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Dependence of Runoff Phosphorus on Extractable Soil Phosphorus.

ABSTRACT

The sustainable management of fertilizer and manure P to minimize freshwater eutrophication requires identification of soil P levels that exceed crop P requirements and have the potential for P enrichment of runoff. Although several states have established such P levels, insufficient data are available to theoretically justify them. Thus, this study investigates the relationship between the concentrations of P in runoff and in soil. Surface samples (0-10 cm) of 10 Oklahoma soils were packed in 0.15 m² boxes, incubated for 7 d with poultry litter (0-20 Mg ha⁻¹) to obtain a range in Mehlich-3 P contents (7-360 mg kg⁻¹), and received five 30-min rainfalls applied at 1-h intervals. The concentration of dissolved, bioavailable, and particulate P in runoff was related to the Mehlich-3 P content of surface soil (0-1 cm), with regression slopes ranging from 2.0 to 7.2, increasing as soil P sorption maxima increased (r² > 0.95). Two soils of 200 mg kg⁻¹ Mehlich-3 P supported a dissolved P concentration in runoff of 280 µg L⁻¹ (San Saba gray, fine, montmorillonitic, thermic Udic Pellustert) and 1360 µg L⁻¹ (Stilger silt loam, fine, mixed, thermic Aquic Paleudalf). Thus, relationships between runoff and soil P will have to be soil specific for use in management recommendations. A single linear relationship described the dependence of dissolved (r² = 0.86) and bioavailable P (r² = 0.85) on soil P sorption saturation. The added complexity of the P saturation approach may limit its application; however, the approach integrates the effect of soil type with soil P content to better estimate the potential for P loss in runoff than soil P alone.

The loss of bioavailable P as dissolved and particulate P in runoff is of increasing concern in certain regions of the U.S. due to the resulting impact on surface water quality (Sharpley et al., 1994b; USEPA, 1994). Much of this concern centers around the accumulation of P at the surface of soils receiving continual long-term applications of fertilizer and/or manures (Daniel et al., 1994; Sharpley et al., 1994b). In many areas of intensive confined livestock production, amounts of P produced in manure often exceed local crop requirements (Kingerly et al., 1994; Sharpley et al., 1993). This can lead to disposal rather than use of manure, with a subsequent build-up of soil P above levels sufficient for optimal crop yields.

As the loss of particulate P in runoff is a function of erosion, conservation control measures that decrease soil loss can cause concomitant decreases in particulate P. Dissolved P loss, however, is more difficult to reduce and control measures are largely limited to preventing soil P accumulation to environmentally sensitive levels. Environmentally sensitive or threshold soil P levels are those above which the potential for P loss in runoff exceeds any crop production concerns. Quantification of these levels is critical if action agencies are to develop nutrient management guidelines for water quality as well as crop production goals.

Several states have identified soil test P levels (e.g., Bray-I, Mehlich-3), above which fertilizer and manure P applications should be reduced or eliminated to minimize the potential for P loss in runoff (Gartley and Sims, 1994). Establishing these soil test P levels is often a highly controversial process. Due to a lack of field data these levels have been based more on intuition than fact. Thus, the validity of such threshold soil P levels can be contested (Wells, 1994). Also, the economic implications of establishing lower soil test P levels that may limit manure applications can be significant. Clearly, more information is needed on the relationship between soil and runoff P, so that defensible fertilizer and manure management recommendations can be developed.

Several studies have shown the dissolved P concentration of runoff is dependent upon surface (0-5 cm) soil Bray-I P content (Olness et al., 1975; Romkens and Nelson, 1974; Schreiber, 1988; Sharpley et al., 1986). For these studies, the dissolved P-soil P relationship varied with management. On average, regression slopes for grass (6.0) were lower than for cultivated land (10.5). Even so, regression slopes varied from 4.1 to 7.0 for grass and from 8.3 to 12.5 for cultivated land showing the risk of using a single or average relationship (Sharpley et al., 1996). Thus, several soil and land management factors will influence the relationship between dissolved
P in runoff and soil-P. These factors will affect the volume of runoff as well as release of soil P to runoff. Consequently, runoff-soil P relationships used to assess volume of runoff as well as release of soil P to runoff be soil and site specific. The potential for P loss in runoff will probably have to be better determined by more reliable and realistic estimators of soil P available to be released to runoff (Sharpley, 1993).

In the Netherlands, the national strategy is to limit P entry into both surface and ground water. One of the fundamental mechanisms for accomplishing this goal is the identification of a soil P saturation level above which P application should not exceed crop removal rates (Breeuwsma and Silva, 1992). The P saturation approach is based on the fact that soil P desorption increases as sorbed P accumulates in soil following P additions, thereby leading to increased P loss via runoff or leaching. To determine the critical level of soil P accumulation, Dutch regulators have set a limit of 100 µg L⁻¹ as dissolved P tolerated in ground water at a given soil depth (mean highest water table level; Breeuwsma and Silva, 1992). The degree of accumulation is related to the degree of P sorption saturation (%):

$$\text{P sorption saturation} = \frac{\text{Extractable soil P}}{\text{P sorption maximum}} \times 100 \quad [1]$$

where the units of extractable soil P and P sorption maximum are unit mass of dissolved P for a given mass of soil (mg kg⁻¹). In the Netherlands, extractable soil P and P sorption maximum are determined from the oxalate-extractable P, Al, and Fe content of noncalcareous soils (Breeuwsma and Silva; 1992). A P sorption saturation of 25% has been established as the critical value, above which the potential for P movement in surface and ground waters becomes unacceptable. The approach, however, has not been tested in the USA.

Clearly, the dependence of runoff dissolved P on soil P is a function of soil physical and chemical properties influencing soil P release to runoff. Thus, more information is needed on the effect of soil type on the relationship between runoff dissolved P and soil P content in order to develop defensible environmental P-management recommendations.

This study investigates the dependence of P loss in runoff on soil P for 10 Oklahoma soils and determines relationships for use in developing recommendations for environmentally sustainable P management. Poultry litter was added at several rates to establish a range of P contents of these soils and traditional agronomic (Mehlich-3 P) and environmental soil P tests (Fe-oxide impregnated paper strip P and P saturation) were used to describe P release to runoff. As rainfall intensity and soil slope and management were constant, the effect of soil type on the relationship between runoff and soil P can be quantified.

### MATERIALS AND METHODS

#### Experimental

Surface samples (0–10 cm) of 10 soils were collected from LeFlore and McCurtain counties in southeast Oklahoma (Table 1). These soils are widespread in this region and are commonly under fescue receiving poultry litter annually (about 10 Mg ha⁻¹). Poultry litter (pine-bark shavings as bedding material) was collected from two broiler houses in McCurtain county on the day of clean-out. The samples were combined, thoroughly mixed and stored as collected at house moisture. Moisture content of the poultry litter, determined by gravimetric analysis (308 °K), was 9.8%. The litter was subsampled, air dried, ground to 0.15 mm, and stored in polyethylene bags at 277 K for P analysis.

Air-dried, sieved (2 mm) soil was packed in duplicate impermeable-bottomed boxes (1 m long, 0.15 m wide, and 0.15 m deep) to field bulk density (about 1.35 Mg m⁻³). Poultry litter was incorporated by mixing with 3-kg increments of soil during packing to a depth of 5 cm. Rates of litter application were 0, 2.5, 5, 7.5, 10, 15, and 20 Mg ha⁻¹, equivalent to 0, 62.5, 125, 187.5, 250, 375, and 500 mg P kg soil⁻¹.

The soils were slowly wetted by a drip system to saturation, with a 7-d incubation before application of rain by a capillary-tube type rainfall simulator (Munns and Huntington, 1976). Rain fell onto the boxes from a height of 2.5 m, to achieve terminal velocity, at an average intensity of 2.54 cm h⁻¹ for 30 min. This intensity has a 5-yr return frequency in southeast Oklahoma. Rains of 30-min duration were applied to each treatment at 1-d intervals for five consecutive days. Soil boxes were inclined to create a 4% slope.

Surface soil samples (0–1 cm) were taken from each box.

### Table 1: Physical and chemical properties of the selected soils.

<table>
<thead>
<tr>
<th>Soil</th>
<th>Classification</th>
<th>pH 6.5</th>
<th>Organic C (%)</th>
<th>Mehlich-3 P mg kg⁻¹</th>
<th>Strip P mg kg⁻¹</th>
<th>Total P mg kg⁻¹</th>
<th>P sorption maxima</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cahaba, vsl</td>
<td>Typic Hapludult</td>
<td>5.5</td>
<td>24.1</td>
<td>27</td>
<td>13</td>
<td>159</td>
<td>904</td>
</tr>
<tr>
<td>Captina, sl</td>
<td>Typic Fragiudult</td>
<td>5.0</td>
<td>15.4</td>
<td>39</td>
<td>18</td>
<td>273</td>
<td>782</td>
</tr>
<tr>
<td>Carnisaw, fsl</td>
<td>Typic Hapludult</td>
<td>5.6</td>
<td>26.1</td>
<td>21</td>
<td>18</td>
<td>206</td>
<td>364</td>
</tr>
<tr>
<td>Durant, l</td>
<td>Vertic Argiustoll</td>
<td>7.1</td>
<td>25.0</td>
<td>7</td>
<td>3</td>
<td>234</td>
<td>940</td>
</tr>
<tr>
<td>Muskogee, l</td>
<td>Aquic Paleudalf</td>
<td>5.6</td>
<td>33.6</td>
<td>23</td>
<td>11</td>
<td>571</td>
<td>1248</td>
</tr>
<tr>
<td>Rexor, sl</td>
<td>Ultic Hapludult</td>
<td>5.6</td>
<td>49.0</td>
<td>29</td>
<td>14</td>
<td>411</td>
<td>844</td>
</tr>
<tr>
<td>Ruston, fsl</td>
<td>Typic Paleudult</td>
<td>5.8</td>
<td>23.7</td>
<td>13</td>
<td>6</td>
<td>240</td>
<td>452</td>
</tr>
<tr>
<td>San Saba, c</td>
<td>Udic Pellustert</td>
<td>7.9</td>
<td>22.1</td>
<td>27</td>
<td>13</td>
<td>481</td>
<td>1510</td>
</tr>
<tr>
<td>Shermore, fsl</td>
<td>Typic Fragiudult</td>
<td>5.1</td>
<td>21.6</td>
<td>34</td>
<td>16</td>
<td>206</td>
<td>684</td>
</tr>
<tr>
<td>Stigler, fsl</td>
<td>Aquic Paleudalf</td>
<td>6.0</td>
<td>22.7</td>
<td>7</td>
<td>8</td>
<td>352</td>
<td>364</td>
</tr>
</tbody>
</table>

† vsl, fsl, sl, sI, l, and c represent very fine sandy loam, fine sandy loam, sandy loam, silt loam, loam, and clay, respectively.
prior to rainfall application, air dried, sieved (2 mm), and stored at 277 K until analysis. Total runoff from each rainfall was collected, thoroughly mixed, and subsampled. A 100-mL aliquot was immediately centrifuged, filtered (0.45 μm), and stored at 277 K for dissolved P analysis. Unfiltered runoff was stored at 277 K for particulate, bioavailable, and total P, and suspended sediment determination. These analyses were completed within 5 d of sample collection.

Suspended sediment concentration of runoff was determined in duplicate as the difference in weights of 250-mL aliquots of unfiltered and filtered samples after evaporation to dryness at 378 K.

### Chemical Analysis

Soil clay content was determined by pipette analysis after dispersion with sodium hexametaphosphate (Day, 1965), organic C by dichromate wet combustion (Raveh and Avnimel-ch, 1972), and pH using a glass electrode at a 5:1 water/soil ratio (w/w). The total P content of soil, poultry litter, and unfiltered runoff was determined following digestion with a semimicro Kjeldahl procedure (Bremner and Mulvaney, 1982). Poultry litter had an average total P content of 16.29 ± 0.34 g kg⁻¹.

The Mehlich-3 P content of soil prior to each runoff event was determined by extraction of 1 g soil with 10 mL of 0.2 M CH₃COOH, 0.25 M NH₄NO₃, 0.015 M NH₄F, 0.013 M HNO₃, and 0.001 M EDTA for 5 min (Mehlich, 1984). The bioavailable P content of soil and runoff (strip P) was determined using Fe-oxide impregnated strips (Sharpley, 1993).

Iron-oxide impregnated strips were prepared by immersing filter-paper circles (15 cm in diam., Whatman no. 50) in a 10% (w/v) solution of FeCl₃·6H₂O. The paper circles were then air dried and immersed in 2.7 M NH₄OH solution to convert FeCl₃ to Fe-oxide. Immersion in NH₄OH was carried out as rapidly as possible to avoid uneven oxide deposition on the paper (Lin et al., 1991). After the paper circles were air dried, they were cut into strips 10 by 2 cm and stored for later use.

One strip was shaken with 1 g soil or 5 mL of unfiltered runoff (made up to 50 mL with distilled water) for 16 h end-over-end at 298 K. The strip was then removed, rinsed free of soil particles, and shaken end-over-end for 1 h in 1 M HCl to remove bioavailable P. Phosphorus in all neutralized extracts was determined by the colorimetric method of Murphy and Riley (1962). The dissolved P concentration of runoff also was determined by the method of Murphy and Riley (1962) on filtered samples.

Phosphorus sorption isotherms were constructed for the 10 untreated soils using the procedure of Nair et al. (1984). One gram of soil was shaken with various additions of P (0–500 mg kg⁻¹ added as KH₂PO₄) in 25 mL of 0.01 M CaCl₂ on an end-over-end shaker at 298 K. After 24 h, the soil suspensions were centrifuged and filtered (0.45 μm) and the solution P concentration (C) determined. The amount of P sorbed (X) is the difference between P added and P remaining in solution. Using the Langmuir sorption equation, soil P sorption maximum was calculated as the reciprocal of the slope of the plot C/X vs. C (Olsen and Watanabe, 1957; Syers et al., 1973). The sorption saturation of each soil was calculated as the percentage of soil P sorption maximum as Mehlich-3 P.

All treatments and analyses were conducted in duplicate and means presented. Statistical difference and significance of treatment effects were evaluated by analysis of variance for paired data. The results are discussed in terms of concentrations only. As the soil boxes had impermeable bottoms, runoff volumes from all treatments and soils were similar (about 4 L). Also, the validity of using small boxes of packed soil to evaluate chemical losses in runoff from fields or watersheds is questionable. Thus, use of the boxes has been limited to studying variables controlling the release of soil chemicals to runoff.

### RESULTS AND DISCUSSION

#### Soils

Incubation of poultry litter and soil for 7 d increased Mehlich-3 and strip P linearly across the range of additions used (Fig. 1). The three soils shown in Fig. 1, San Saba, Shermore (fine-loamy, siliceous, thermic Typic Fragiudalf), and Stigler, provide a range in soil physical and chemical properties, particularly P sorption maxima, and exhibit differing increases in extractable soil P as P is added (Table 1). Similar increases were also observed for the other seven soils.

The relationship between P added as poultry litter and Mehlich-3 or strip P was significant for all soils at P < 0.01 (Table 2); however, slope of the linear relationship for Mehlich-3 P ranged from 0.34 to 0.70 and from 0.16 to 0.32 for strip P (Table 2). Regression slopes were consistently greater for Mehlich-3 than strip P, reflecting the greater extraction of soil P by the former method.

Although no information is available for poultry litter P, several studies have shown a linear relationship between Bray-I and Mehlich-3 P and the amount of inorganic fertilizer P added to soil (Barber, 1979; Fuleky, 1978; Peterson and Kreuger, 1980). For a 180-d incuba-
Table 2. Relationship between extractable soil P (mg kg$^{-1}$ as Mehlich-3 and strip P) and P added (mg kg$^{-1}$) as poultry litter after a 7-d incubation.$^\dagger$

<table>
<thead>
<tr>
<th>Soil</th>
<th>Mehlich-3 P</th>
<th>Strip P</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Slope</td>
<td>Intercept</td>
</tr>
<tr>
<td>Cahaba</td>
<td>0.49</td>
<td>21.9</td>
</tr>
<tr>
<td>Captina</td>
<td>0.62</td>
<td>36.8</td>
</tr>
<tr>
<td>Carnasaw</td>
<td>0.40</td>
<td>32.5</td>
</tr>
<tr>
<td>Durant</td>
<td>0.49</td>
<td>14.4</td>
</tr>
<tr>
<td>Muskogee</td>
<td>0.43</td>
<td>24.7</td>
</tr>
<tr>
<td>Rexor</td>
<td>0.48</td>
<td>29.6</td>
</tr>
<tr>
<td>Ruston</td>
<td>0.70</td>
<td>10.9</td>
</tr>
<tr>
<td>San Saba</td>
<td>0.34</td>
<td>13.9</td>
</tr>
<tr>
<td>Shermore</td>
<td>0.53</td>
<td>29.2</td>
</tr>
<tr>
<td>Stigler</td>
<td>0.67</td>
<td>28.5</td>
</tr>
</tbody>
</table>

$^\dagger$ All relationships significant at $P < 0.01$.

The slope of the soil P-added P regression represents an index of litter P availability, with increasing slope values indicating an increasing availability of added litter P in soil. Regression slope or litter P availability varied as a function of soil P sorption maxima (Fig. 2; $P < 0.01$). As P sorption maxima increased, a greater proportion of added poultry litter P was fixed in forms not extracted by Mehlich 3 or Fe-oxide strips. Thus, the availability of litter P in soil and potential for P loss in runoff is greater for low than high P sorbing soils.

**Runoff**

Following incubation of soil and poultry litter in the runoff boxes for 7 d, rainfalls (2.54 cm h$^{-1}$ for 30 min) were applied at 1-d intervals for five consecutive days, and runoff collected. The dissolved P concentration in runoff from Shermore soil decreased with successive rainfall events for each litter treatment (Fig. 3). The other nine soils behaved similarly. Similar trends of particulate and bioavailable P decrease were also observed for all soils (data not shown). As expected, the concentrations of dissolved P (Fig. 3), particulate P, and bioavailable P increased with greater amounts of P added as poultry litter. For presentational clarity, only P concentrations for the first runoff events are subsequently shown.

The concentrations of dissolved and bioavailable P in the first runoff were related ($P < 0.01$) to surface soil (0-1 cm) Mehlich-3 and strip P content prior to rainfall (Table 3). Particulate P concentration also was related to surface soil Mehlich-3 and strip P ($P < 0.01$; data not shown). The relationships between each form of P in runoff and Mehlich-3 and strip P for all treatments and events were significant ($P < 0.05$; $r^2$ of 0.70-0.80). The proportion of bioavailable P as dissolved P in runoff increased as extractable soil P content increased, with an average 33% for the check and 68% for the 320 kg P ha$^{-1}$ treatment. Thus, the immediate bioavailability and potential impact on surface water quality is greater as surface soil P increases.

The slope of the relationship between both dissolved and bioavailable P in runoff and extractable soil P differed with soil type, with the highest slopes occurring for soils with low P sorption maxima.

**Table 3.** Slope of the regression of dissolved and bioavailable P concentration of runoff (µg L$^{-1}$) and extractable soil P (mg kg$^{-1}$) and P sorption saturation (%) 7 d after poultry litter addition.$^\dagger$

<table>
<thead>
<tr>
<th>Soil</th>
<th>Dissolved P</th>
<th>Bioavailable P</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mehlich-3 P</td>
<td>Strip P</td>
</tr>
<tr>
<td>Cahaba</td>
<td>5.8</td>
<td>12.9</td>
</tr>
<tr>
<td>Captina</td>
<td>4.0</td>
<td>8.9</td>
</tr>
<tr>
<td>Carnasaw</td>
<td>2.4</td>
<td>5.1</td>
</tr>
<tr>
<td>Durant</td>
<td>3.1</td>
<td>6.5</td>
</tr>
<tr>
<td>Muskogee</td>
<td>2.0</td>
<td>4.0</td>
</tr>
<tr>
<td>Rexor</td>
<td>3.3</td>
<td>7.2</td>
</tr>
<tr>
<td>Ruston</td>
<td>5.7</td>
<td>12.4</td>
</tr>
<tr>
<td>San Saba</td>
<td>1.6</td>
<td>3.3</td>
</tr>
<tr>
<td>Shermore</td>
<td>4.7</td>
<td>10.1</td>
</tr>
<tr>
<td>Stigler</td>
<td>7.2</td>
<td>13.7</td>
</tr>
</tbody>
</table>

$^\dagger$ All relationships had an $r^2 > 0.90$ at a significance of $P < 0.01$.

$^\ddagger$ Phosphorus sorption saturation is represented by the percentage of P sorption maximum as Mehlich-3 P.
among soils (Table 3; Fig. 4). Thus, a single relationship cannot be used to describe the release of surface soil P to runoff. For example, a surface soil Mehlich-3 P content of 200 mg kg\(^{-1}\) would support a runoff dissolved P concentration of 280 µg L\(^{-1}\) for San Saba, but 1360 µg L\(^{-1}\) for Stigler (Fig. 4).

The slope of the dissolved P--soil P regression was related to soil P sorption maxima (Fig. 5). As P sorption maxima increased, the release of soil P to runoff at any given Mehlich-3 or strip P content decreased. Thus, the differential release of soil P can be accounted for by P sorption maxima (Fig. 5). Regression slopes for bioavailable (\(r^2 = 0.83\)) and particulate P (\(r^2 = 0.78\)) with Mehlich-3 or strip P also decreased as a function of soil P sorption maxima (data not shown).

The P sorption saturation of surface soil (0–1 cm) before each rainfall was calculated using Eq. [1] and Mehlich-3 P as extractable soil P for each P treatment and soil type. The dissolved P concentration in runoff was related (\(r^2 > 0.90\) and \(P < 0.01\)) to soil P sorption saturation for each soil (Table 3). More than 90% of the variability in bioavailable P concentrations in runoff from each soil also was accounted for by P sorption saturation (Table 3). Data in Table 3 are for the first runoff event after incubation; the dissolved and bioavailable P concentrations in the other events also were related to P sorption saturation (data not shown; \(r^2 > 0.86\) and \(P < 0.01\)). Slopes of the dissolved P--P sorption saturation regression were similar for all soils, as were slopes of the bioavailable P--P sorption saturation regression (\(P < 0.05\) as determined by analysis of variance for paired data). As a result, a single relationship (\(r^2 = 0.86\)) can be used to describe the dependence of dissolved P in runoff on P sorption saturation for all soils (Fig. 6). A single relationship (\(r^2 = 0.85; P < 0.01\)) between bioavailable P and P sorption saturation also was obtained for all soils. When P sorption saturation was calculated using strip P as the estimate of extractable soil P in Eq. [1], relationships of similar significance were obtained with both dissolved P (\(r^2 = 0.87, P < 0.01\)) and bioavailable P (\(r^2 = 0.88, P < 0.01\)).

When data for all ten soils are combined, it is apparent that a greater proportion of the variability in dissolved and bioavailable P (>85%) was accounted for by soil P sorption saturation than by Mehlich-3 (49 and 53%, respectively) or strip P (54 and 58%, respectively). Therefore, P saturation better describes the effect of soil type on the differential release of soil P to runoff and potential for P loss in runoff than conventional soil test P measures.

Using the relationship between runoff P and soil P saturation, we can determine either a P saturation that will support an acceptable P concentration in runoff or vice-versa, a P concentration that could be expected from a soil of given P saturation. For example, a P saturation of 25%, the critical value used in The Netherlands (Breeuwsma and Silva, 1992), would support a dissolved P concentration in runoff of 690 µg L\(^{-1}\) using Mehlich-3 P and 1584 µg L\(^{-1}\) using strip P. Clearly, the actual value of P sorption saturation for a given soil will vary with the method used to estimate extractable P in Eq. [1]. Once calibrated for specific methodology, however,
the P sorption saturation approach has the potential to describe the release of P from a wide range of soils to runoff.

CONCLUSIONS

The application of poultry litter increased extractable soil P content, measured as Mehlich-3 and strip P. The increase was greatest for low P-sorbing soils. Consequently, the potential for P enrichment of runoff will be greater following land application of poultry litter to low rather than high P sorbing soils.

The concentration of dissolved, bioavailable, and particulate P in runoff is related to the amount of Mehlich-3 or strip P in the surface layer of soil (0-1 cm), which reacts with rainfall-runoff. This relationship can be used as the theoretical basis to establish threshold or environmental soil test P levels, above which P enrichment of runoff becomes unacceptable. The first step in this recommendation process is to determine what concentration of P in runoff is unacceptable to specific receiving waters. If a dissolved P concentration of 250 μg L⁻¹ in runoff is determined unacceptable, threshold soil levels would range from 50 (Stigler) to 165 mg kg⁻¹ (San Saba) for Mehlich-3 P and from 20 to 80 mg kg⁻¹ for strip P. The range in Mehlich 3 values is similar to threshold levels used by several states in P-management recommendations (75 to 200 mg kg⁻¹; Gartley and Sims, 1994; Sharpley et al., 1994a). Thus, threshold Mehlich-3 P levels proposed by some states may indeed be reasonable.

As the relationship between runoff and soil P varies with soil type, use of these relationships in the P-management process would have to be soil dependent; however, for the 10 soils studied under a given slope, rainfall intensity, and duration, a single relationship described the dependence of runoff P on soil P sorption saturation. A dissolved P concentration of 250 μg L⁻¹ would result from a P saturation of 10% using Mehlich-3 P and 5% using strip P for any of the 10 soils. The added complexity of the P saturation approach compared to standard soil P test methods, in terms of obtaining reliable estimates of soil P sorption maxima, may limit its acceptability at the present time. Because this method is sensitive to how soil properties affect P sorption and desorption, this approach integrates the effect of soil types with soil P content to better estimate the potential for P loss in runoff from a given soil. In practice, use of the P saturation approach could be limited to soils already identified as being vulnerable to P loss due to high soil P, erodibility or leachability.

An added advantage of the P saturation approach is that it not only describes the potential for P release from soil, but also indicates how close the P sorption sites of a soil are to being saturated. In other words, measuring P sorption describes both the potential of a soil to enrich runoff with dissolved P (high degree of P saturation) and also helps to predict how much of the P added in fertilizers and manures will be retained by the soil in a form that is relatively resistant to release to runoff (low degree of P saturation).

If dissolved P concentrations of 250 μg L⁻¹ or greater are determined unacceptable, then a lower threshold soil test P or P saturation would be necessary. A dissolved P concentration of 250 μg L⁻¹ in runoff is not being promoted here as an unacceptable value; it is used merely as an example of how threshold soil P levels may be justified.

Once an unacceptable P concentration in runoff has been established for a given physiographic region by considering the presence or proximity of P-sensitive waters to edge-of-field losses, a threshold soil test P or P saturation value which has the potential to support this unacceptable P concentration can then be determined. As this study used sieved soil, packed boxes, and simulated rainfall, caution must be exercised in using these relationships quantitatively under field conditions. It does, however, provide a framework to establish threshold soil P levels above which the increased potential for P enrichment of runoff may necessitate a change in management of fertilizer and manure inputs of P.

While the concentration of P in runoff is related to soil P, the potential for P loss from a site will be dependent on runoff potential. Thus, a comprehensive approach that integrates soil P levels with the variability in runoff volume and erosion, resulting from climatic, topographic, and agronomic factors, will be needed for reliable yet flexible recommendations of fertilizer and manure P management.

REFERENCES

Murphy, J., and J.P. Riley. 1962. A modified single solution method for the determination of phosphate in natural waters.