
Applied source P loss (lbs P ac^{-1})‡	0.22
Total P loss (lbs P ac^{-1})‡	1.08

†Additional base conditions are that the site is a loam soil located in the Coastal Plain.
‡Base values for outputs are calculated from input parameters at their base values.
§ Source available P = application rate (unit ac^{-1}) * P content (lbs P $unit^{-1}$) * Attenuation factor.

Given the diverse situations for which PLAT is expected to perform and because of the site specificity of the analysis (Ferreira et al., 1995), additional base scenarios were designed to represent conditions that were dominant in the state or that we had particular curiosity about. In this way, the sensitivity of inputs in different situations could be assessed since some parameters may have differing impacts on the tool predictions, depending on conditions under which it is run. In particular, we made comparisons between sites under different drainage conditions in order to examine the impacts of the different methods and inputs involved in calculations of surface runoff volume and subsurface drainage volume for each drainage condition.

The PLAT index classifies all soils into one of four soil groups based on their relative ability to hold P. These groups are generally referred to based on soil texture. Above a certain threshold STP level, it is assumed that soil in a particular group will retain P less strongly and release P into solution at a greater rate than it did below the threshold STP. Therefore, we made comparisons between PLAT predictions as impacted by STP among the different soil groups. In this case all other factors beside STP were held at the same base values across all soil types. Although this may not represent conditions that would be realistically found in the field, i.e. an organic soil will not have as much soil erosion as a clay soil, we wanted to avoid the confounding effect of other input parameters. Due to the importance of STP in PLAT, the impact of a soil's P status on predicted P loss was evaluated. We also examined the effect of using different P amendments on overall PLAT predictions, by comparing different types of animal wastes or inorganic fertilizer. Baseline values corresponding to the different scenarios for which sensitivity analysis was performed are shown in Tables 3, 4 and 5.

Uncertainty Analysis

Probability distributions were generated for input parameters based on our 1465 sampled sites. The availability of these 1465 data points is fortuitous as the lack of information on the mean and standard deviation of a population can make it difficult to produce input probability distributions, from which inputs can be randomly sampled. Again base values were based on the means of our collected data. The choice of a specific probability density function (pdf) also requires some knowledge of the input population. In the current study, target probability density functions were chosen based on visual observation of input distributions. Additionally, the Kolmogorov-Smirnov goodness-of-fit test was performed in order to statistically examine the nature of our probability distributions. Random samples of input parameters of interest were created using a Monte Carlo Simulation (MCS). This involved using the assumed pdf of each parameter and its accompanying statistics, to randomly choose parameters from the parent distribution. Values from these selected distributions were then used as input parameters for running PLAT. The

Table 3. Base values for naturally and artificially drained scenarios used for PLAT sensitivity analysis.†

Parameter	Natural Drainage	Artificial Drainage
Erosion rate (tons ac^{-1})	1.6	1.6
STP (mg kg^{-1})	119	119
% clay	14	14
Erosion P loss (lbs P ac^{-1})	0.12	0.12
Runoff concentration (mg L^{-1})	0.60	0.60
Runoff volume (in)	1.60	2.61
CN	70	--
Drainage intensity (ft hr^{-1})	--	0.73
Tp-1 (ft^2 hr^{-1})‡	--	1.27
Drain spacing (ft)	--	175
Drain depth (ft)	--	2.75
Runoff P loss (lbs P ac^{-1})	0.22	0.35
Subsurface STP (mg kg^{-1})	26	26
Rainfall (in)	48.92	50.97
Evapotranspiration (in)	34.9	--
T_{30}/T_p§	--	0.49
Drainage volume (in)	12.2	15.07
Drainage P loss (lbs P ac^{-1})	0.37	0.22
Source available P (lbs P ac^{-1})¶	3.94	3.94
Application timing/method	0.99	0.99
Runoff fraction	0.0327	0.0512
Applied source P loss (lbs P ac^{-1})	0.127	0.200
Total P loss (lbs P ac^{-1})	0.833	0.894

†Additional conditions are that both scenarios are on loam soils in the Coastal Plain.
‡Transmissivity of the soil profile excluding the top 1 foot of soil.
§Transmissivity factor = transmissivity of the 30-80 cm (12-30 in) soil depth / transmissivity of the entire soil profile.
¶Source available P = application rate (unit ac^{-1}) * P content (lbs P unit^{-1}) * Attenuation factor.

Table 4. Base values of STP and consequent values of P loss pathways for different site scenarios.

Base scenario	Surface STP	Subsurface STP	Erosion P loss	Runoff P loss	Drainage P loss	Applied source P loss	Total P loss
	mg kg^{-1}		lbs P ac^{-1}				
Organic	119	26	0.123	1.483	0.372	0.217	2.195
Sand	119	26	0.123	0.741	0.372	0.217	1.453
Loam	119	26	0.123	0.371	0.372	0.217	1.083
Clay	119	26	0.123	0.148	0.372	0.217	0.860
High STP	450	26	0.466	1.402	0.372	0.217	2.456
Low STP	50	0	0.052	0.156	0	0.217	0.424

Table 5. Parameter base values for different P source type scenarios.

Parameter	Inorganic fertilizer	Dairy†	Poultry‡	Swine§
P content (lbs P application $unit^{-1}$)	0.2024	6.16	34.54	23.32
P solubility¶	1.0	0.75	0.25	0.80
Application rate (application unit ac^{-1})	87.4	20.0	2.8	2.5
Source available P (lbs P ac^{-1})#	1.77	29.89	14.69	5.50
Buffer reduction factor††	--	0.683	0.683	0.683
Applied source P loss (lbs P ac^{-1})	0.11	1.45	0.72	0.27

†Slurry.
‡Broiler house litter.
§Lagoon liquid.
¶Expressed as a proportion of 100% solubility (0 - 1.0)
#Source available P = application rate (unit ac^{-1}) * P content (lbs P $unit^{-1}$) * Attenuation factor.
††Reduction factor applied to non-soluble portion of P amendment.

outputs from these index runs were used to generate output probability distributions which could then be used to asses the model (Haan et al., 1995). This process was performed 2000 times for each input of interest. In some cases the randomly sampled input parameters were truncated in order to fall within realistic ranges. For example, mass balance dictates that the volume of runoff cannot exceed the amount of rainfall at a location. Therefore, values of runoff and subsurface drainage exceeding the maximum annual rainfall for North Carolina were excluded. Curve Number values above 100 were also discarded. Additionally, if random sampling from the fitted pdf produced physically meaningless negative values for input parameters, these were not used to obtain output pdfs. Cumulative probability functions were generated from the outputs of the PLAT index runs. The cumulative pdf's were then used to estimate confidence intervals for each objective function produced from an individual input distribution.

RESULTS

Sensitivity analysis

Relative sensitivities of input parameters at their base values are shown in Table 6. The two input parameters that are used to predict P loss in more than one loss pathway, STP and runoff volume, had the greatest impact on P loss predictions. Soil test P is used to estimate the total P content of eroded soil as well as the amount of P in solution that is lost by runoff. Both P loss via surface runoff and the runoff fraction used to calculate applied source P loss use an estimate of runoff volume. Because of the multiplicative approach of PLAT calculations, input parameters within the same P loss pathway have the same relative sensitivity unless they are non-linear and or used in more than one calculation. For example, percent clay is an input used to estimate erosion P loss that is not linear (Figure 1). Sensitivity of the model to clay percentage increases as clay increases. Its contribution to total P loss, however, is still relatively low for the average amounts of clay found in soils of North Carolina. The sensitivity of each of the four loss pathways on the total estimated P loss can be seen in Figure 2a. Drainage and runoff P have a greater impact on overall loss predictions as compared to erosion and applied source P losses. For this base scenario, soluble P, both in surface runoff and subsurface drainage, dominated overall P losses while the loss of particulate P through erosion or applied source were of minor impact to final P loss predictions.

Table 7 presents a comparison of the relative sensitivities of inputs from two different drainage scenarios. Sensitivity to erosion losses was relatively minor and similar for both cases. Input parameters used to calculate runoff losses of P (runoff P concentration, runoff volume) were more sensitive in the case of artificial drainage and less so for subsurface drainage (Figures 2b and 2c, Table 7). Loss of P via applied source was slightly more impacted by input parameters in the artificially drained scenario, most likely because runoff volume factors into the estimation of the runoff fraction parameter. The modeled P losses were 1.7 fold more sensitive to runoff volume for artificial drainage compared with natural drainage.

The finding that runoff losses were more sensitive in artificially drained situations and drainage losses more sensitive in naturally drained situations may at first be counter-intuitive as the presence of artificial drainage is generally thought to increase the volume of water draining through the soil profile. However, this result may be due to the estimated amounts of the two hydrologic pathways and the different methods PLAT uses to estimate them. The method of computing flow volumes on artificially drained fields has been shown to estimate greater amounts of both surface and subsurface water loss than does the more approximate technique used to predict these parameters under natural drainage. Despite the greater projected volume under drained situations, the subsurface drainage loss of P is reduced by the T_{30}/T_p factor.

Table 6. Relative sensitivities (S_r) of predicted P loss to input parameters from the baseline scenario involving mean inputs for all sites.

PLAT Parameter	Relative Sensitivity
Erosion rate	0.114
STP	0.456
% clay	Varies
Sediment trapping practices factor	0.114
Fe-P factor	0.114
Erosion P loss	0.114
Runoff concentration	0.342
Runoff volume	0.543
Surface runoff P loss	0.342
Subsurface STP	0.344
Subsurface drainage volume	0.344
Subsurface drainage P loss	0.344
Source available source P	0.200
Application method/timing factor	0.200
Runoff fraction	0.200
Applied source P loss	0.200

This factor assumes that drainage water and any dissolved P that it carries encounters only the top 30 inches of soil before it contacts drain pipes, which presumably redirect it out of the soil system. This factor acts to reduce the amount of P in subsurface drainage by a significant factor. No such phenomenon occurs in natural drainage situations, or at least according to the PLAT index. Therefore, naturally drained sites are predicted to experience greater P loss through subsurface drainage. This may account for the greater sensitivity of PLAT outputs to runoff volumes and lower drainage volumes in the artificially drained scenario, as the parameters that have a greater estimated value would be weighted more heavily in relation to PLAT outputs.

Table 7. Relative sensitivities (S_r) of PLAT predictions of P loss to inputs for two different drainage scenarios.

Parameter	Natural Drainage	Artificial Drainage
Erosion rate	0.148	0.138
STP	0.407	0.532
Erosion P loss	0.148	0.138
Runoff P concentration	0.259	0.394
Runoff volume	0.356	0.617
CN	Varies	--
Drainage intensity	--	Varies
Tp-1	--	Varies
Drain spacing	--	Varies
Drain depth	--	Varies
Runoff P loss	0.259	0.394
Subsurface STP	0.439	0.496
Rainfall	1.730	--
Evapotranspiration	-1.234	--
T_{30}/T_p‡	--	0.245
Drainage volume	0.439	0.245
Drainage P loss	0.439	0.245
Source available P	0.153	0.223
Runoff fraction	0.153	0.223
Applied source P loss	0.153	0.223

A few input parameters that are used to estimate runoff P loss are non-linear, and their sensitivity will therefore vary according to the value of the input parameter. Figure 3 shows that as curve number (CN) increases, the sensitivity of P loss increases. That is, the impact of CN on PLAT predicted P outputs increases as the CN of a site increases. In addition, CN values for coarser textured soils have a greater impact on outputs than CN for either loams or clays.

Figures 4a-d show how various intermediate parameters used to estimate runoff and drainage volumes on artificially drained sites vary with respect to relative sensitivity. These parameters are, in general, more influential on PLAT estimates of surface runoff loss than on subsurface drainage loss. Drain spacing, drain depth and transmissivity (T_p-1) are parameters that are used to calculate drainage intensity (DI) that is then used to estimate runoff and drainage volume. All four parameters approach S_r values of zero as they increase. In some cases, relative sensitivity can be quite high (10) at low parameter values, as for transmissivity, but very quickly decrease to having almost no impact on P loss.

The sensitivity of erosion and runoff P loss to STP was the same regardless of soil type, although relatively high (Table 8). Comparisons between total P loss predictions between the four soil groups show that as P threshold decreases, sensitivity to values of STP increase, with STP being relatively sensitive in organic soils and insensitive in clay soils. This result suggests the importance of accurately measuring STP on soils with lower P sorption capacities. Figure 5 shows the absolute sensitivities of predicted total P loss to STP in the different soil types as well as the two different STP status scenarios. Sites with a relatively low level of STP buildup

Table 8. Relative sensitivities (S_r) of PLAT objective functions to STP for 4 different soil groups.

Soil Group	P Threshold (mg kg^{-1})	Objective Function		
		Erosion P Loss	Surface Runoff P Loss	Total P Loss
		Relative Sensitivity		
Clay	500	1.0	1.0	0.316
Loam	200	1.0	1.0	0.456
Sand	100	1.0	1.0	0.595
Organic	50	1.0	1.0	0.732

have less of an impact on PLAT outputs as compared with soils that have been built up to a high level of STP (Table 9).

Table 10 compares relative sensitivities of total P loss to input parameters for various P amendment types. Inputs on sites receiving dairy wastes were more sensitive than on sites receiving other sources of P. The absolute sensitivities shown in Figure 6a further illustrate the importance of components of dairy waste. Swine was the least sensitive of the animal wastes while inorganic fertilizer had very little impact on PLAT outputs. One parameter associated with the reduction of non-soluble P is buffer width. Vegetative buffers can act to trap particulate matter from waste containing a non-soluble fraction. Figure 6b shows that although some reduction in P loss can be obtained from the presence of a vegetative buffer, it is relatively minor, with the sharpest decrease occurring in the first 15 feet of buffer width.

Table 9. Relative sensitivities (S_r) of PLAT objective functions to STP for two P status scenarios.

STP Scenario	Objective Function		
	Erosion P Loss	Surface Runoff P Loss	Total P Loss
	Relative Sensitivity		
High P Status	1.0	1.0	0.760
Low P Status	1.0	1.0	0.489

Table 10. Relative sensitivities of predicted P loss to input parameters for different P source types.

	Input Parameters				
P Amendment Type	Inputs related to soluble nature of P amendment†	Inputs related to non-soluble nature of P amendment†	Application Rate	Source Available P	Applied source P loss
	Relative Sensitivity				
Dairy	0.610	0.046	0.655	0.655	0.655
Poultry	0.320	0.164	0.483	0.483	0.483
Swine	0.221	0.038	0.259	0.259	0.259
Inorganic§	0.101	--	0.101	0.101	0.101

†Includes soluble P content and soluble P attenuation factor of each P amendment type.
‡Includes non-soluble P content, non-soluble P attenuation factor, and buffer reduction factor.
§Inorganic fertilizer is assumed to be 100% soluble.

Uncertainty analysis

Output pdfs generated from the Monte-Carlo simulation (MCS) can quantify uncertainty in P loss predictions made by PLAT. Confidence intervals may be placed on these to give an indication of the reliability of the estimate and can be used as a measure of validity of the model (Haan, C.T. 1989). Cumulative probability distributions of predicted P loss for specific input parameters of interest are shown in the Appendix. Lines indicating the 5% and 95% cumulative percent levels are included. These generated probability distributions describing outputs were visually examined to fit a normal or lognormal distribution and analyzed statistically using the Kolmogorov-Smirnov (K-S) goodness-of-fit test. The results of the K-S test are shown in Table A1. Additionally, Haan and Skaggs (2003) state that another check of how well the MCS performed is to examine how closely the probability density functions of randomly generated input parameter values fit the population density function from which they came from. Comparisons of parent and simulated parameter distributions of the current study are shown in Table A1. In general, randomly generated input distributions seemed to fit their parent distributions, although not always statistically. The exceptions are the soil scenarios examining the effect of STP on model uncertainty. With the exception of the low P status scenario, the means and standard distributions of the simulated data for the different soil type scenarios are quite different from the means and standard deviations of the parent distributions. This disparity suggests that in the case of the STP scenarios, the MCS did not perform well in terms of accurately simulating data from the population data distribution. Morgan and Henrion (1990) stated that with large, empirical data sets it is likely that statistical tests will reject any parameteric distribution function even if it provides a reasonable approximation to the observed distribution.

Confidence intervals can be determined from cumulative probability distributions (Figures 7-20) and are shown in tables 11, 12 and 13 for the different test scenarios. Table 11 compares the uncertainty in parameters related to the two different drainage scenarios. For example, the results indicate that 90% of the time PLAT will predict a total P loss between 0.797 – 1.022 lbs P ac^{-1}, given the values of subsurface drainage volume that are likely to be found in artificial drainage situations for which PLAT will be used. This is a relatively small confidence interval and suggests that PLAT is performing well in predicting P loss, at least under these particular set of conditions. The width of the confidence intervals for P loss when

Table 11. Comparison of confidence intervals on total P loss predictions from sites with natural and artificial drainage. Confidence intervals are based on empirical distributions generated by Monte Carlo simulation with runoff and drainage volume used as an input parameters.

Objective Function	Confidence Level %	Confidence Intervals	
		Natural drainage	Artificial drainage
		lbs P ac^{-1}	
Runoff volume (in)	80	0.757 – 1.393	0.657 – 2.625
	90	0.725 – 1.607	0.618 – 3.756
	95	0.704 – 1.963	0.597 – 5.173
Drainage volume (in)	80	0.626 – 0.990	0.820 – 0.996
	90	0.579 – 1.040	0.797 – 1.022
	95	0.543 – 1.090	0.774 – 1.044

Table 12. Comparison of confidence intervals on total P loss predictions from four different P threshold soil groups and two different STP levels. Confidence intervals are based on empirical distributions generated by Monte-Carlo simulation with STP used as an input parameter.

Scenarios	Confidence Level (%)	Confidence Intervals
		lbs P ac^{-1}
Organic	80	0.592 – 10.856
	90	0.590 – 16.944
	95	0.589 – 20.911
Sand	80	0.590 – 6.202
	90	0.589 – 9.219
	95	0.589 – 11.435
Loam	80	0.589 – 5.651
	90	0.589 – 9.640
	95	0.589 – 13.032
Clay	80	0.589 – 3.398
	90	0.589 – 5.043
	95	0.589 – 6.767
High STP	80	0.590 – 7.054
	90	0.589 – 11.107
	95	0.589 – 14.358
Low STP	80	0.255 – 0.410
	90	0.240 – 0.432
	95	0.232 – 0.451

considering runoff volume as the uncertain input parameter are much greater for artificially drained situations while confidence intervals for the naturally drained scenario had a slightly greater width for predicted P loss with subsurface drainage as the uncertain input.

Width of confidence intervals was greatest when estimating P loss from organic soil types and least from clay soil types (Table 12). Confidence intervals were quite wide for the different soil type scenarios, suggesting that uncertainty in STP should be reduced, especially in organic soil types and high P soils, in order to reduce output uncertainty. The ability of PLAT to predict P loss from low P soils, based on uncertainty in STP, appears to be relatively good as indicated by the low confidence intervals.

Table 13 implies that where dairy waste is applied, predicted outputs will be less certain than other P amendment source types based on available P in the particular source. Predicted P loss from sites receiving poultry waste had slightly narrower confidence intervals followed by sites receiving swine waste. Where inorganic fertilizer is used, we can predict the mean P loss value with greater certainty.

Table 13. Comparison of confidence intervals on P loss predictions from sites receiving different P source types. Confidence intervals are based on empirical distributions generated by Monte Carlo simulation with source available P as an input parameter.

P Source Type	Confidence Level (%)	Confidence Intervals
		lbs P ac^{-1}
Dairy	80	0.871 – 3.562
	90	0.867 – 4.516
	95	0.867 – 4.935
Poultry	80	1.142 – 2.015
	90	1.051 – 2.148
	95	0.983 – 2.261
Swine	80	0.983 – 1.418
	90	0.939 – 1.489
	95	0.915 – 1.548
Inorganic fertilizer	80	0.907 – 0.983
	90	0.901 – 1.003
	95	0.897 – 1.028

SUMMARY

A method to evaluate the predictive ability of the PLAT index where there are no observed data on the quantities being predicted was presented. Input parameters expected to have the greatest impact were identified and uncertainty in outputs were quantified in the form of probability density functions.

Our results showed that soluble loss pathways, surface runoff, and subsurface drainage impact predictions of P loss more than either loss of P through erosion or applied source. Future research efforts should focus on these components of PLAT. Runoff P loss was more sensitive in artificially drained situations while drainage was more sensitive in naturally drained situations. Loss of P from soil types with lower P threshold values were more impacted by STP, as were soils with relatively higher STP levels. Available P from applied source impacted total P loss predictions more in sites receiving dairy wastes than sites on which other P amendments were applied.

Future initiatives

The aim of future research is to decrease uncertainty in the predicted P loss estimates of the PLAT index. A quantitative estimate of which input parameters are the largest contributors to variability in the tool is needed. An input parameter may have a high level of sensitivity to predicted outcomes, but the uncertainty it contributes to the overall uncertainty of the tool's predictive ability is an important factor to consider. Additional base scenarios for different situations should be considered as the present study only concerned a few of the myriad of conditions for which PLAT will be used. An examination of the correlation structure of PLAT would be beneficial as the present study assumed parameter independence, which may not be the case.

References

Ferreira, V.A., G.A. Weesies, D.C. Yoder, G.R. Foster, and K.G. Renard. 1995. The site and condition specific nature of sensitivity analysis. J. Soil Water Conservation. 50:493-497.

Haan, C.T. 1989. Parameter uncertainty in hydrologic modeling. Trans ASAE 32:137-146.

Haan, C.T., B. Allred, D.E. Storm, G. J. Sabbagh, S. Prabhu. 1995. Statistical procedure for evaluating hydrologic/water quality models. Trans. ASAE 38:725-733.

Haan, P.K., and R.W. Skaggs. 2003. Effect of parameter uncertainty on DRAINMOD predictions: II. Nitrogen loss. Trans. ASAE 46: 1069-1075.

Havlin, J.L., S. Hodges, D. Osmond, A. Johnson, D. Crouse, W. Skaggs, R. Evans, J. Parsons, P. Westerman, L. Price, R. Reich. 2001. Assessing the risk of phosphorus delivery to North Carolina waters. Proceedings International Symposium: *Addressing Animal Production and Environmental Issues.* Oct. 3-5 2001, Raleigh, NC.

Helton, J.C. 1993. Uncertainty and sensitivity analysis techniques for use in performance assessment for radioactive waste disposal. Reliability Engineering and System Safety. 42:327-367.

Hession, W.C., D.E. Storm, and C.T. Haan. 1996. Two-phase uncertainty analysis: An example using the Universal Soil Loss Equation. Trans. ASAE. 39:1309-1319.

Morgan, M.G. and M. Henrion. 1990. Uncertainty. Cambridge, Mass.: Cambridge University Press.

Sharpley, A.N., S.C. Chapra, R. Wedephol, J.T. Sims, T.C. Daniels, and K.R. Reddy. 1994. Managing agricultural phosphorus for protection of surface waters: Issues and options. J. Environ. Qual. 29:1462-1469.

U.S. Department of Agriculture – Natural Resources Conservation Service (USDA-NRCS). 2001. Field Office Technical Guide. USDA-NRCS. Washington, DC.

Vicens, G.J., I. Rodriquez-Iturbe, and J. C. Shaake. 1975. A Bayesian framework for the use of regional information in hydrology. Water Resources Research. 11:405-414.

Appendix

Table A1. Population statistics and input pdfs, randomly generated sample statistics and Kolmogorov-Smirnov (K-S) goodness-of-fit test results for predicted P loss used in Monte-Carlo simulation.

Parameter	Population properties	Sampled from MCS	K-S test results for output distributions
Runoff volume (in), naturally drained sites	? = 2.43 $s = 1.98$ lognormal	$x = 3.06$ $s_x = 2.32$	Lognormal Reject
Runoff volume (in), artificially drained sites	? = 3.79 $s = 3.18$ lognormal	$x = 6.76$ $s_x = 10.64$	Lognormal Reject
Subsurface drainage volume (in), naturally drained sites	$\mu = 11.57$ $s = 4.76$ normal	$x = 11.58$ $s_x = 4.88$	Normal Do not reject
Subsurface drainage volume (in), artificially drained sites	$\mu = 16.11$ $s = 4.84$ normal	$x = 16.05$ $s_x = 4.71$	Normal Do not reject
Available P (lbs P ac^{-1}), sites receiving dairy waste	$\mu = 20.13$ $s = 31.98$ lognormal†	$x = 17.12$ $s_x = 24.72$	Lognormal Reject
Available P (lbs P ac^{-1}), sites receiving poultry waste	$\mu = 16.37$ $s = 8.41$ normal	$x = 16.86$ $s_x = 7.81$	Normal Do not reject
Available P (lbs P ac^{-1}), sites receiving swine waste	$\mu = 7.03$ $s = 3.83$ normal†	$x = 7.28$ $s_x = 3.63$	Normal Do not reject
Available P (lbs P ac^{-1}), sites receiving inorganic fertilizer	$\mu = 1.13$ $s = 0.66$ lognormal	$x = 1.24$ $s_x = 0.53$	Lognormal Reject
STP (mg kg^{-1}), organic soil types	$\mu = 99.2$ $s = 76.1$ lognormal†	$x = 224.1$ $s_x = 413.1$	Lognormal Reject
STP (mg kg^{-1}), sand soil types	$\mu = 178.3$ $s = 115.6$ lognormal	$x = 226.4$ $s_x = 420.2$	Lognormal Reject
STP (mg kg^{-1}), loam soil types	$\mu = 129.6$ $s = 130.1$ lognormal	$x = 393.2$ $s_x = 855.1$	Lognormal Reject
STP (mg kg^{-1}), clay soil types	$\mu = 94.7$ $s = 122.5$ lognormal	$x = 376.1$ $s_x = 822.3$	Lognormal Reject
STP (mg kg^{-1}), high P status soils	$\mu = 165.5$ $s = 133.5$ lognormal	$x = 480.1$ $s_x = 946.0$	Lognormal Reject
STP (mg kg^{-1}), low P status soils	$\mu = 26.4$ $s = 15.45$ normal	$x = 27.8$ $s_x = 14.0$	Normal Reject

† The distributions of these inputs fit their assumed probability distribution function based on the Kolmogorov-Smirnov goodness-of-fit statistical test.

Recommendations for Future Research

The following are recommendations based on the research contained herein, provided to assist in determining which areas to focus future research efforts in an effort to improve and update PLAT.

- The PLAT index uses STP in addition to dividing soils into one of four threshold groups, each with a critical P threshold level, to indicate the susceptibility of P to be dissolved from soil. Our study showed a correlation between STP and P saturation determined by oxalate extraction, a more rigorous measure of potential dissolved P loss. However, more study should be initiated to determine: (i) if critical threshold values for the different threshold soil groups used in PLAT adequately describe a soil's vulnerability to dissolved P loss; and (ii) appropriate alpha values that are involved in estimating soil P sorption capacity for the different threshold soil groups.
- It is clear that interactions of organic acids with iron and aluminum in organic soils of North Carolina are poorly understood. Therefore, the behavior of these unique soils with respect to P retention needs to be studied further.
- This study showed that Mehlich-3 is a poor extractor of iron, yet there was a significant correlation between M3P / (M3Al +M3Fe) and P_{ox} / (Al_{ox} + Fe_{ox}). The question of whether iron is an important factor affecting P loss needs to be addressed. The dynamic behavior of Fe and its interactions with P is not fully understood, especially in drainage areas experiencing cycles of reduction and oxidation.
- An estimate of the variability that each input parameter of PLAT is contributing to the overall variability of the index is necessary.
- Sensitivity of embedded factors, such as soil hydrologic group when estimating curve number or hydraulic conductivity when estimating drainage, intensity should be assessed to .
- The sensitivity/uncertainty analysis done in this study assumed no correlation between individual input parameters. However, this may not be a good assumption and, an examination of the level of dependence between PLAT input parameters should be undertaken.

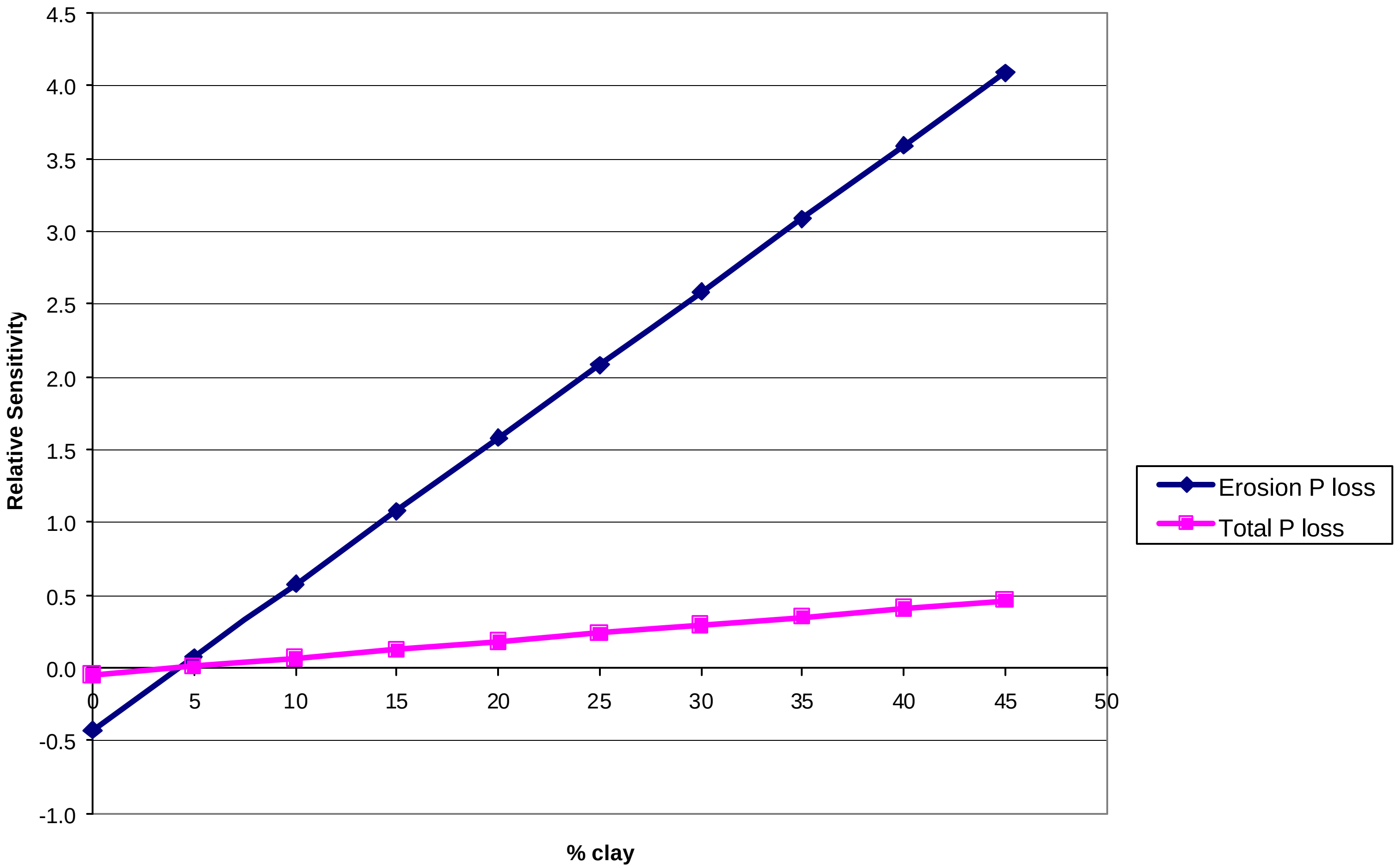

Figure 1. Changes in relative sensitivity of various objective functions with percent clay.

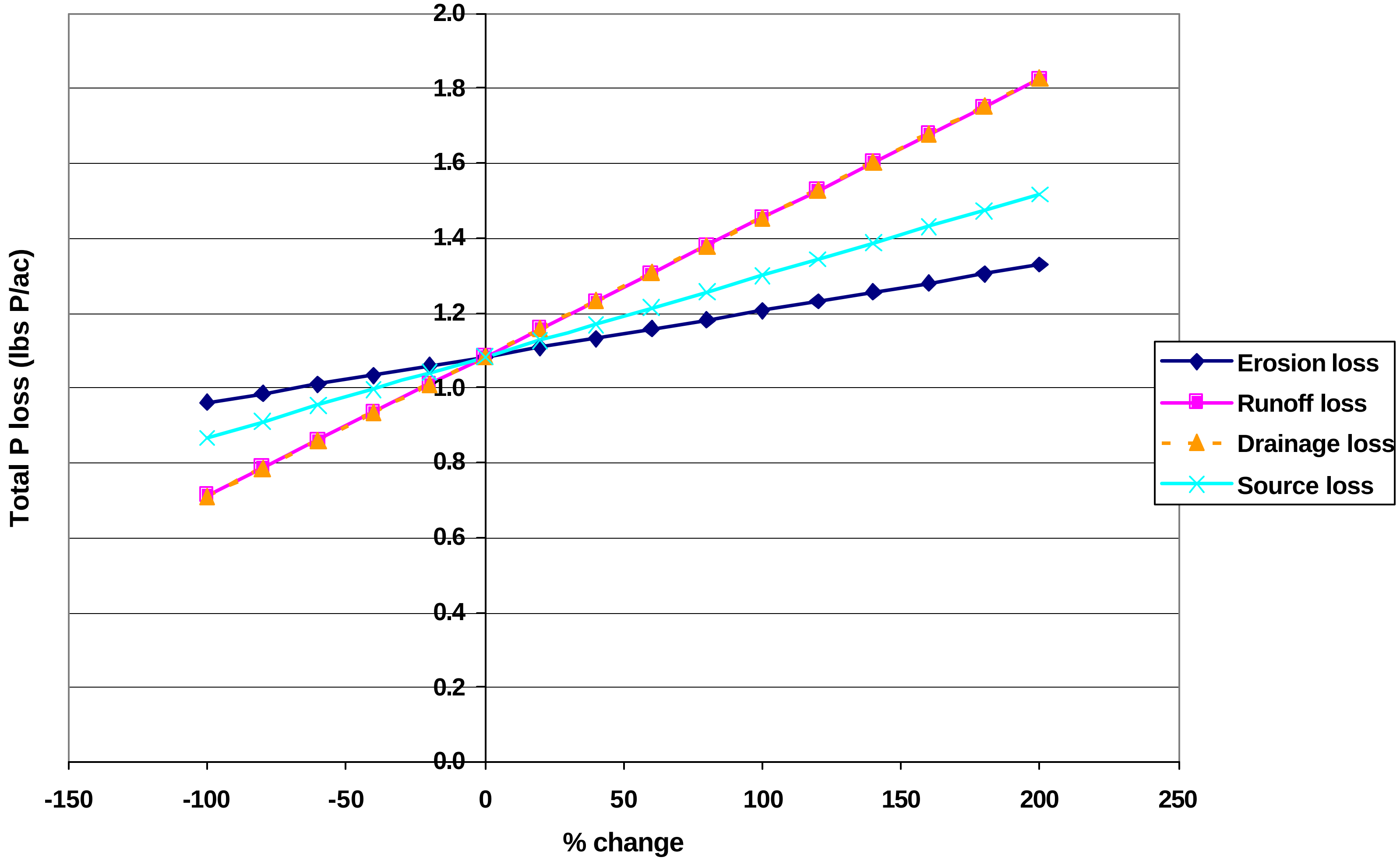

Figure 2a. Sensitivity of P loss predictions to changes in the four PLAT P loss pathways under baseline conditions.

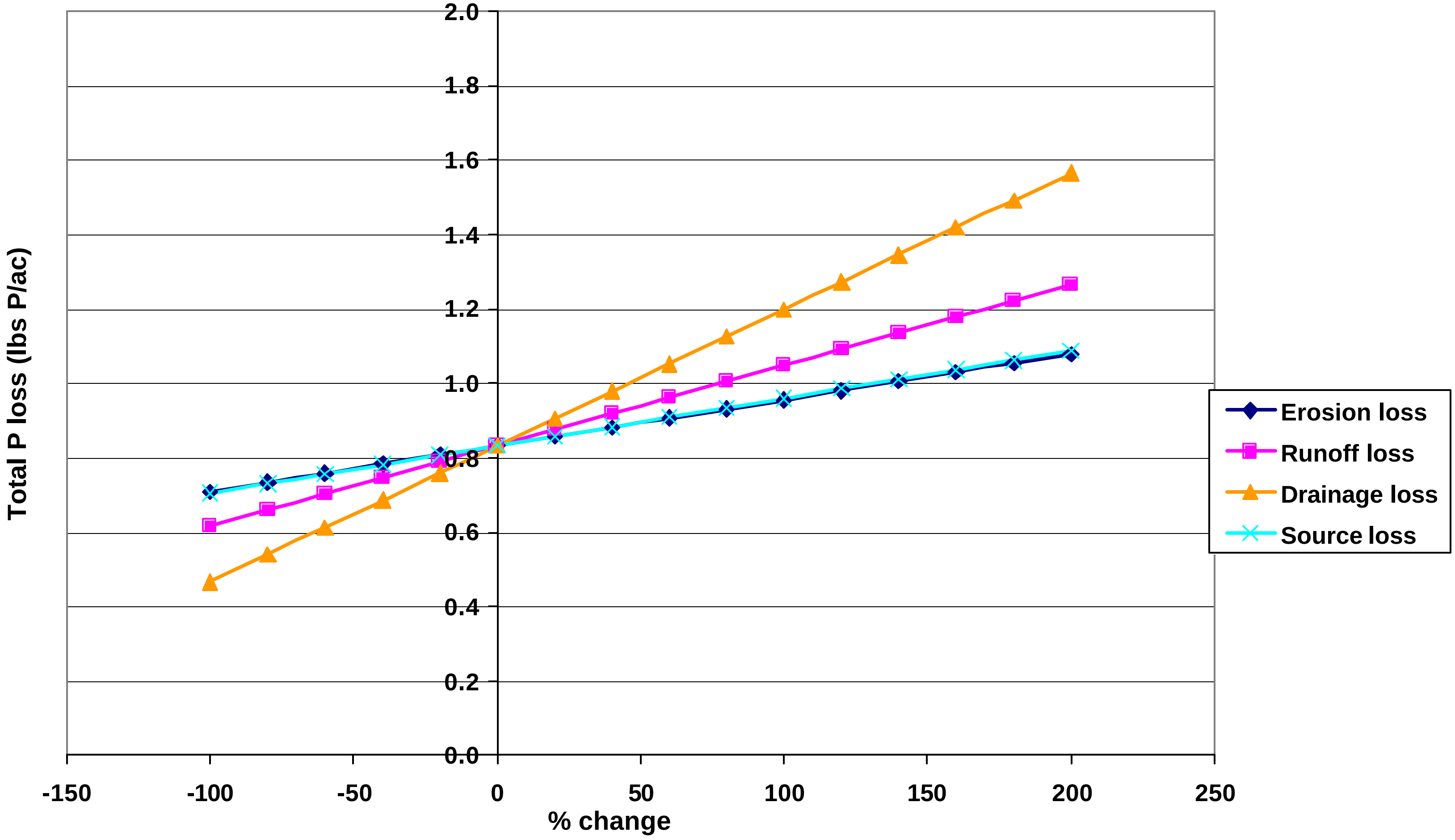

Figure 2b. Sensitivity of P loss predictions to changes in the four PLAT P loss pathways under baseline conditions for the natural drainage scenario.

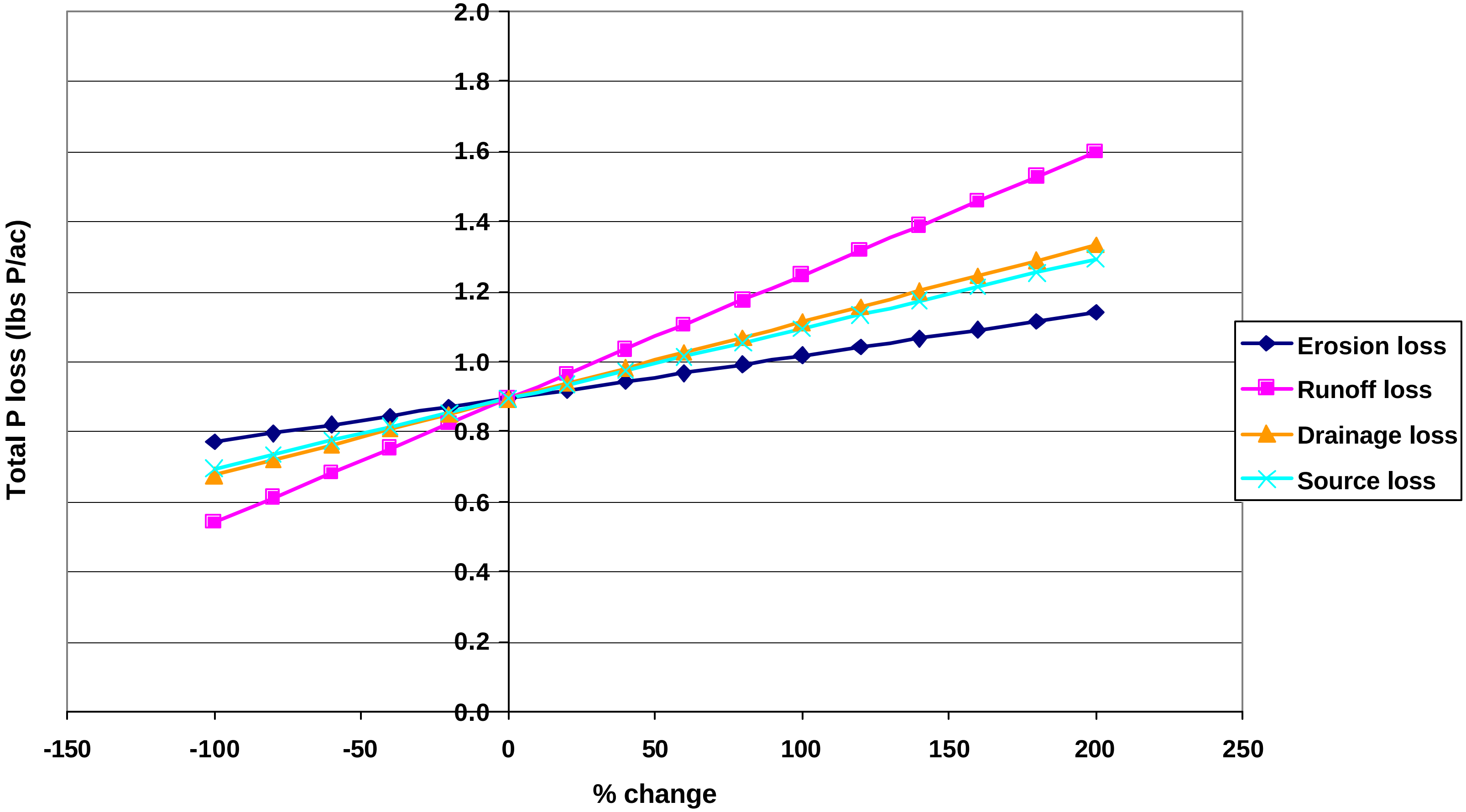

Figure 2c. Sensitivity of P loss predictions to changes in the four PLAT P loss pathways under baseline conditions for the artificial drainage scenario.

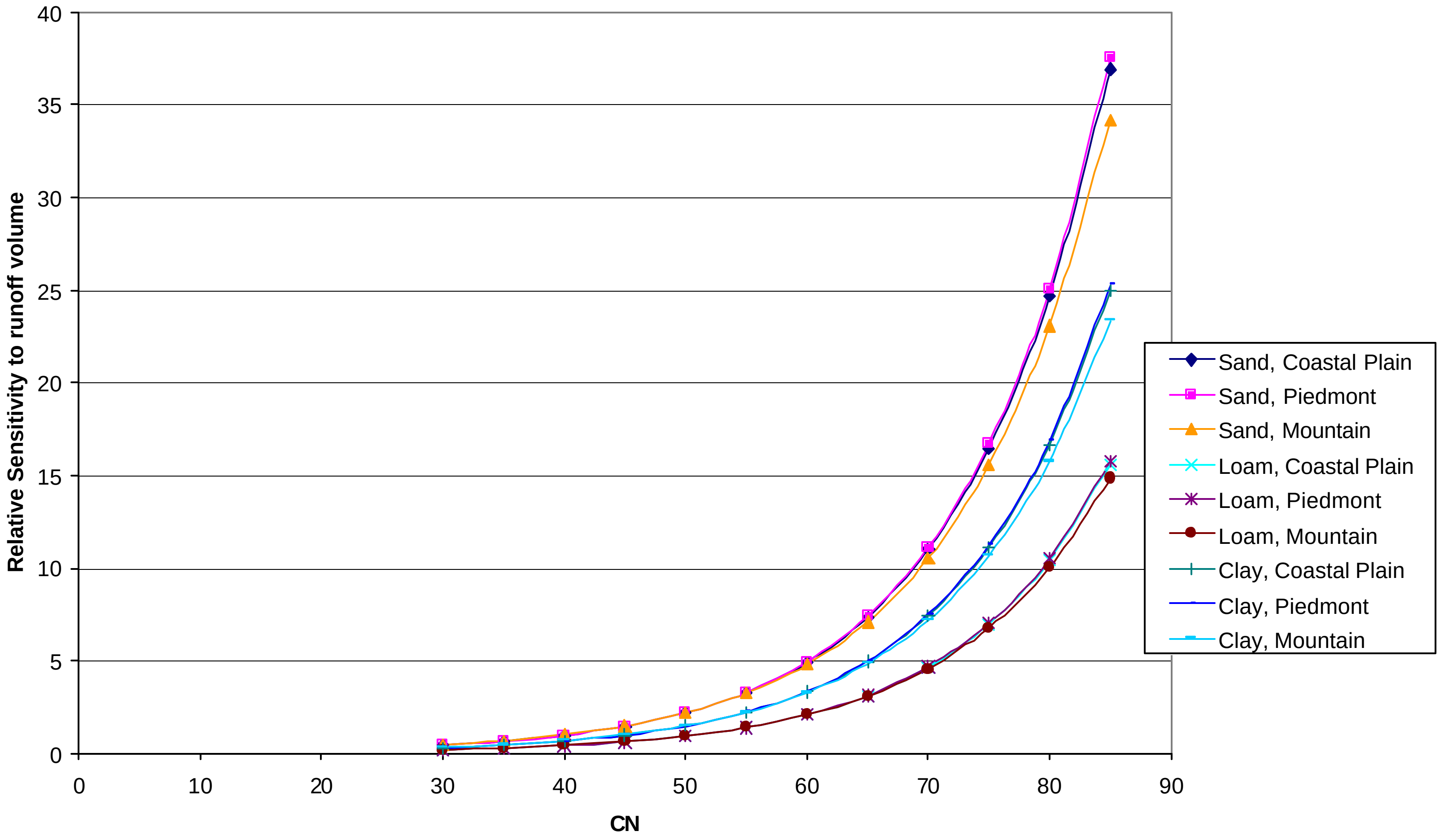

Figure 3. Changes in relative sensitivity to runoff volume with curve number for various soil types and regions.

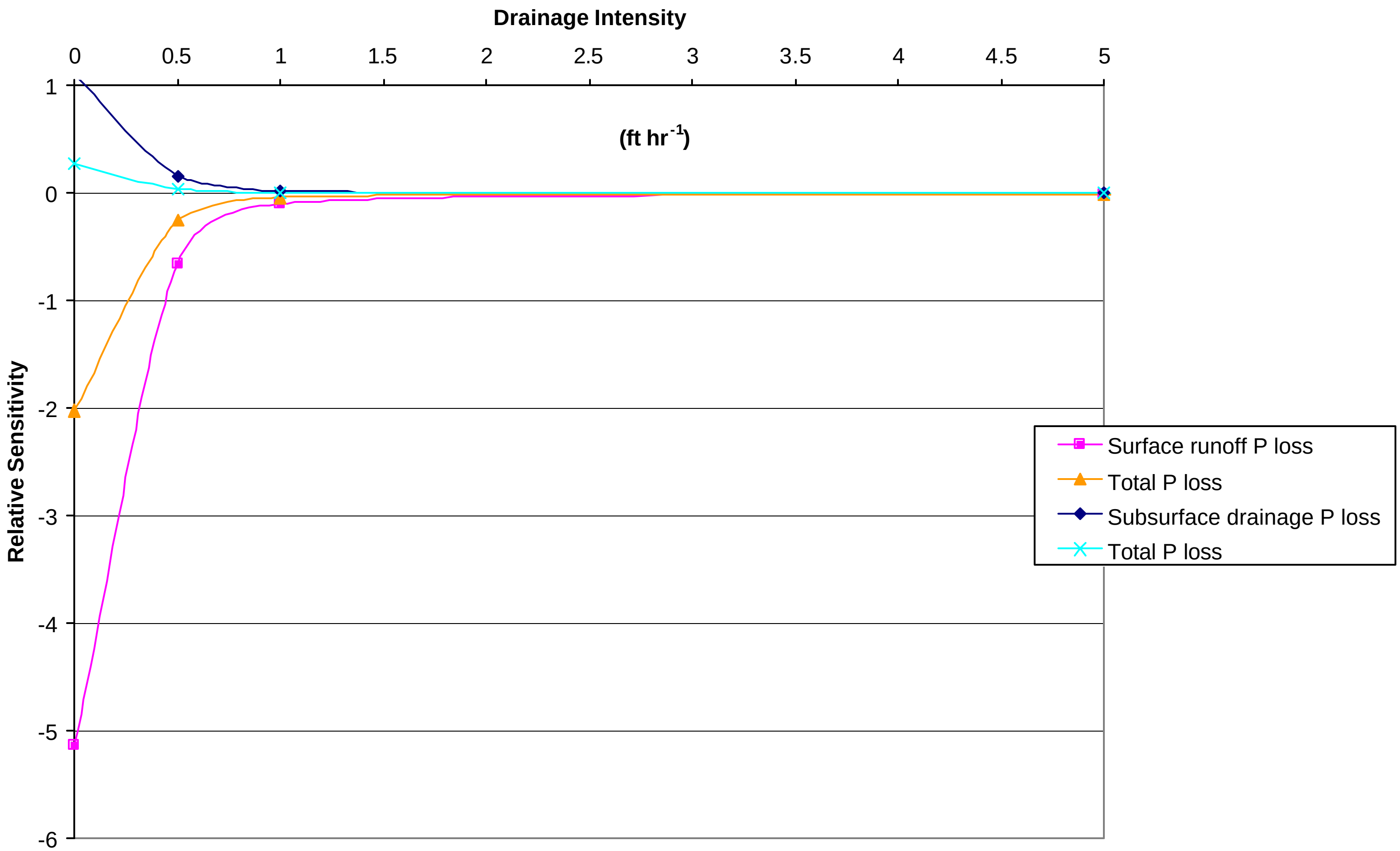

Figure 4a. Changes in relative sensitivity of various objective functions with drainage intensity.

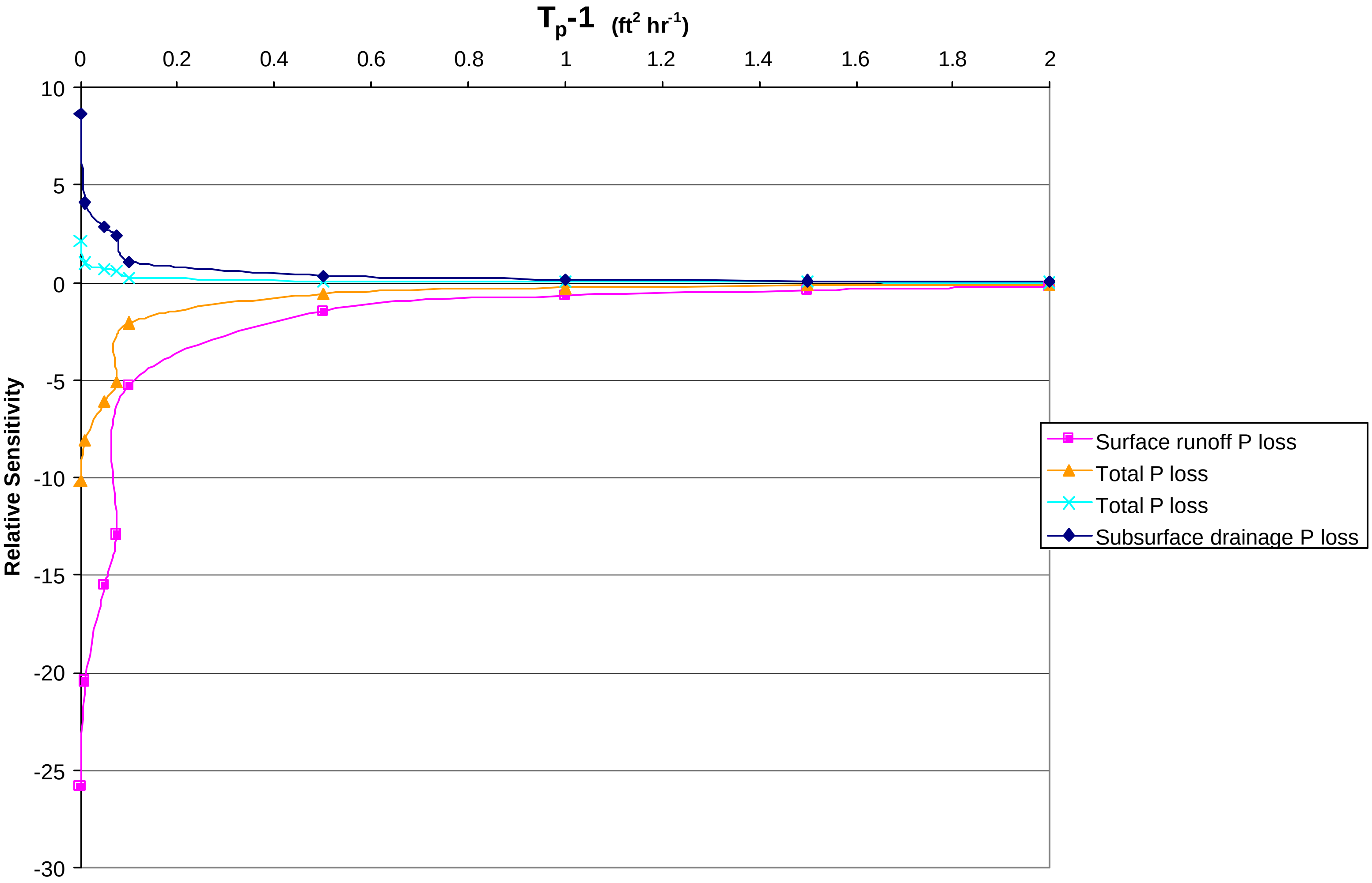

Figure 4b. Changes in relative sensitivity of various objective functions with transmissivity (T_p-1).

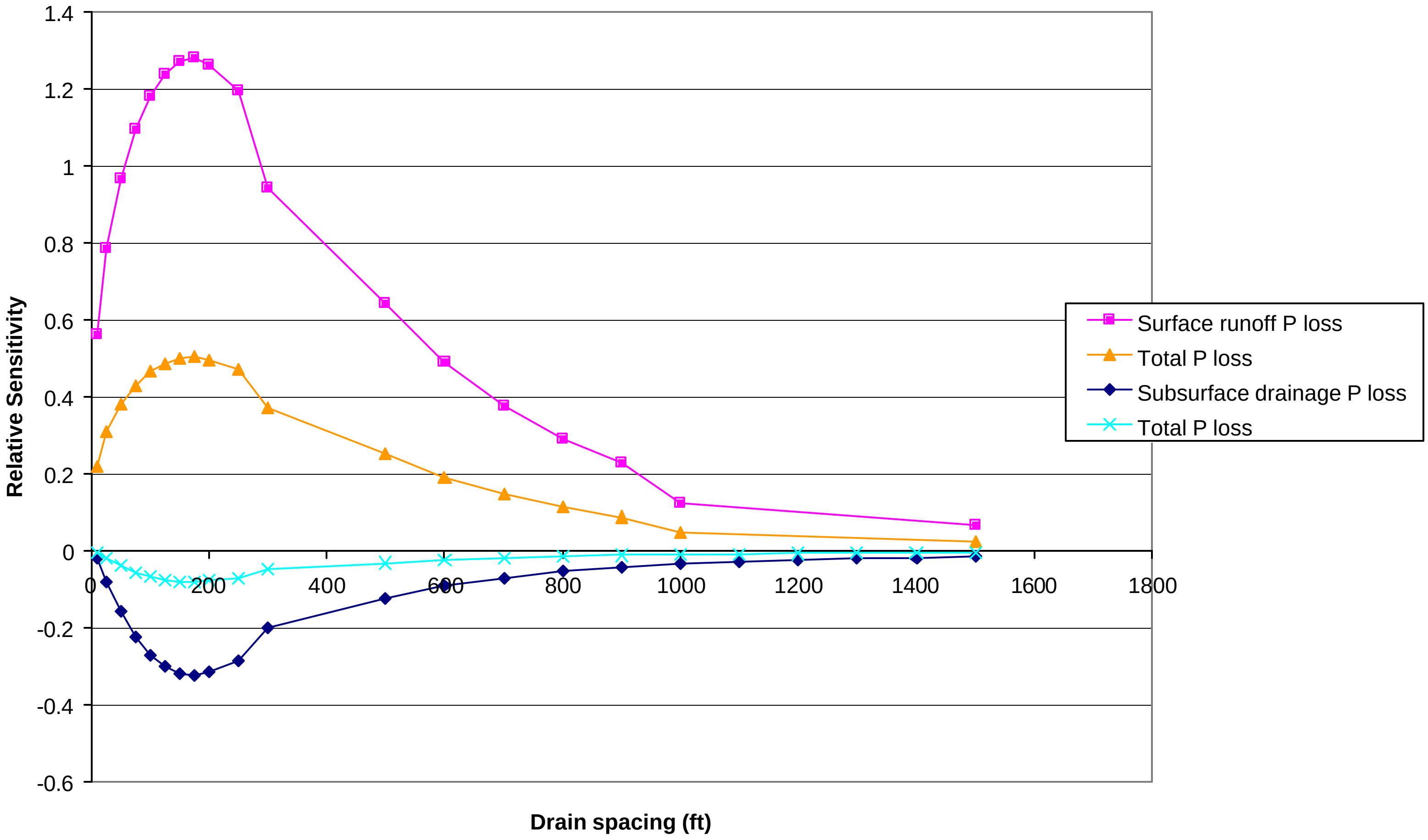

Figure 4c. Changes in relative sensitivity of various objective functions with drain spacing.

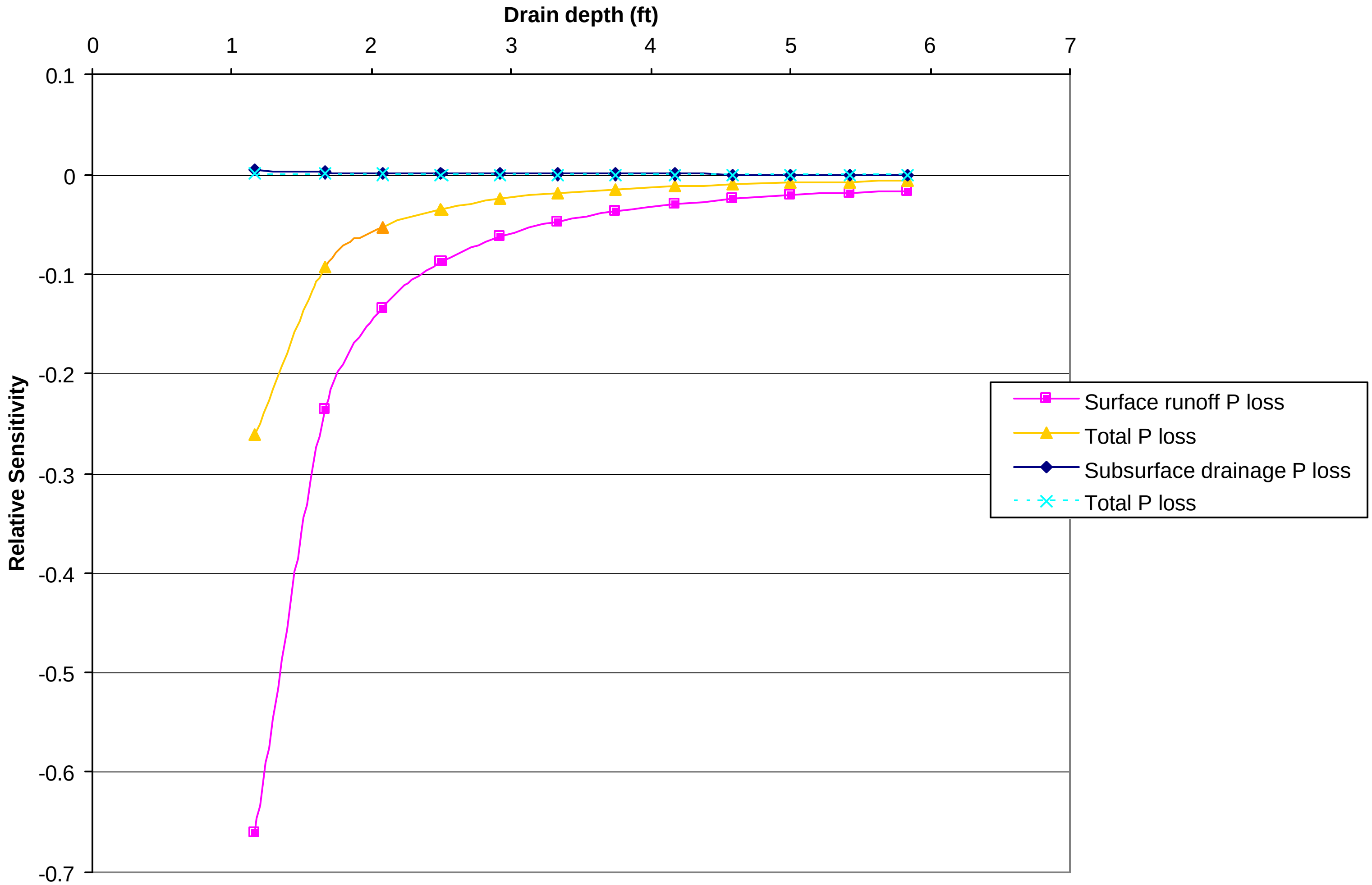

Figure 4d. Changes in relative sensitivity of various objective functions with drain depth.

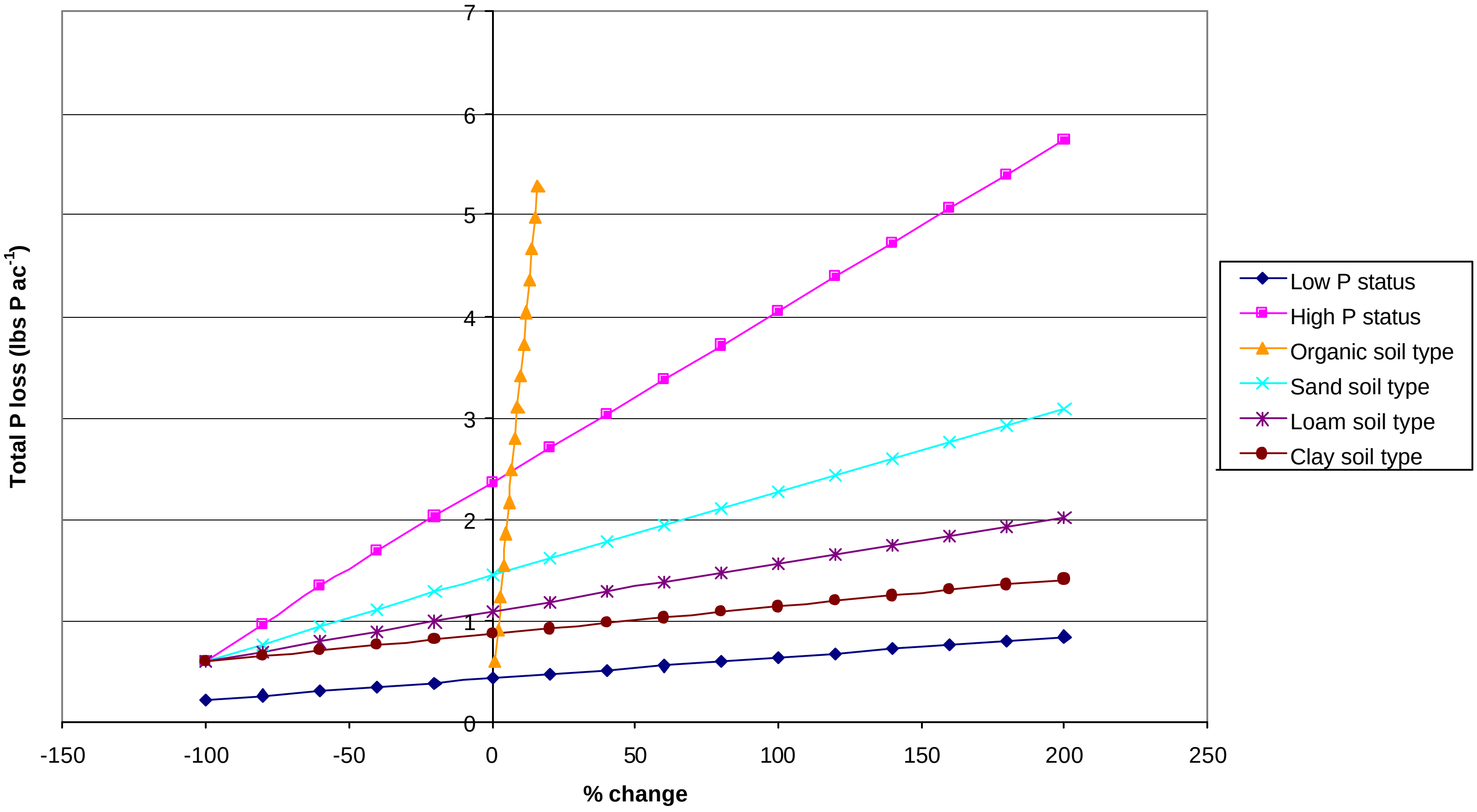

Table 5. Sensitivity of P loss predictions to changes in STP for different soil status or soil types.

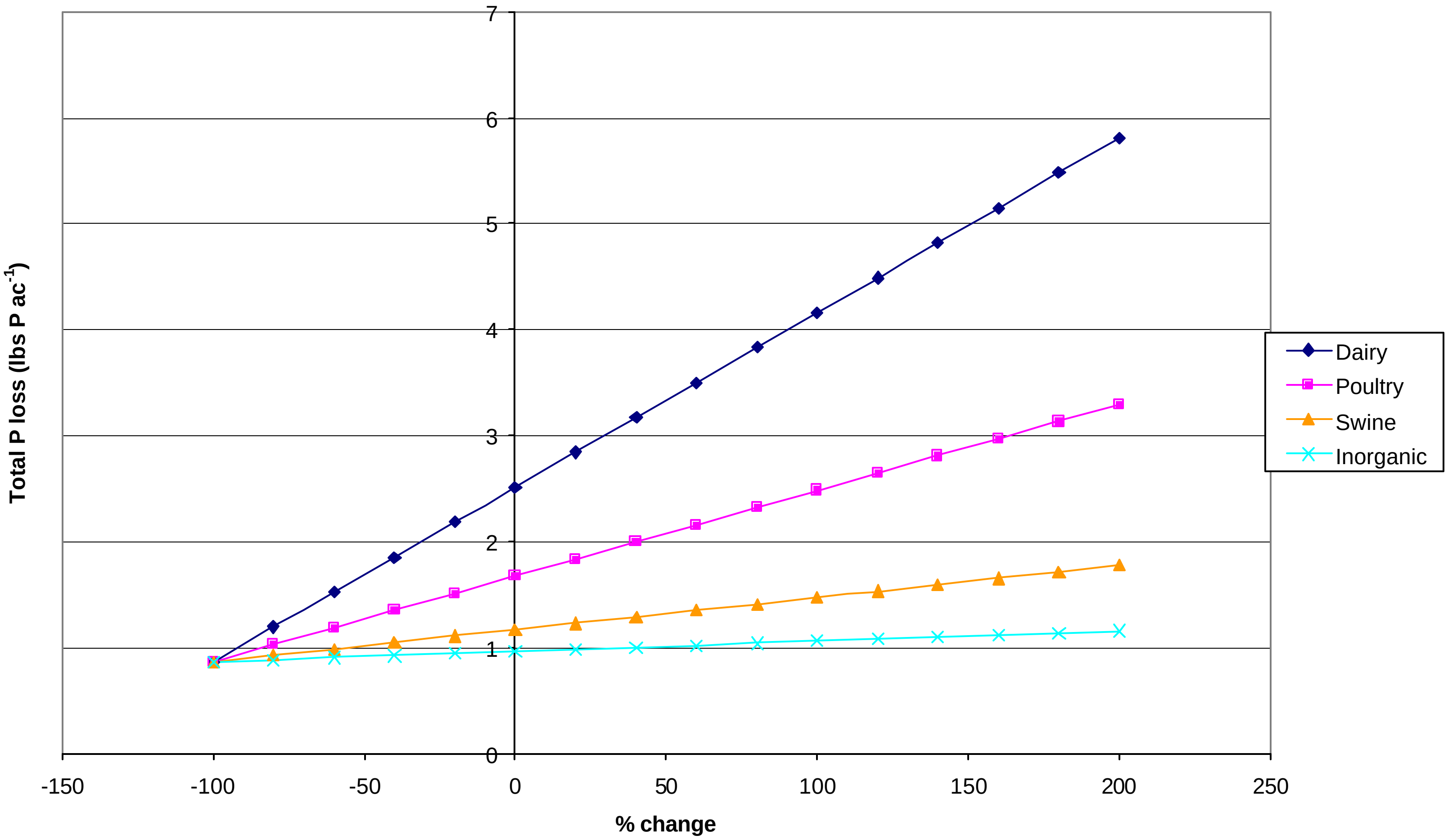

Table 6a. Sensitivity of P loss predictions to changes in source available P for different P amendment types.

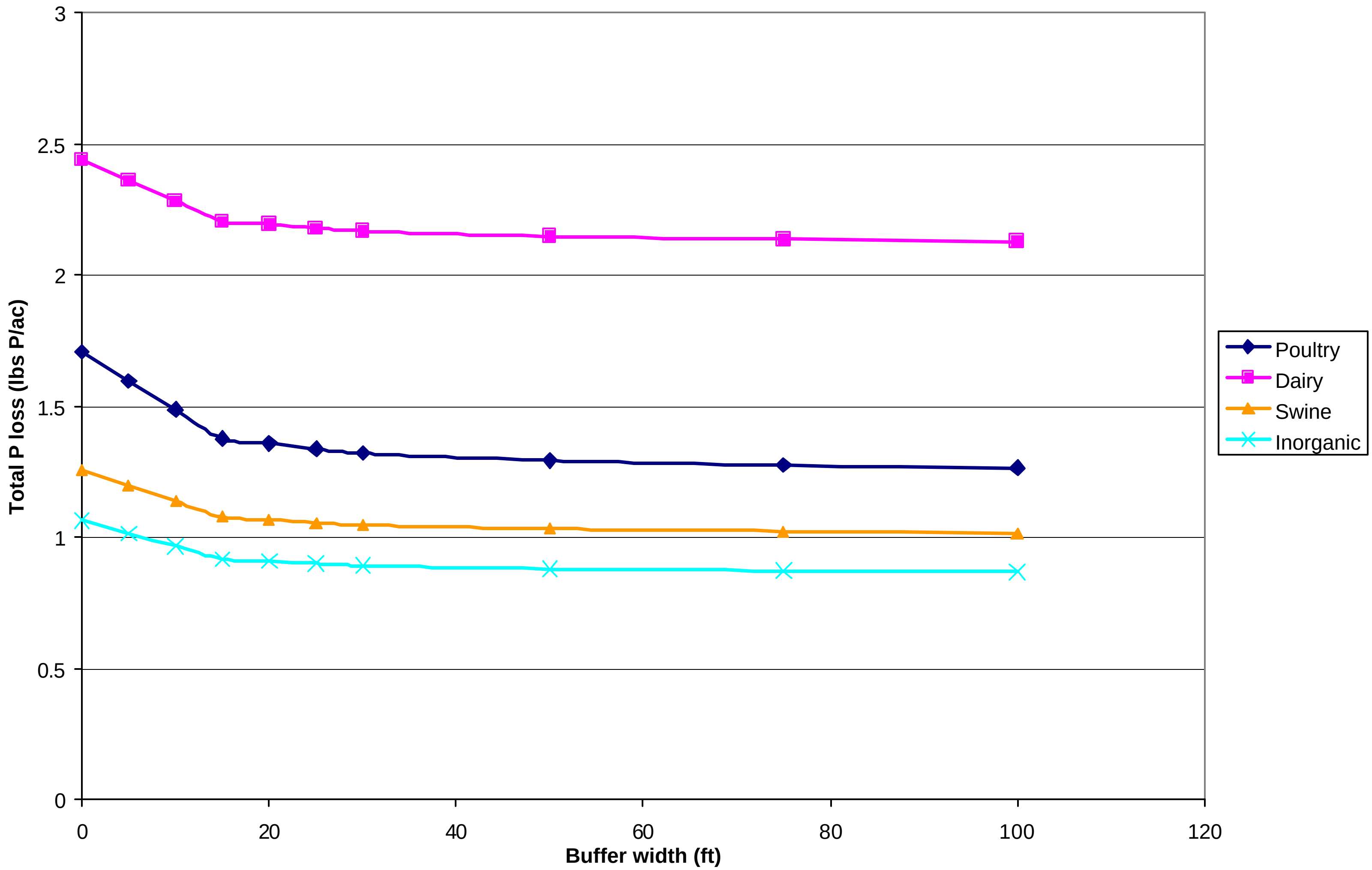

Figure 6b. Comparisons of predicted P loss with changes in vegetative buffer width on sites receiving different P amendment types.

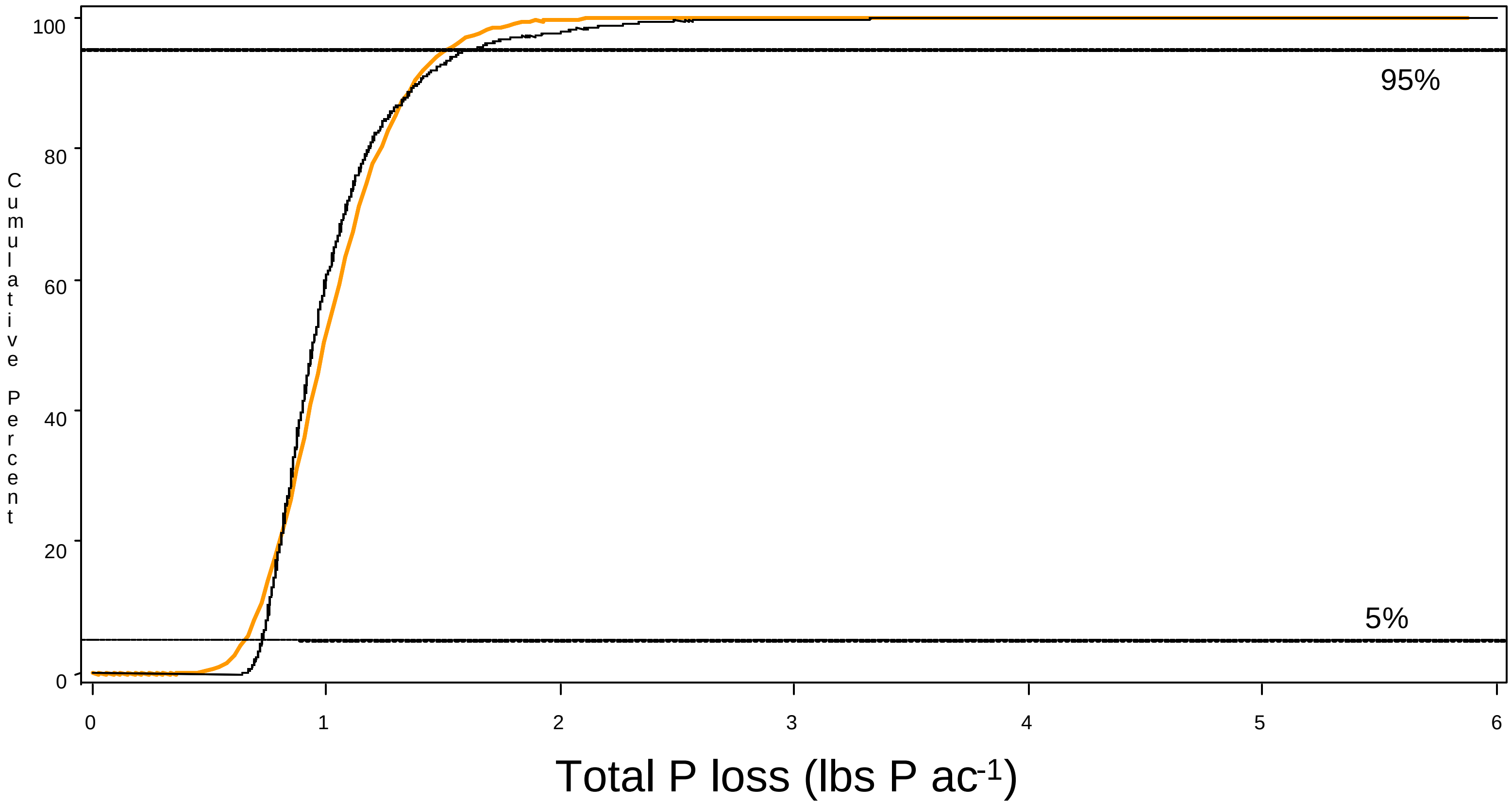

Figure 7. Cumulative probability density function for predicted P loss from naturally drained sites with runoff volume as the simulated input. Darkened line represents fitted lognormal curve.

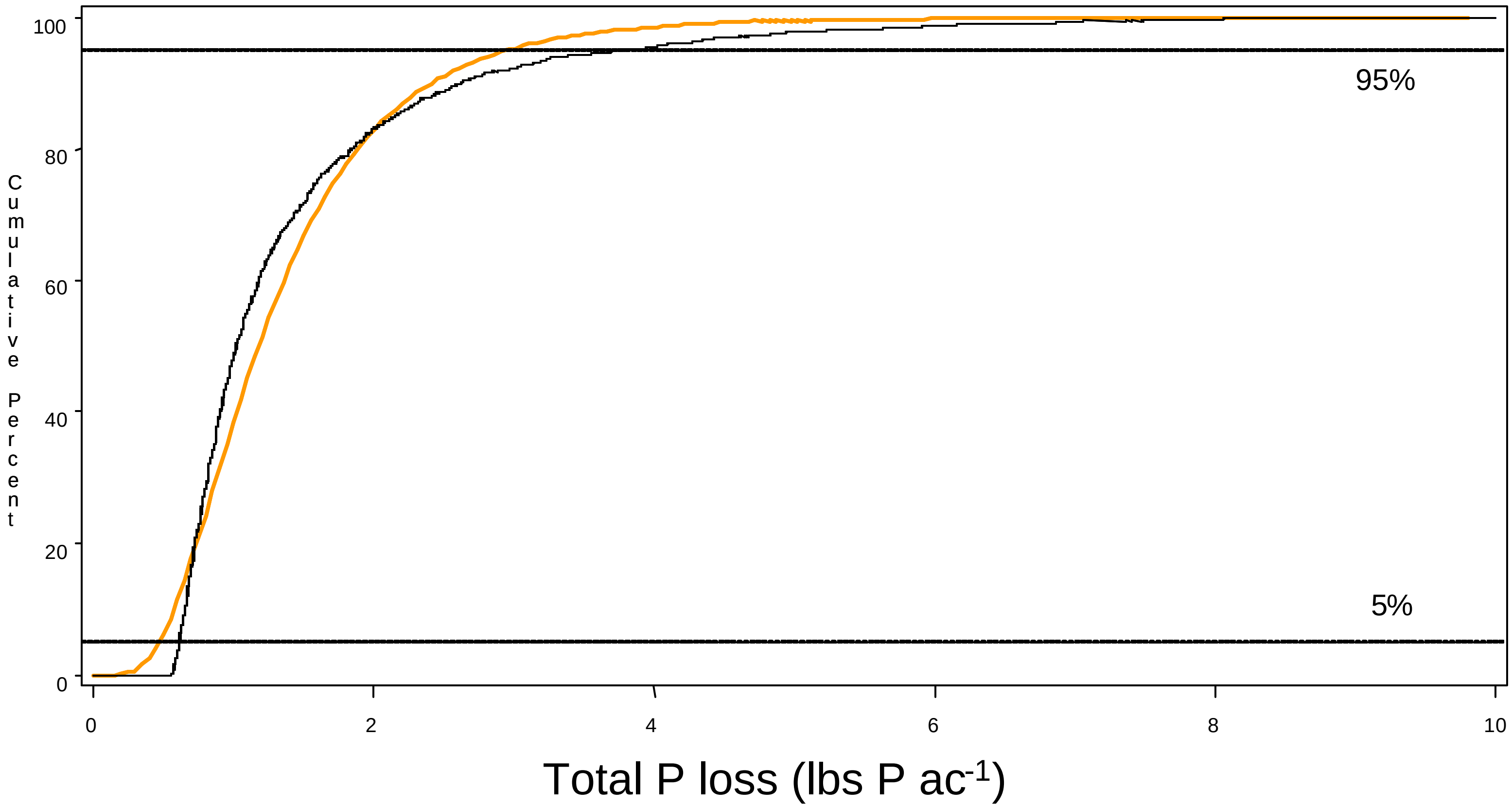

Figure 8. Cumulative probability density function for predicted P loss from artificially drained sites with runoff volume as the simulated input. Darkened line represents fitted lognormal curve.

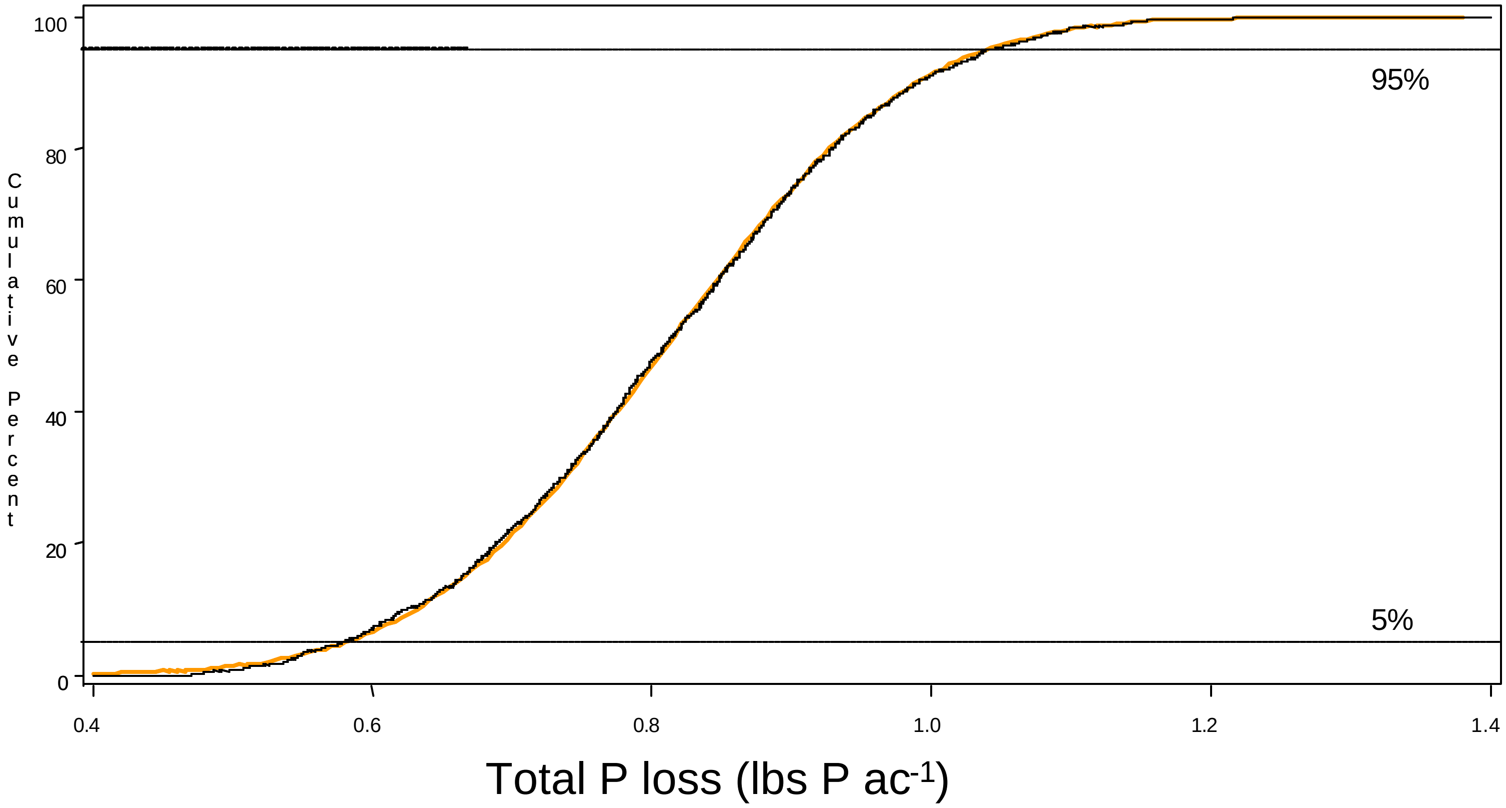

Figure 9. Cumulative probability density function for predicted P loss from naturally drained sites with subsurface drainage volume as the simulated input. Darkened line represents fitted normal curve

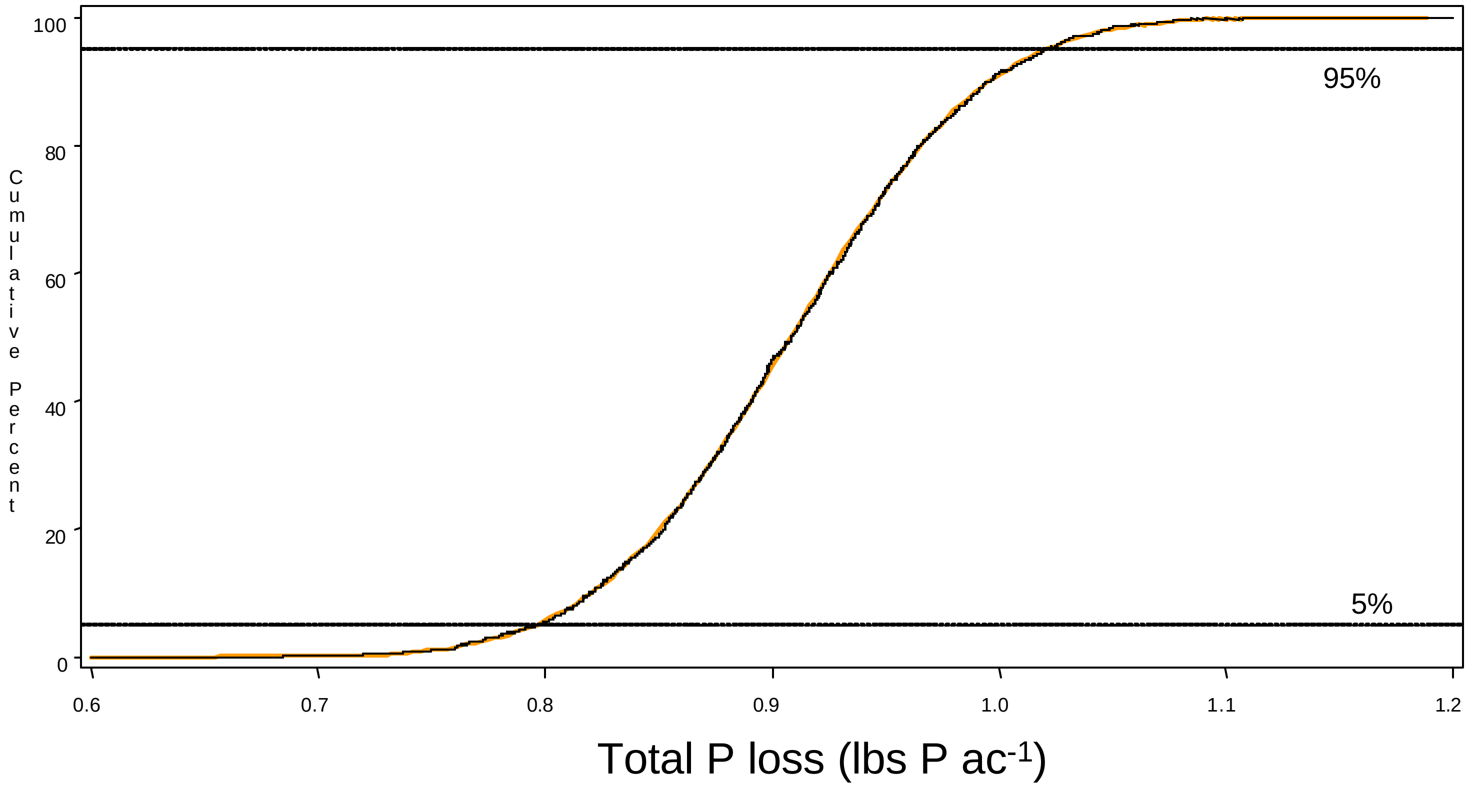

Figure 10. Cumulative probability density function for predicted P loss from artificially drained sites with subsurface drainage volume as the simulated input. Darkened line represents fitted normal curve.

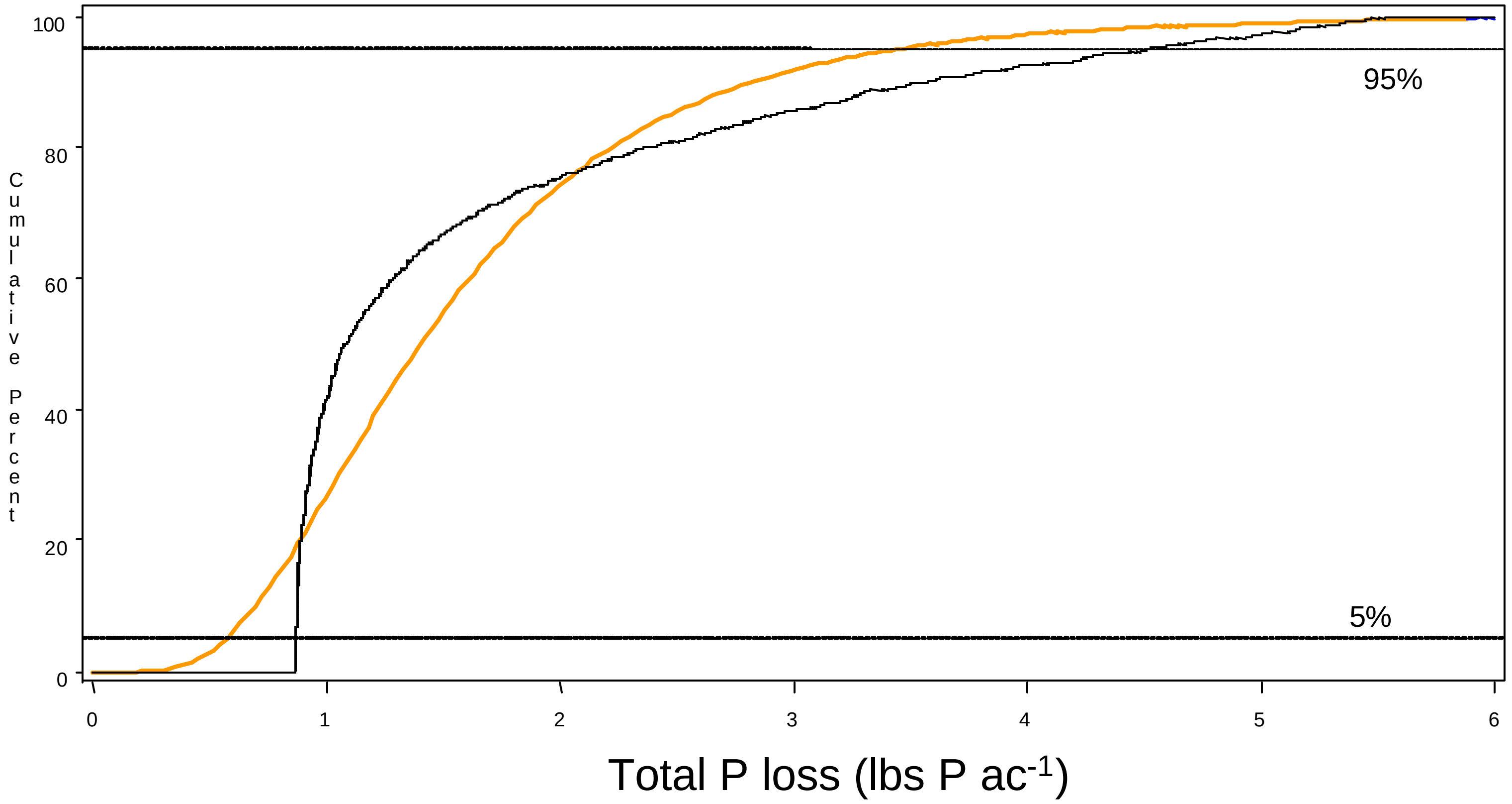

Figure 11. Cumulative probability density function for predicted P loss from sites receiving dairy waste with available P as the simulated input. Darkened line represents fitted lognormal curve.

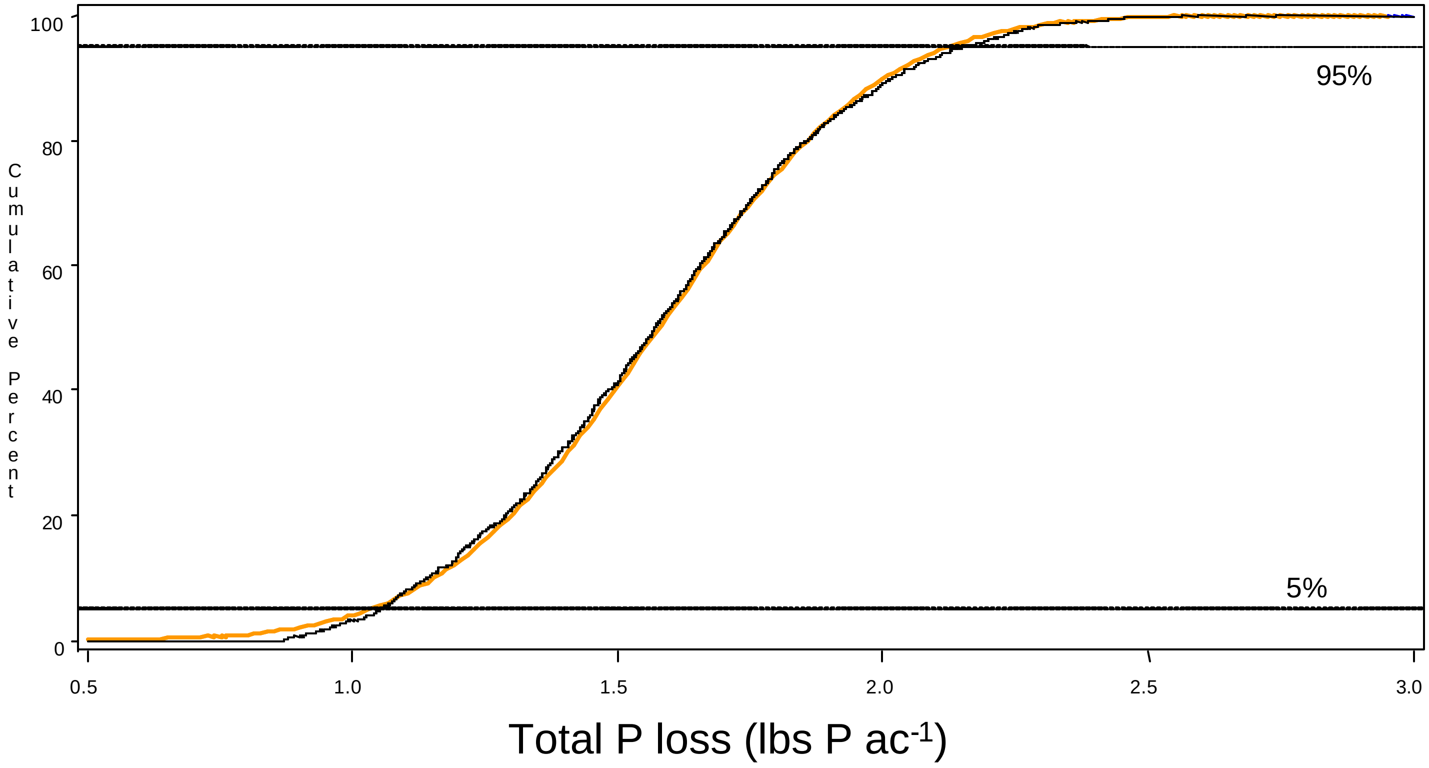

Figure 12. Cumulative probability density function for predicted P loss from sites receiving poultry waste with available P as the simulated input. Darkened line represents fitted normal curve.

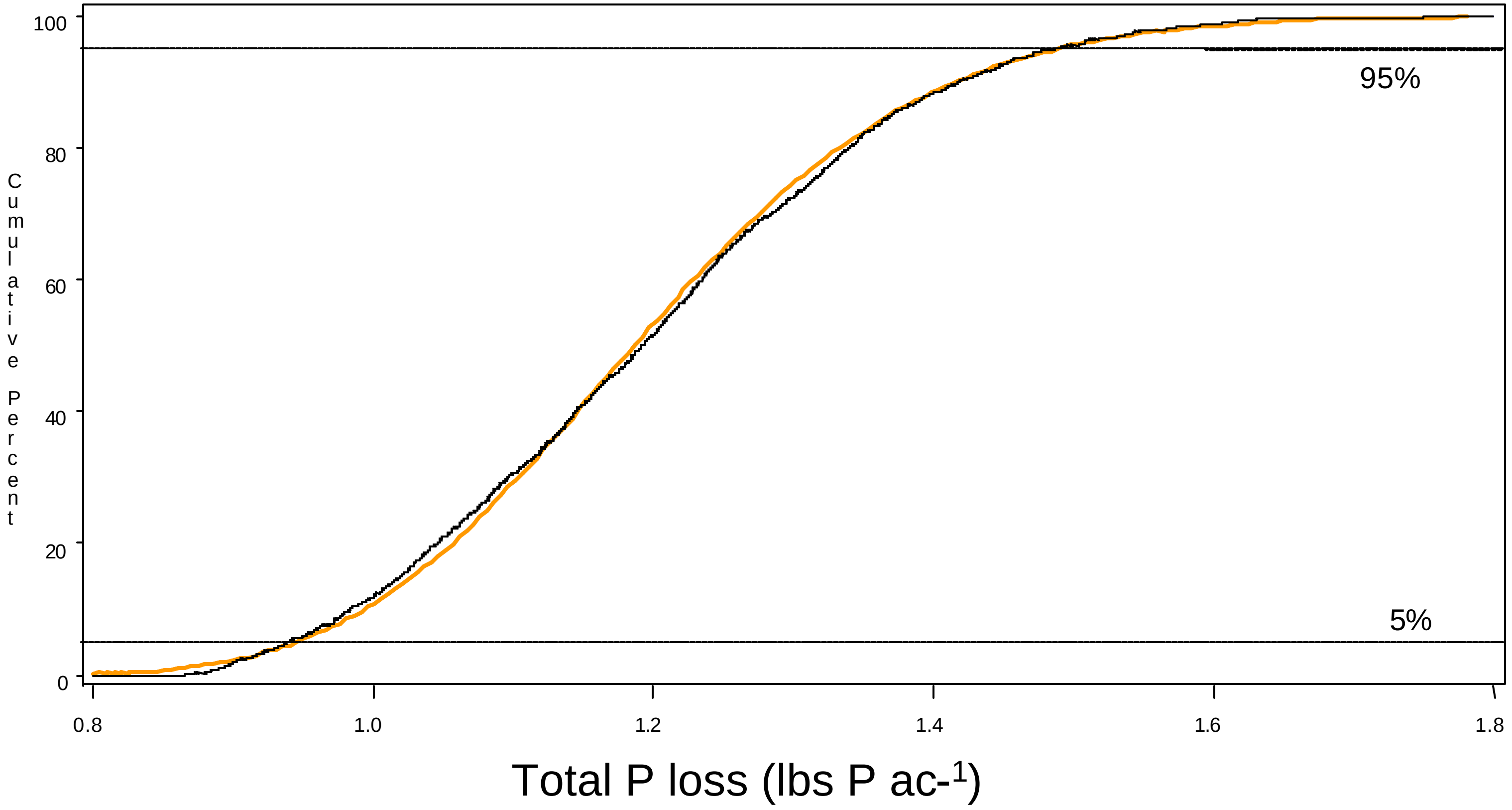

Figure 13. Cumulative probability density function for predicted P loss from sites receiving swine waste with available P as the simulated input. Darkened line represents fitted normal curve.

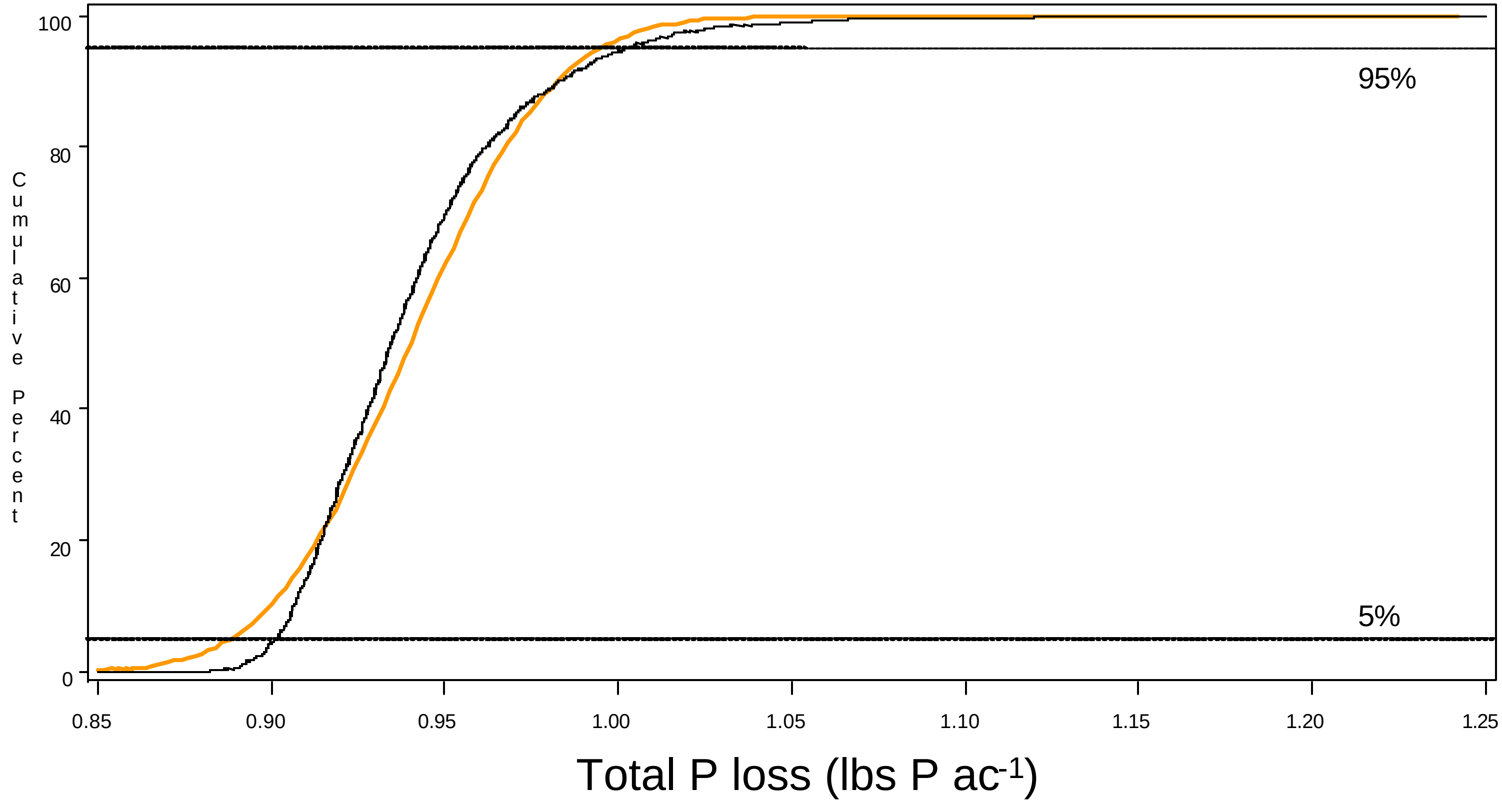

Figure 14. Cumulative probability density function for predicted P loss from sites receiving inorganic fertilizer with available P as the simulated input. Darkened line represents fitted lognormal curve.

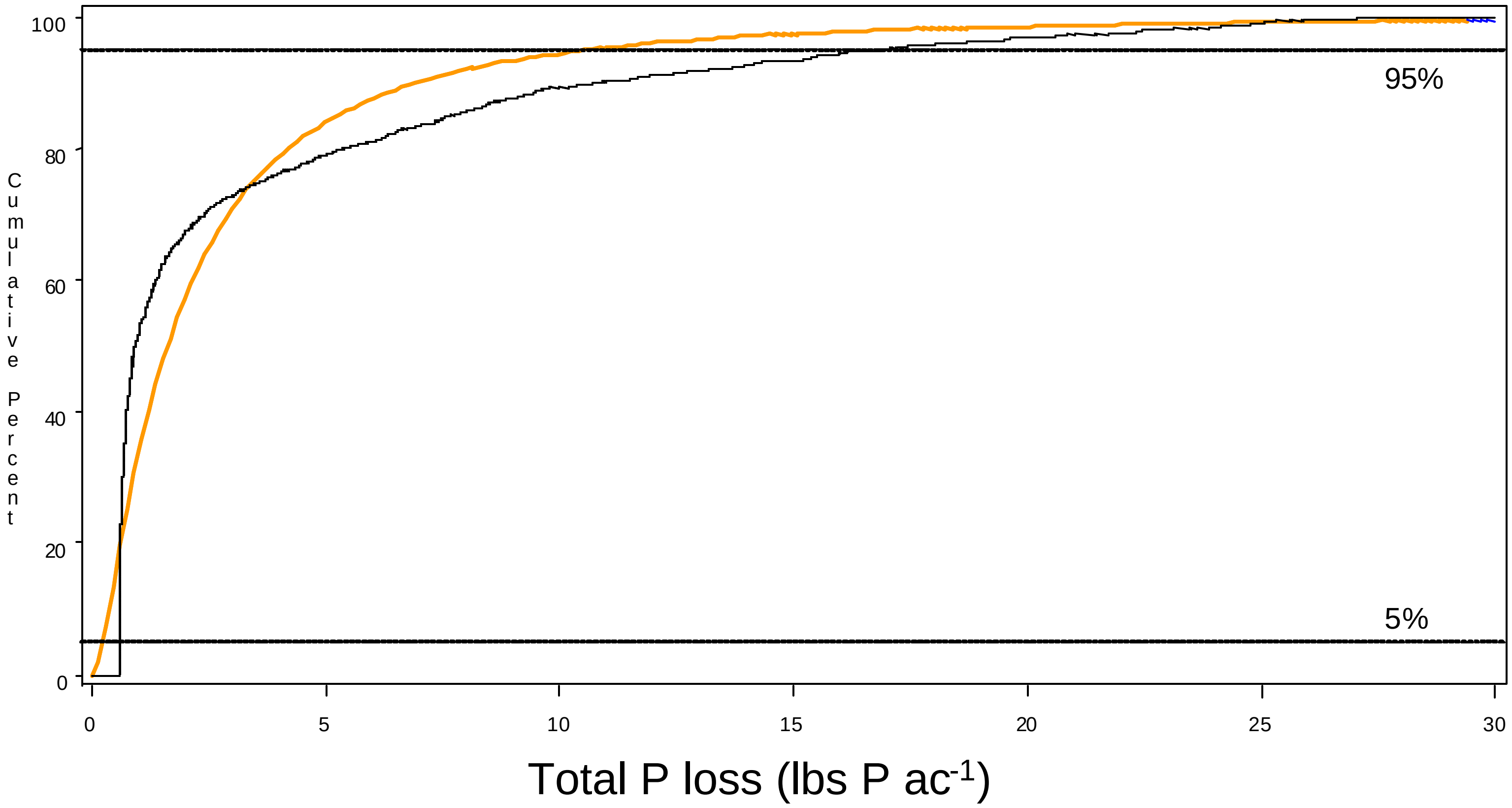

Figure 15. Cumulative probability density function for predicted P loss from organic soil types with STP as the simulated input. Darkened line represents fitted lognormal curve.

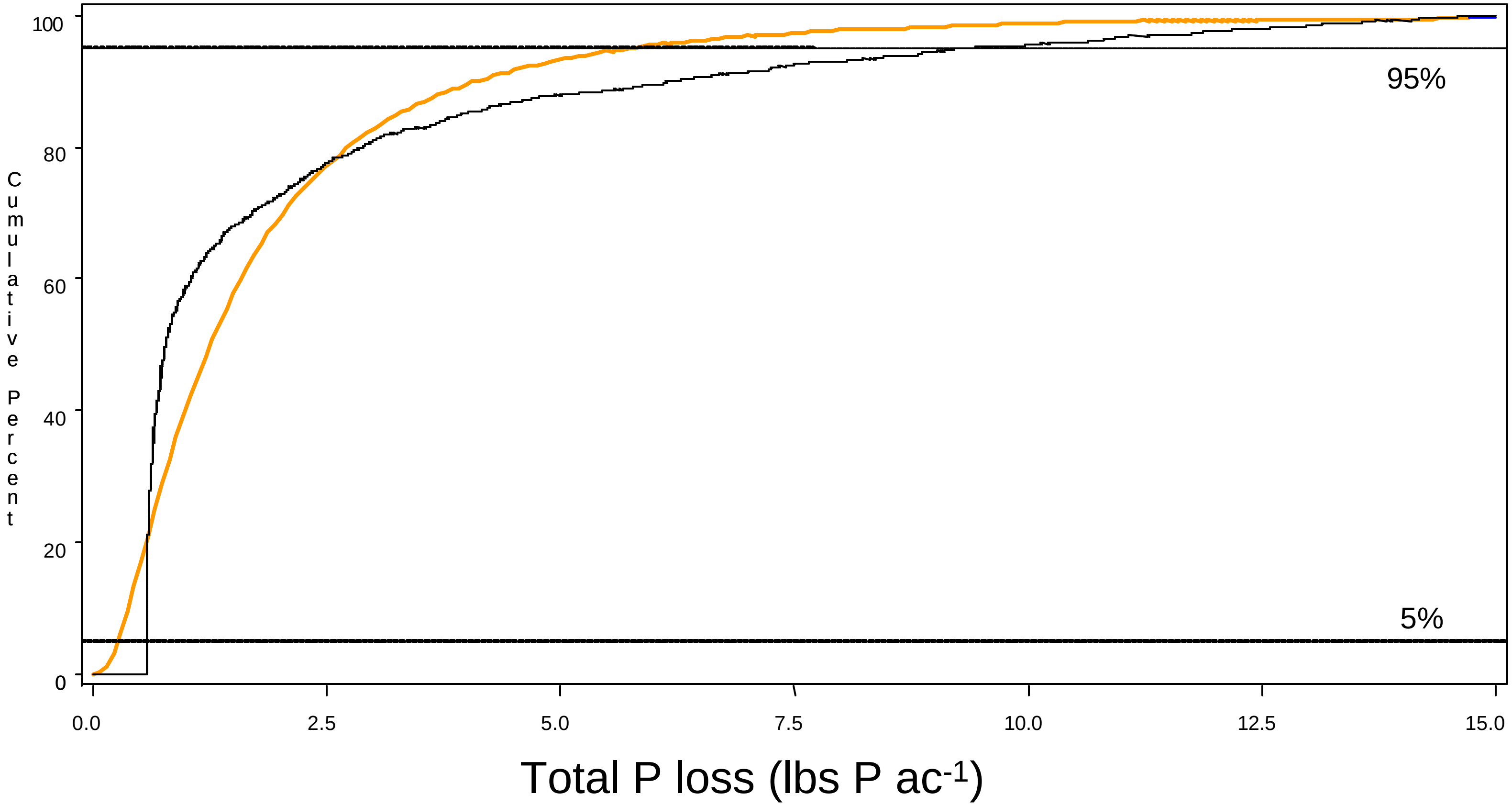

Figure 16. Cumulative probability density function for predicted P loss from sand soil types with STP as the simulated input. Darkened line represents fitted lognormal curve.

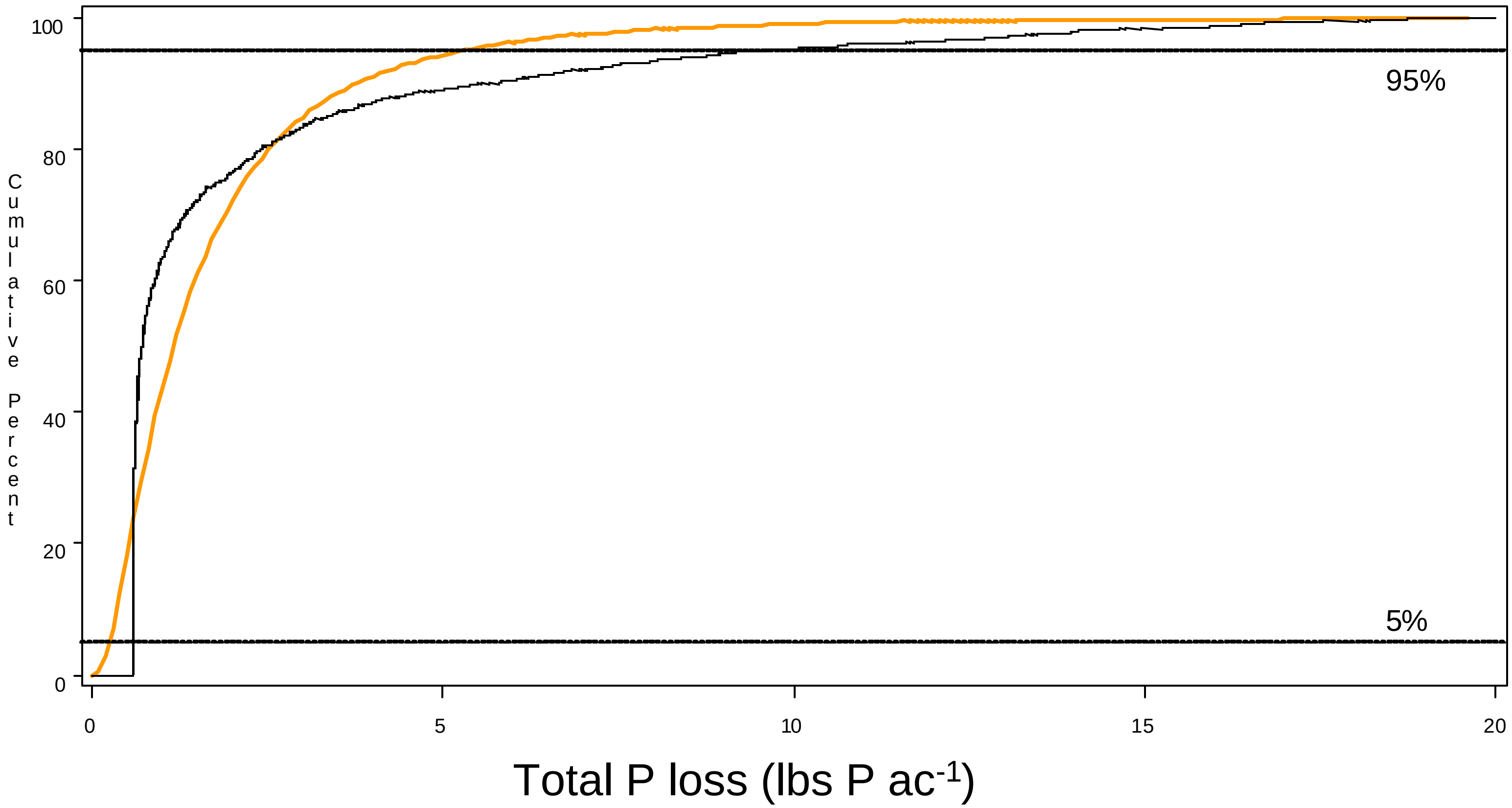

Figure 17. Cumulative probability density function for predicted P loss from loam soil types with STP as the simulated input. Darkened line represents fitted lognormal curve.

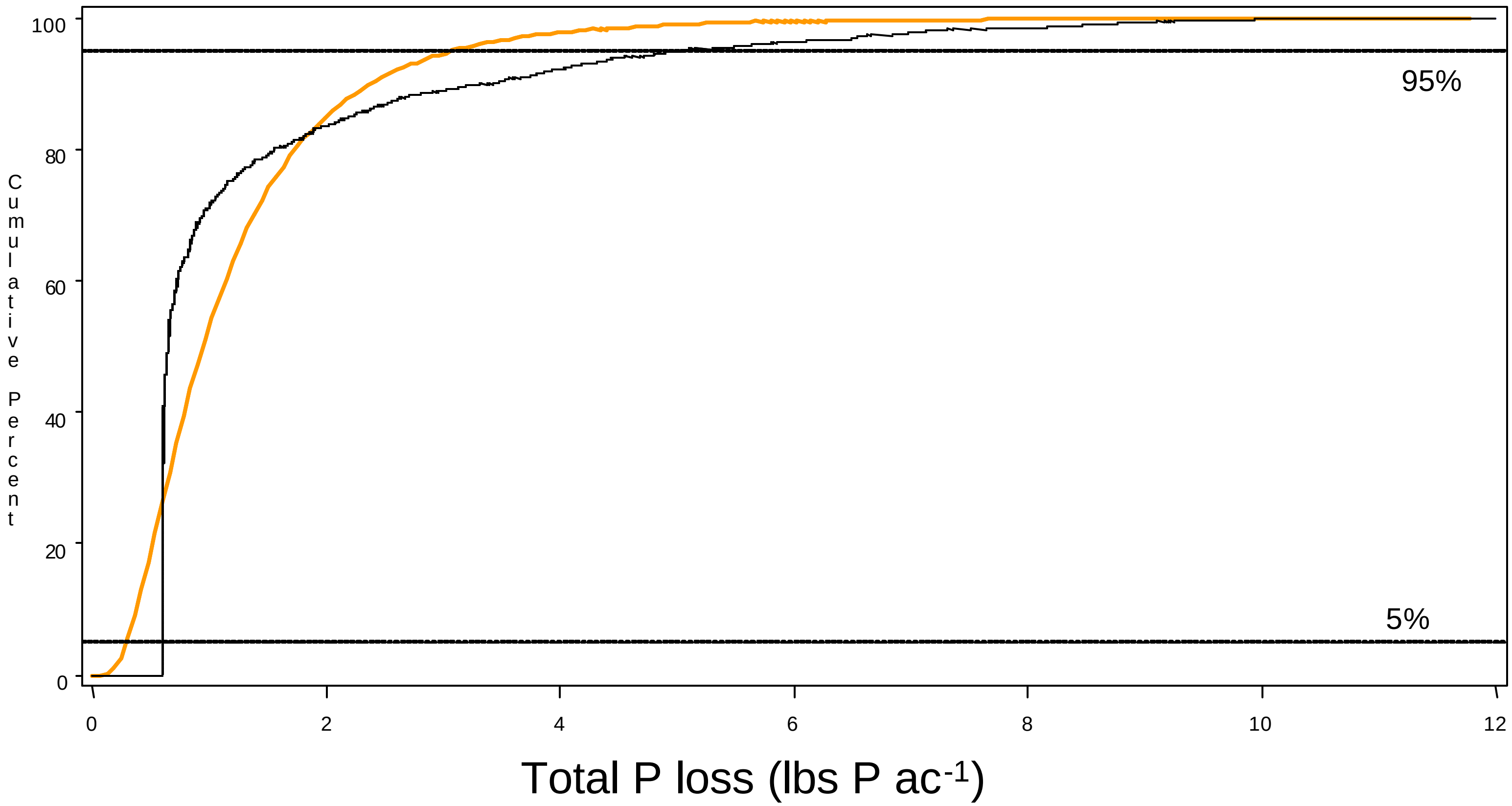

Figure 18. Cumulative probability density function for predicted P loss from clay soil types with STP as the simulated input. Darkened line represents fitted lognormal curve.

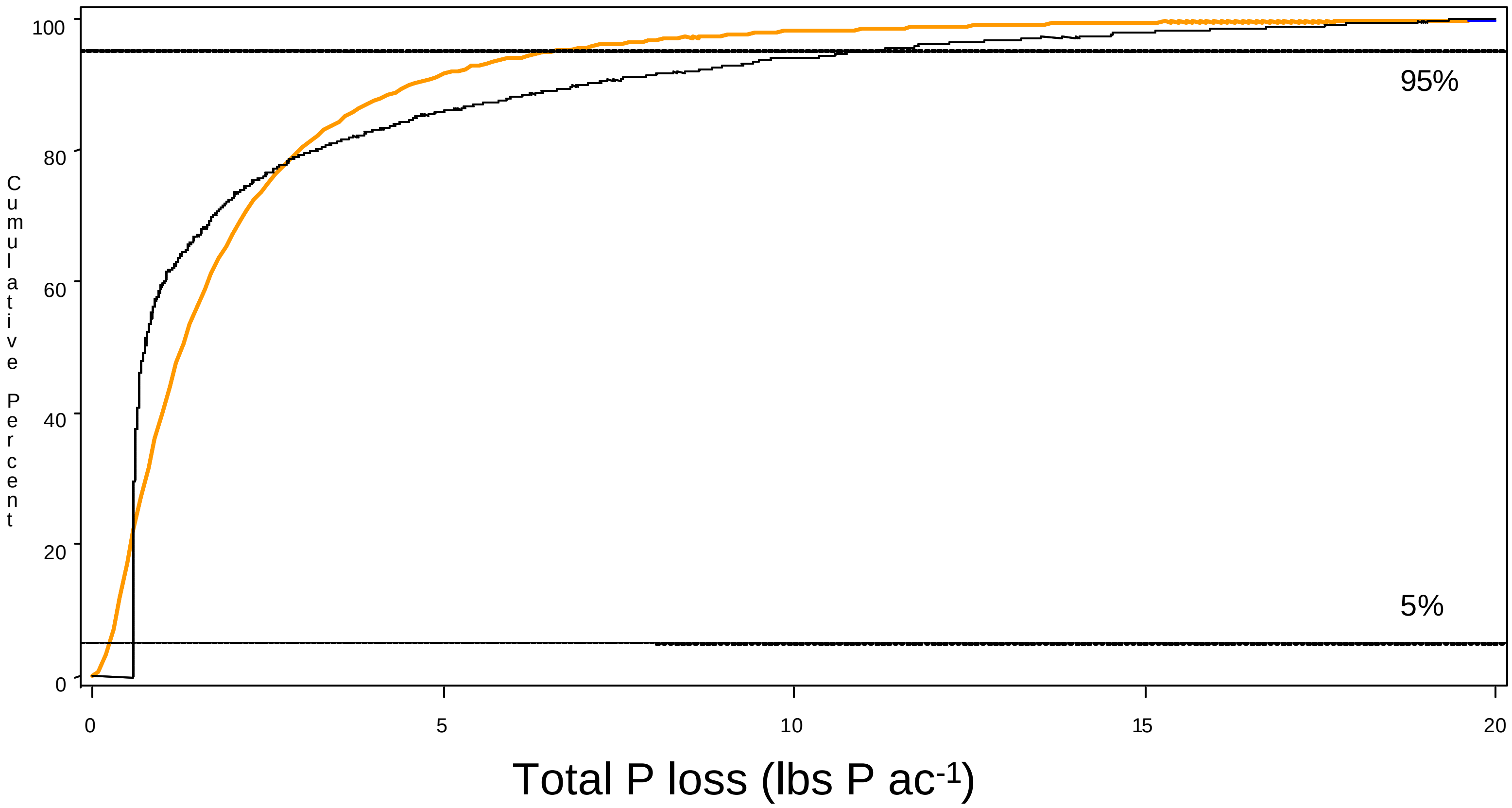

Figure 19. Cumulative probability density function for predicted P loss from high P status soils with STP as the simulated input. Darkened line represents fitted lognormal curve.

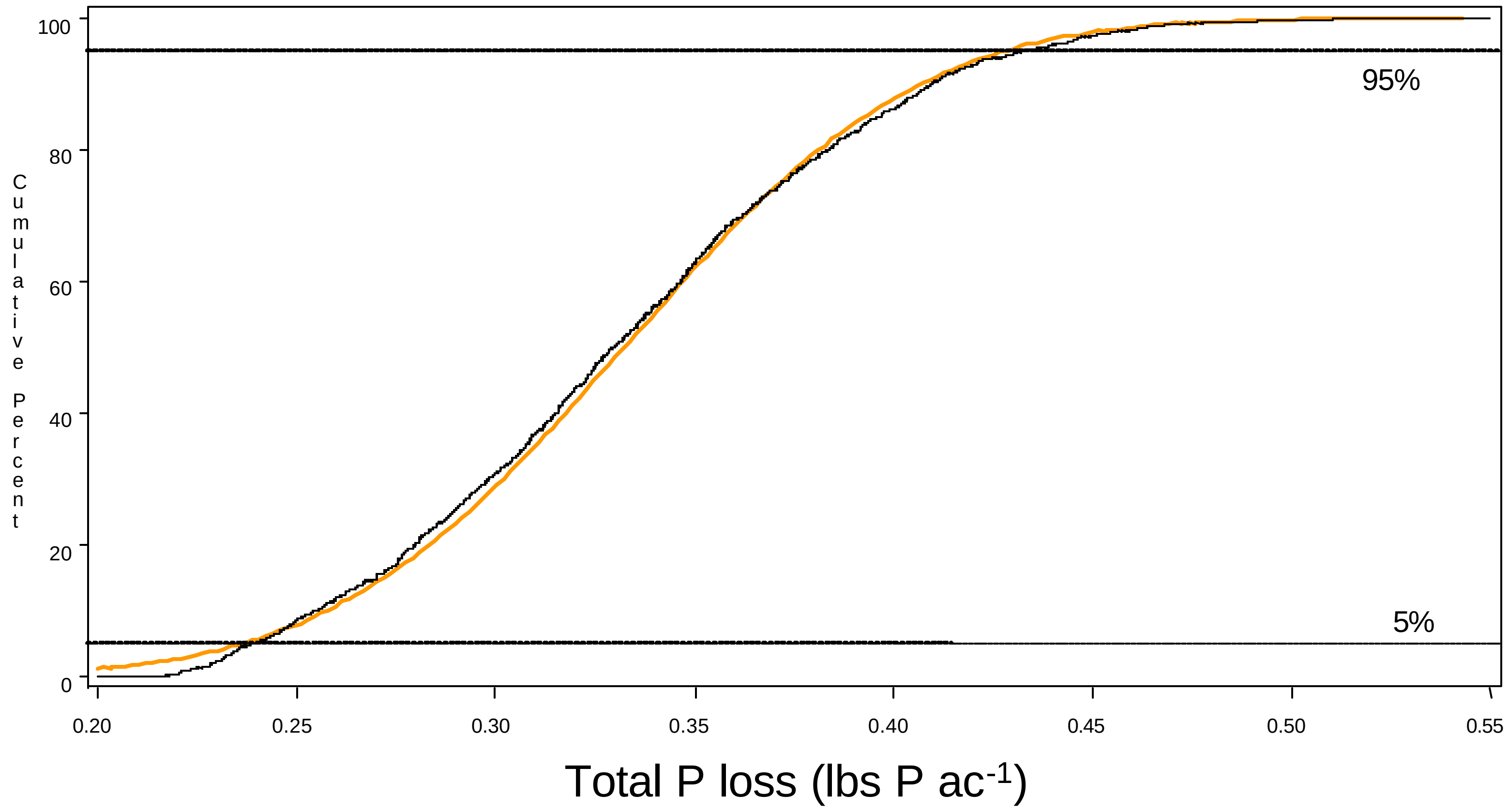

Figure 20. Cumulative probability density function for predicted P loss from low P status soils with STP as the simulated input. Darkened line represents fitted normal curve.