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677 South Segoe Road, Madison, WI 53711 USA

Estimating Labile and Dissolved Inorganic Phosphate Concentrations in Surface Runoff

R. C. WENDT AND E. E. ALBERTS

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R. C. WENDT AND E. E. ALBERTS²

ABSTRACT

Mathematical models are needed to better estimate effects of alternative management methods on the amounts of bioavailable P lost in surface runoff over a variety of soil, topographic, and climatic conditions. In this study, methods for estimating concentrations of soil-derived labile (dissolved plus adsorbed) and dissolved P in runoff are proposed and tested. The methods are based on the assumption that most labile P contributed by soil to runoff is derived from soil particles that are detached and transported with runoff water. Effects of erosive selectivity are accounted for by dividing soil and sediment into five undispersed size fractions. Isotherms relating adsorbed labile and dissolved P levels were developed for each from soil analyses. Labile P concentrations in runoff were estimated by summing the product of labile P concentrations prior to entrainment in runoff and measured sediment concentrations for each of the size fractions. Dissolved P concentrations were estimated by distributing labile P between dissolved and adsorbed forms using a mass balance approach. Methods of estimation were tested with runoff from 14 natural rainstorms occurring on 10 fallow plots having soil P levels ranging from about 1.3 to 3.2 mmol kg⁻¹ as labile P. Linear regressions of observed on estimated values of both labile and dissolved P were significant. For most events, slopes and intercepts of regression lines were not significantly different from one and zero, respectively, indicating good absolute agreement. Results indicate that soil analyses in combination with models for predicting the amount and size distribution of eroded soil form a basis for estimating movement of bioavailable P forms in runoff. However, additional studies are needed to determine the range of soil types for which the assumptions inherent in the development of the approach are valid.

Additional Index Words: bioavailable P, eutrophication, non-point-source pollution, aggregates, adsorption isotherms.

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The bioavailability of P in runoff is an important consideration in developing management strategies for reducing rates of eutrophication of receiving waters (Sonzogni et al., 1982). Phosphorus forms and their bioavailability have been addressed in recent reviews by Sonzogni et al. (1982) and Nelson and Logan (1982). The P form that is directly bioavailable is dissolved inorganic phosphate (predominantly HPO₄²⁻ and H₂PO₄⁻ at normal pH's). Other P forms become available through conversion to this phosphate form. In most cases, these forms in runoff either convert to dissolved inorganic P relatively slowly or are usually present in relatively small amounts. However, dissolved inorganic P equilibrates rapidly with P adsorbed on particulate surfaces; hence, the particulates serve as a readily accessible reservoir of bioavailable P. Collectively, the dissolved and adsorbed P are often referred to as labile P.

To aid in assessing effects of alternative management strategies on P transport in runoff over a variety of soil,

topographic, and climatic conditions, mathematical models for predicting P movement have been proposed. Conceptually, these models have addressed interactions of rainfall and runoff water with soil in developing predictive equations, most of which are empirically derived. In some instances, only dissolved P in runoff has been considered (Romkens & Nelson, 1974; Sharpley et al., 1978, 1981), which may neglect significant amounts of bioavailable P on sediments. Models that have addressed both dissolved and sediment-associated P have often attempted to describe only total P quantities on sediments (Davis & Donigian, 1979; Frere et al., 1980), of which only a fraction may become bioavailable in aquatic systems.

Other modeling approaches (in most cases developed to predict pesticide movement in runoff) have considered quantities of labile chemical (Frere et al., 1975; Haith, 1980; Steenhuis & Walter, 1980; Leonard & Wauchope, 1980). These approaches have used adsorption isotherms and mass balance considerations to predict amounts of dissolved and adsorbed chemicals in or at the interface of a surface soil zone assumed to interact with rainfall and runoff water. Instantaneous equilibrium between dissolved and adsorbed forms has usually been assumed. Movement of dissolved chemical to runoff has been variously estimated by: (i) assuming concentrations in runoff to equal those in water in the surface soil zone (Frere et al., 1975; Steenhuis & Walter, 1980), (ii) by assuming dissolved chemical is removed from the surface soil zone in proportion to the fraction of rainfall as runoff (Haith, 1980) and, (iii) by assuming the concentration in runoff is equal to that predicted at the interface of the surface soil zone and runoff stream using an empirically derived, effective solution/soil ratio (Leonard & Wauchope, 1980). Adsorbed chemical loss in runoff has been estimated from the amount of soil eroded and the concentration of adsorbed chemical on the eroded soil. The latter has been derived from adsorption isotherms using predicted concentrations of dissolved chemical in the surface soil zone, both with corrections for selective removal of smaller-sized particles (Leonard & Wauchope, 1980; Frere et al., 1975) and without such corrections (Haith, 1980; Steenhuis & Walter, 1980).

Approaches similar to the above may be useful for predicting concentrations of dissolved and labile P in surface runoff. However, because P is strongly adsorbed on soil, simplifying assumptions regarding the mechanism of P transfer to runoff that would result in relatively simple predictive equations may be appropriate. This study proposes and tests such approaches for estimating concentrations of soil-derived labile and dissolved P in surface runoff.

CONCEPTUAL APPROACH

Identifying and describing the major transfer processes at the boundary between the soil surface and the moving film of runoff water are critical to predicting the

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²Soil scientists, USDA-ARS, Columbia, MO.

movement of soil-derived chemicals in runoff (Bailey et al., 1974). Because the majority of labile P in soil is adsorbed onto soil particles, detachment and suspension of soil in runoff is likely to be a major mechanism by which labile P traverses the soil-runoff boundary. Once suspended in runoff, net adsorption or desorption of P on particles may occur, depending on conditions in the runoff water including the dissolved P concentration and factors affecting the distribution of labile P between dissolved and adsorbed forms mentioned below. Smaller-sized soil particles and aggregates are assumed to have the greatest amounts of labile P and, once detached and suspended in runoff, are assumed to remain suspended and move with the runoff water. Larger soil particles and aggregates, which remain in place or which may be suspended for short time periods, are assumed to have less labile P and to desorb relatively little into runoff water. Similarly, P inputs to runoff from displaced soil water resulting from impacting raindrops or subsurface return flows are assumed to be relatively small. If these assumptions are valid, soil-derived labile P in runoff may be estimated from the amount of eroded soil and its labile P concentration prior to entrainment. Because P is strongly adsorbed, labile P concentrations of surface soil are not expected to be reduced greatly during an event by infiltrating water. Thus, labile P concentrations on soil are assumed to remain relatively constant during events. Because erosion may result in the selective removal of smaller-sized soil particles having higher P concentrations than their larger counterparts (Stoltenberg & White, 1953), the concentration of labile P for eroded soil may be greater, or enriched, relative to that for the source soil. With the proposed approach, effects of particle sorting on P movement are approximated by dividing soil into several size fractions and treating the movement of each separately. Because most soil is transported in runoff as aggregates (Alberts & Moldenhauer, 1981), size fractions considered are undispersed and, thus, consist of both primary particles and aggregates. Labile P concentrations in runoff are then estimated from

$$C_{pl} = \sum_{i=1}^n (P_i i \times C_{si}), \quad [1]$$

where

C_{pl} = Concentration of labile P in runoff ($\mu\text{mol P L}^{-1}$),

$P_i i$ = Concentration of labile P on the i th undispersed size fraction of soil ($\mu\text{mol P g}^{-1}$), and

C_{si} = Concentration of the i th undispersed sediment size fraction in runoff (g L^{-1}).

The distribution of labile P in runoff between dissolved and adsorbed forms as a steady state between these forms is approached will depend on the concentration of sediment and its P buffer capacity. A steady state between dissolved and adsorbed forms will not be reached instantaneously. However, rates of P adsorption and desorption are sufficiently rapid that the majority of the net change in the amount of P adsorbed taking place over several hours' time often occurs within the first few minutes of equilibration (Evans & Jurinak,

1976; Ryden et al., 1977). Thus, with the proposed approach, the distribution of labile P between dissolved and adsorbed forms is approximated using P buffer capacity estimates derived for near steady-state conditions. Because P buffer capacities may be greater for small-sized particles and aggregates than for larger ones, selective erosion of the former may result in buffer capacities for sediments that differ from those of the source soil. As with labile P, this effect is approximated by dividing sediments into several undispersed particle-size fractions. From a mass balance,

$$C_{pl} = C_{pd} \times V + \sum (C_{pai} \times C_{si}),$$

where

C_{pd} = Dissolved P concentration in runoff water ($\mu\text{mol P L}^{-1}$),

C_{pai} = Concentration of adsorbed labile P for the i th size fraction ($\mu\text{mol P g}^{-1}$),

V = Volume of water per unit volume of runoff (L L^{-1}), and

C_{pl} and C_{si} are as previously defined.

For each size fraction, adsorption isotherms for steady-state conditions may be expressed as

$$C_{pai} = C_{pd} \times K_i + B_i,$$

where K_i and B_i are isotherm slope and intercept, respectively. The adsorption isotherms are assumed to be independent of hysteresis and sediment concentration effects. Substituting these expressions for C_{pai} into the mass balance relationship and solving for C_{pd} gives

$$C_{pd} = \frac{C_{pl} - \sum_{i=1}^n (B_i \times C_{si})}{V + \sum_{i=1}^n (K_i \times C_{si})}. \quad [2]$$

Several additional factors may influence the steady-state distribution of labile P in runoff between dissolved and adsorbed forms, including aggregate stability (Barrow & Shaw, 1979), the type and amount of electrolyte dissolved in the runoff water (Barrow, 1972; Ryden & Syers, 1975), and temperature (Barrow, 1979). Because these factors are not included as variables, the accuracy of predictions will depend somewhat on how well experimental conditions used to derive P adsorption isotherms approximate conditions in runoff.

METHODS

The study was conducted using 10 runoff and erosion plots selected from those located at the Midwest Claypan Experiment Farm near Kingdom City (formerly McCredie), MO. Each was 3.2 m wide and 27.5 m long on the axis parallel to the slope. Plots for this study were selected to provide an approximately uniform distribution over a range of soil P levels.

The soil type at the site is a Mexico silt loam (fine montmorillonitic mesic, Udollic Ochraqualf) having clay, sand, and organic matter contents of approximately 240, 60, and 27 g kg^{-1} , respectively. The surface horizon is underlain by a claypan horizon at a depth of 0.2 to 0.3 m. The slope is 3 to 3.5%.

Plots were kept fallow during the study with periodic cultivation to keep the soil surface loose and weed-free. Runoff was collected by two tanks in series. The first tank collected up to about 650 L of runoff with a constant fraction of the overflow being directed to the second tank (for additional details see Jamison et al., 1968). Because overflow of the first tank might bias any relationship between labile P and sediments within tanks, only events for which all or almost all of the accumulated runoff was contained within the first tank were sampled as part of this study. Runoff was sampled as soon as possible after each event. However, because the minimum sampling time was about 4 h, the observed events were often composed of a series of smaller events occurring within a few minutes or hours of one another.

Runoff from 14 naturally occurring rainfall events was analyzed (Table 1). For each event, duplicate subsamples of runoff suspension were collected from each plot. One subsample was passed through a 0.45- μm filter and the filtrate stored at 4°C prior to analysis for dissolved P using the method of Murphy and Riley (1962). The second subsample was stored at 4°C prior to analyses for labile P and the amount and size distribution of eroded soil particles and aggregates.

Labile P in runoff was determined by isotopic dilution. Duplicate 25-mL aliquots of runoff suspension were spiked with carrier-free ^{32}P as NaH_2PO_4 in a 0.5-mL volume of distilled water and equilibrated for 24 h at $25 \pm 1^\circ\text{C}$ on an end-over-end shaker at 5 revolutions per minute (rpm). Solutions were clarified by centrifuging at $15\,000 \times g$ for 15 min. No differences in measured amounts of labile P were found between this method of clarification and filtration using a 0.45- μm membrane filter, which is commonly used as an operational means of separating solid and solution phases in water-quality studies. Activities of ^{32}P in the supernatant were determined by counting a 1-mL aliquot of supernatant evaporated to dryness on a metal planchet using a gas-flow counter. Labile P was determined from the amount of ^{32}P in solution and the fraction of added ^{32}P remaining in solution (Olsen & Sommers, 1982).

Size distributions of undispersed particles were determined based on settling velocities using the pipette method (Day, 1965). A density of 2.65 Mg m^{-3} was assumed for all particles. Because actual densities were probably somewhat lower and variable among aggregates, reported sizes are operationally defined. Size analyses were performed in native water except for samples having sediment concentrations $> 25 \text{ g L}^{-1}$. The latter were diluted with distilled water to concentrations $< 25 \text{ g L}^{-1}$ in order to minimize any errors resulting from particle interaction during settling. Total sediment was determined by drying an aliquot of runoff suspension at 105°C and weighing the residue. All results were corrected for dissolved solids content.

The surface 25 mm of soil on each plot was sampled at approximately 30-d intervals. Each sample consisted of the composite of 20 randomly selected, 20-mm diameter cores. Samples were air-dried and crushed to pass a 2-mm sieve. Duplicate subsamples of each were equilibrated with water at $25 \pm 1^\circ\text{C}$ for 24 h at a 50:1 solution/soil ratio. Dissolved P concentrations in equilibrating solutions passing a 0.45- μm filter were determined using the method of Murphy and Riley (1962). These concentrations were used with P adsorption isotherms, also determined at a 50:1 solution/soil ratio, to estimate labile P concentrations on each of the aggregate-size fractions. Predicted labile P concentrations were corrected for P desorbed into the equilibrating solution by distributing dissolved P among size classes in proportion to the fraction of the total adsorbed labile P on each size class.

The above soil samples were composited over time for each plot by combining equal weights of soil from each sampling time. Composited samples were used to determine P adsorption isotherms as a function of aggregate size. Duplicate 10-g samples of the composite sample for each plot were equilibrated overnight with 0.5 L of distilled water in 1-L plastic bottles on an orbital shaker at 100 rpm. A preliminary study had shown little change in the size distribution of particles after 8 and up to at least 16 h with the solution/soil ratio and shaking vigor used. It was assumed that the resulting particles were representative of those withstanding detachment and transport energies and moving with runoff water. Equilibrations with P were performed in distilled water because the resulting electrolyte solution was thought to better represent that for runoff water than dilute electrolytes, such as 0.01M CaCl_2 , which are often used as supporting solutions in P adsorption studies. Equilibrations were performed at normal laboratory temperatures ($25 \pm 1^\circ\text{C}$). After shaking, the contents of the two containers

Table 1—Selected characteristics for runoff monitored from fallow Mexico silt loam soil.

Date	Rainfall amount mm	Runoff volume L plot ⁻¹		Sediment conc g L ⁻¹		Mean weight diameter of aggregates [†] μm		Number of observations
		Mean	SD	Mean	SD	Mean	SD	
11 Apr.	38	59	24	31	6	24	2	10
14 Apr.	8	34	7	12	2	15	3	8
19 Apr.	30	136	45	6	2	24	8	10
23 May	11	191	24	10	3	18	3	9
30 May	15	94	44	26	7	9	1	9
4 June	10	45	25	9	3	9	2	7
5 June	5	32	13	11	2	9	2	9
16 June	18	310	63	39	11	21	4	9
20 June	17	590	76	19	4	18	2	8
1 July	31	948	124	24	7	24	2	10
25 Aug.	26	122	60	42	6	31	5	10
31 Aug.	25	715	84	51	8	25	5	6
13 Sept.	18	366	65	27	2	24	3	9
19 Oct.	23	790	183	9	1	20	2	9

[†] MWD = $\frac{\sum_{i=1}^n (\text{mean}_i \times w_i)}{\sum_{i=1}^n w_i}$ where mean_i and w_i are the mean diameter and the proportion of the total sediment weight, respectively. A mean diameter of $100 \mu\text{m}$ was assumed for aggregates $> 50 \mu\text{m}$. The MWD for bulk soil was $33 \mu\text{m}$.

were transferred to a 1-L graduated cylinder. Twenty-five-milliliter samples of the < 50 -, < 20 -, < 5 -, and < 2 - μm size fractions were obtained by the same procedures used to determine undispersed sediment-size distributions. Labile P was determined on each fraction using the previously described isotopic dilution method. Labile P on the bulk soil was determined using separate soil samples at a 50:1 solution/soil ratio. Because dissolved P levels remained relatively constant for soil from a given plot, amounts of adsorbed labile P within the > 50 -, 50- to 20-, 20- to 5-, 5- to 2-, and < 2 - μm size fractions at the ambient dissolved P level could be determined by difference. The mass of solids within size fractions were similarly determined by difference and used to express adsorbed labile P on a per-unit mass basis. The combined data for all plots gave adsorbed labile P concentrations on aggregates at several different dissolved P levels for each size fraction from which adsorption isotherms were constructed.

RESULTS AND DISCUSSION

P Adsorption Isotherms for Undispersed Soil Particles

Phosphate adsorption isotherms for size fractions of undispersed soil particles and the bulk soil are approximately linear over the observed concentration ranges (Fig. 1). Although most P adsorption isotherms reported in the literature are curvilinear, linear relationships between labile and dissolved P have similarly been observed by Olsen and Watanabe (1963) over comparable P concentration ranges. Most variation among size fractions is presumably due to differences in the numbers and/or energies of adsorption sites as affected by the size distribution of primary particles within the aggregates. Physical limitations to accessibility of sites on particles within aggregates may also contribute somewhat to observed differences. For size classes $< 50 \mu\text{m}$, amounts of labile P adsorbed at equivalent dissolved P levels vary inversely with size. In most cases, however, particles $> 50 \mu\text{m}$ contain more adsorbed labile P per unit mass at equivalent dissolved P levels than the bulk soil or the 50- to 20- μm and 20- to 5- μm sizes. This may be due in part to a higher proportion of smaller-sized primary particles within those particles $> 50 \mu\text{m}$ relative to that for the bulk soil or particles 50-

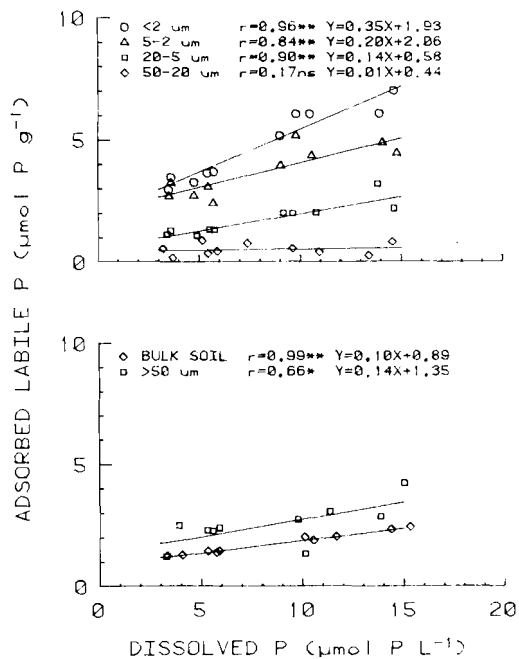


Fig. 1—P adsorption isotherms by undispersed particle size and for bulk soil for Mexico sil soil.

to 20- μm and 20- to 5- μm in size. Recently, Alberts et al. (1983) observed such a trend among eroded aggregates derived from a Monona silt loam soil (fine-silty, mixed, mesic Typic Hapludoll) for which percentage clay by weight followed the relative order: 10 to 2 μm > 2000 to 50 μm > bulk soil > 20 to 10 μm > 50 to 20 μm .

It is apparent from the isotherms in Fig. 1 that labile P concentrations and P buffer capacities of undispersed particles are not a strictly increasing function of decreasing size distribution. Because labile P concentrations are greater for the > 50- μm size fraction than for the bulk soil, selective removal of particles < 50 μm would result in sediments having lower adsorbed labile P concentrations than the bulk soil at equivalent dissolved P concentrations. Likewise, insufficient capacity to transport particles > 20 μm could have a compensating effect such that labile P concentrations on the resulting sediments would not be greatly different than that of the source soil. Generally, however, smaller-sized undispersed particles do contain more labile P per unit mass and have greater P buffer capacities than larger ones. The relative importance of these differences to labile and dissolved P in runoff will, however, also depend on amounts of sediment being transported in each of the size fractions.

Adsorption isotherms such as those in Fig. 1 may be used with relatively simple soil analyses, such as P equilibrating with water, to estimate concentrations of labile P on aggregate-size fractions. Similar P adsorption isotherms might be experimentally developed on the basis of soil type. Amounts of P on particles for a specific source area might then be estimated from results of a simple equilibration procedure and the appropriate set of adsorption isotherms. The usefulness of such an approach would, however, depend on the variability of isotherms within a soil type.

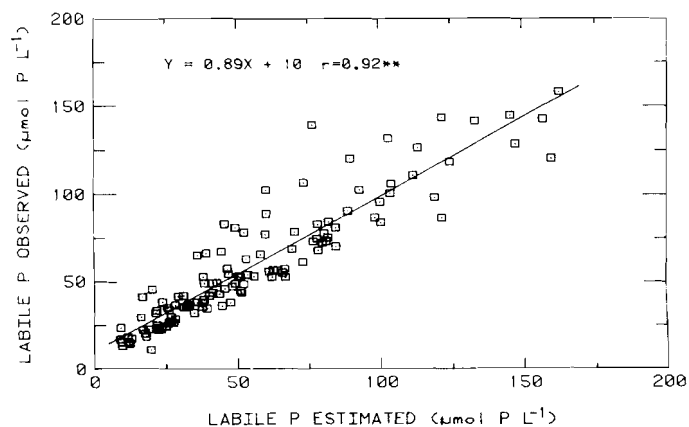


Fig. 2—Comparison of observed and estimated labile P concentrations in runoff for all plots and events.

Labile P Concentrations in Runoff

Labile P concentrations in runoff were predicted from Eq. [1] using measured sediment concentrations and predicted labile P concentrations for each of the > 50-, 50- to 20-, 20- to 5-, 5- to 2-, and < 2- μm size fractions. A comparison of observed and predicted labile P concentrations in runoff for all plots and events using Eq. [1] shows a highly significant relationship with reasonable absolute agreement (Fig. 2). Temporal variability in P on surface soil as indicated by variability in dissolved P levels equilibrating with water was small. The labile P concentrations for undispersed soil particles were, therefore, derived from P adsorption isotherms using the mean dissolved P concentration over the seven 30-d samplings for each plot. Because the analysis for combined events may be biased somewhat by data from events with higher concentrations, similar regression analyses were performed on an event basis. In all cases, relationships between observed and predicted concentrations are significant (Table 2). In most cases, slopes and intercepts are not significantly different than one and zero, respectively, indicating good absolute agreement between the observed and predicted values. There are no apparent effects of the size or the time of occurrence of events on regression parameters.

Table 2—Results of linear regression analysis of observed (y) on predicted (x) labile P concentrations in runoff by event.

Event	<i>r</i>	Slope†	Intercept	Observed range
			$\mu\text{mol P L}^{-1}$	$\mu\text{mol P L}^{-1}$
11 Apr.	0.90**	0.95	38*	67-143
14 Apr.	0.93**	1.47	-13	11- 57
19 Apr.	0.96**	1.55**	0	13- 38
23 May	0.67*	0.96	12	20- 65
30 May	0.97**	0.80*	5	32-142
4 June	0.83*	0.55*	20*	17- 56
5 June	0.95**	1.02	3	25- 69
16 June	0.93**	0.76	19	53-158
20 June	0.98**	0.89	2	25- 77
1 July	0.97**	0.83	5	23- 74
25 Aug.	0.91**	0.79	24	53-141
31 Aug.	0.98**	1.02	0	54-144
13 Sept.	0.97**	1.09	-3	36-102
19 Oct.	0.92**	1.17	0	19- 36

*,** Significant at the 0.05 and 0.01 levels, respectively.

† Slopes and intercepts followed by an asterisk are significantly different from one and zero, respectively.

Slopes of regression lines appreciably > 1 suggest that P is contributed to runoff from sources not associated with soil movement. However, the high correlations between observed and predicted values show errors of estimation to vary linearly with the magnitude of the estimate. For a given event, the size distribution of eroded particles among plots tends to be similar. Consequently, differences in estimates for a given event are due in large part to differences in soil P levels making the magnitude of estimation errors proportional to soil P level. Potential P sources consistent with these observations and not accounted for in the present approach might be P dissolved in a relatively constant volume among plots of subsurface return flow or P desorbed from a relatively constant amount of soil not transported with runoff water.

Slopes of regression lines appreciably < 1 show overprediction. Again, however, errors are systematic. These errors may be due to a disparity between labile P concentrations on undispersed soil particles and those that are eroded. Variation in the stability of soil aggregates may contribute to such errors. Intercepts significantly greater than zero, such as that for the event of 11 April, indicate a relatively constant addition of unaccounted-for P to all plots, regardless of soil P level. Potential sources of such P additions are not readily apparent.

Dissolved P Concentrations in Runoff

For most runoff events, dissolved P was a relatively minor portion of labile P. Mean percentages of labile P as dissolved P by event ranged from 6 to 22%, with a mean value for all observations of 13%.

Dissolved P concentrations in runoff were estimated using Eq. [2] with the estimated concentration of labile P from Eq. [1] and adsorption isotherms and measured sediment concentrations in runoff for the same undispersed size fractions. Comparisons between observed and estimated dissolved P levels in runoff using Eq. [2] were highly significant for all events and plots combined (Fig. 3) and for individual events (Table 3). In most cases, slopes and intercepts of regression lines for individual events are not significantly different from

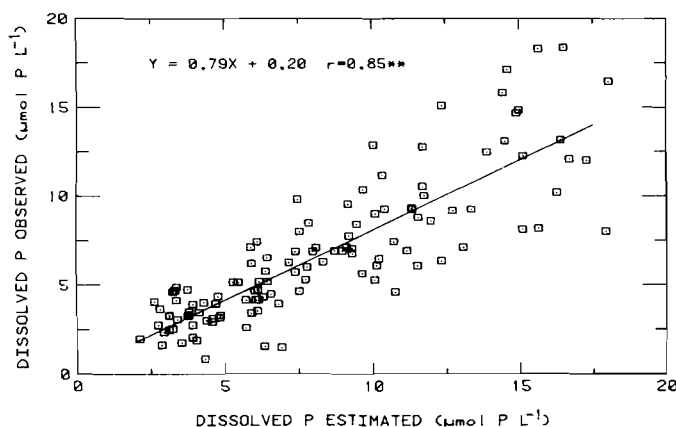


Fig. 3—Comparison of observed and estimated dissolved P concentrations in runoff water for all plots and events.

one and zero, respectively, indicating good absolute agreement between observed and estimated values. However, for five events, predicted values are significantly greater than those observed. Because labile P concentrations were reasonably well estimated for these events, the source of error appears to be a failure to predict the distribution of the labile P between dissolved and adsorbed forms.

As previously mentioned, factors such as the type and amount of electrolyte in runoff, aggregate stability, and temperature may influence the distribution of labile P between dissolved and adsorbed forms. Effects of these factors may be partially accounted for by selecting experimental conditions for derived P adsorption isotherms that approximate those in runoff. However, conditions in runoff dissimilar from those selected could cause systematic errors such as those observed for some events in Table 3. For example, increasing the electrolyte concentration in the supporting solution is known to favor a shift in the distribution of labile P toward adsorbed P. Hence, electrolyte concentrations in runoff higher than those that existed in deriving adsorption isotherms could cause dissolved P concentrations in runoff to be a relatively constant fraction of those predicted. Similarly, lower temperatures in runoff than those used for laboratory determination of isotherms could result in overprediction. In order to achieve more accurate estimates, effects of factors such as these may need to be considered quantitatively.

In most cases, dissolved P concentrations in runoff are within the range of observations on experimentally derived P adsorption isotherms. The effectiveness of these isotherms for prediction purposes outside this range is uncertain. Although linear over the observed range, the adsorption isotherms would not be expected to remain so over wider concentration ranges. This may be particularly important at lower P concentrations where isotherms would be expected to tend toward the origin resulting in buffer capacities greater than those predicted by extrapolating the linear isotherms. Lower concentration ranges might be approached as runoff mixes with runoff from other sources or is greatly diluted in surface impoundments.

Table 3—Results of linear regression analysis of observed (y) on predicted (x) dissolved P concentrations in runoff by event.

Event	r	Slope†	Intercept		Observed range
			$\mu\text{mol P L}^{-1}$		
11 Apr.	0.89**	0.80	-0.3		3.2-15.8
14 Apr.	0.94**	1.15	-0.3		3.2-15.2
19 Apr.	0.85**	0.88	0.7		1.9- 9.7
23 May	0.95**	0.77*	0.0		1.6-10.0
30 May	0.98**	1.08	-1.3		1.6-18.4
4 June	0.94**	0.91	-0.3		2.9-12.6
5 June	0.99**	0.87	-1.0		2.6-12.3
16 June	0.98**	0.73**	0.0		2.6-12.0
20 June	0.99**	1.12	0.3		4.5-18.4
1 July	0.98**	0.98	-0.3		3.2-14.9
25 Aug.	0.89**	0.47**	0.7		1.6-10.0
31 Aug.	0.99**	0.76**	0.7*		3.9-13.2
13 Sept.	0.95**	0.58**	-0.7		1.6- 8.1
19 Oct.	0.94**	0.86	-1.0		1.0- 9.0

** Significant at the 0.05 and 0.01 levels, respectively.

† Slopes and intercepts followed by an asterisk are significantly different from one and zero, respectively.

CONCLUSIONS

This study shows that soil-derived labile P concentrations in runoff may be estimated from results of soil analyses and knowledge of the amount and size distribution of eroded soil particles. Thus, models for predicting the latter in combination with soil analyses should provide a basis for predicting labile P movement in runoff. Accuracy of predictions with such an approach would, of course, also be dependent on errors associated with predicting size distributions. Results also show that the distribution of labile P between dissolved and adsorbed forms may be estimated. However, additional factors affecting this distribution, such as electrolyte concentration and temperature, may need to be considered in order to provide consistently accurate predictions. The approach used in this study may also be useful for predicting movement in runoff of other strongly adsorbed chemicals such as some pesticides. However, additional information is needed in order to determine the range of adsorption energies and soil types over which the assumptions inherent in the development of this approach apply.

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