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677 South Segoe Rd., Madison, WI 53711 USA

Phosphorus Runoff from Two Water Sources on a Calcareous Soil

J. K. Aase, D. L. Bjorneberg, and D. T. Westermann*

ABSTRACT

Phosphorus (P) in irrigation runoff may enrich offsite water bodies and streams and be influenced by irrigation water quality and antecedent soil surface conditions. Runoff, soil loss, and P fractions in runoff using reverse osmosis (RO) water or mixed RO and well water (RO/Tap) were studied in a laboratory sprinkler study to evaluate water source effects on P transport. A top- or subsoil Portneuf silt loam (coarse-silty, mixed, superactive, mesic Durinodic Xeric Haplocalcid), either amended or not amended with manure and/or with cheese whey, with Olsen P from 20 to 141 mg kg^{-1} and lime from 108 to 243 g kg^{-1}, was placed in 1.5 × 1.2 × 0.2-m-deep containers with 2.4% slope and irrigated three times from a 3-m height for 15 min, applying 20 mm of water. The first irrigation was on a dry loose surface, the second on a wet surface, and the third on a dry crusted surface. Surface (ca. 2 cm) soil samples, prior to the first irrigation, were analyzed for Olsen P, water-soluble P (P_{ws}), and iron-oxide impregnated paper-extractable P (FeO-P) analyses. Following each irrigation we determined runoff, sediment, dissolved reactive phosphorus (DRP) in a 0.45-μm filtered sample, and FeO-P and total P in unfiltered samples. Soil surface conditions had no effect on P runoff relationships. Water source had no significant effect on the relationship between DRP or FeO-P runoff and soil test P, except for DRP in RO runoff versus water-soluble soil P (r^2 = 0.90). Total P in RO runoff versus soil P were not related; but weakly correlated for RO/Tap (r^2 < 0.50). Water source and soil surface conditions had little or no effect on P runoff from this calcareous soil.

In the quest to reduce phosphorus (P) enrichment of water bodies and streams by runoff from agricultural fields, many factors and variables must be considered. Included are soil physical (texture and aggregation) and chemical properties (P content, pH, mineralogy), land management (tillage, P application, slope, erosivity), and offsite transport processes. These factors affect not only susceptibility for erosion and runoff to occur, but also the potential for soil P release to runoff water. Many of these factors and relations among them are discussed by contributors to the volume edited by Tunney et al. (1997). Their discussions center mainly around rainfed agricultural practices on acidic soils largely covered by vegetation.

Dissolved reactive phosphorus (DRP) concentrations in soil solution required for plant growth are in the range of 0.2 to 0.3 mg L^{-1}. Total P concentrations as low as 0.02 mg L^{-1} may cause eutrophication of lakes and streams (USEPA, 1996). The USEPA (1986) recommended a limit of 0.05 mg total P L^{-1} in streams flowing into lakes, and 0.1 mg total P L^{-1} in other waters. Therefore, P entering lakes and streams from agricultural runoff could seriously affect growth of algae and other aquatic plants. Research is under way to identify and develop management practices that will minimize potential P effect on water bodies (Sharpley et al., 2000; Tunney et al., 1997).

Generally, total P losses leaving agricultural fields are not large and depend greatly on sediment amounts carried off the fields (Berg and Carter, 1980). In irrigated agriculture, most erosion occurs from surface irrigation. However, depending on field slopes and water application rates, runoff and erosion also occur from overhead sprinkler irrigation systems. Some of this runoff may reach water bodies. Therefore, development of management practices to minimize potential runoff and concomitant P loss is also germane where sprinkler irrigation is practiced.

Soil tests are available relating soil P concentrations to crop needs. Whether or not the same soil tests are related to P in runoff remains uncertain (Sibbesen and Sharpley, 1997). Pote et al. (1996) found on fescue (*Festuca arundinacea* Schreb.)-covered acid soil that dissolved reactive P and biologically available P in runoff were better related to soil P extracted by distilled water, iron-oxide impregnated paper strip (Sharpley, 1993), or acidified ammonium oxalate than to soil P extracted by Mehlich III (Mehlich, 1984), Bray–Kurtz P1 (Bray and Kurtz, 1945), and Olsen (Olsen et al., 1954) methods. In a subsequent simulated rainfall field study on three Ultisols, Pote et al. (1999) considered site hydrology by dividing runoff DRP concentrations by runoff volume and found that runoff P from each soil had statistically (P = 0.05) the same correlations to water-extractable soil P.

Water quality available for runoff studies varies depending on source and may differ from rain water quality (Lentz et al., 1996). Therefore, a postulate is that water from various sources may influence runoff and P losses. Soil surface conditions, whether a loose dry seedbed, a wet surface following a rain, or a dry crust, may also influence runoff and P loss. Thus, our objectives were, on a calcareous soil in a laboratory sprinkler study, to evaluate effects of water from two sources on runoff, soil loss, and phosphorus forms in runoff from soil with different surface conditions and P concentrations.

MATERIALS AND METHODS

We conducted the study in the hydraulics laboratory of the Northwest Irrigation and Soils Research Laboratory of the USDA-ARS at Kimberly, ID. Soil samples were from

USDA-ARS, Northwest Irrigation and Soils Research Lab., 3793 N. 3600 E., Kimberly, ID 83341. Received 8 Sept. 2000. *Corresponding author (dtw@kimberly.ars.pn.usbr.gov).

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Abbreviations: DRP, dissolved reactive phosphorus; EC, electrical conductivity; FeO-P_s, iron-oxide impregnated paper–extractable phosphorus in soil; FeO-P_w, iron-oxide impregnated paper–extractable phosphorus in runoff water; P_i, Olsen P (inorganic phosphorus); PSI, phosphorus sorption index; P_{ws}, water-soluble phosphorus; RO, reverse osmosis; RO/Tap, 50:50 mix of RO and well water; SAR, sodium adsorption ratio.

Table 1. Characteristic average soil test values of soils used in the study.

Treatment	OC†	Lime	Olsen P	Olsen P, organic	FeO-P‡	Water-soluble P	Total P	pH	EC§	SAR¶
	g kg^{-1}		mg kg^{-1}						dS m^{-1}	
					Subsoil					
Conventional	5.2	242	33.1	3.7	46.9	2.2	800	7.9	0.6	0.8
Manure 1991	8.9	238	86.9	12.3	85.1	10.7	884	7.9	0.7	0.6
Manure 1994	8.9	246	107	15.4	96.5	11.5	922	7.8	0.9	0.6
Whey	6	249	63	5.9	62.5	5.7	907	7.9	0.6	0.6
					Topsoil					
Conventional	8.5	107	23.9	4.3	41.2	5.7	766	7.8	0.5	0.4
None	9.1	76	21.1	4	40.7	6.4	753	7.7	0.5	0.5
Manure 1994	10.6	119	69.4	12.5	83.2	17.4	810	7.7	0.8	0.5
Whey	9	135	51.3	9.3	64.8	10.9	828	7.7	0.6	0.5

† Organic carbon.
‡ Iron-oxide impregnated paper–extractable phosphorus.
§ Electrical conductivity.
¶ Sodium adsorption ratio.

previously established erosion and reclamation plots, details of which are described by Robbins et al. (1997, 1999). We recount a brief description of the plots and treatments.

The soil was a Portneuf silt loam. Topsoil was removed from some plots in 1991 to create a pattern of topsoil and exposed subsoil. Conventional subsoil plots were fertilized according to soil test with monocalcium phosphate in 1991. Subsoil whey plots received acid cheese whey (with H_3PO_4) in the spring and fall of 1991, and sweet cheese whey in the fall of 1994. Subsoil manure 1994 plots received acid whey in the spring and fall of 1991, and an application of fresh dairy manure in the fall of 1994. Subsoil manure 1991 received fresh dairy manure in the spring and fall of 1991. Topsoil manure 1994 received fresh dairy manure in the fall of 1994, and topsoil whey received sweet whey in the fall of 1994. Conventional topsoil plots received no fertilizer or amendments. General background soil test concentrations are listed in Table 1. Average Olsen P concentrations ranged from 21 mg kg^{-1} in the subsoil to 107 mg kg^{-1} in the topsoil. Topsoil plots averaged 9.2 g kg^{-1} organic carbon, nonmanured subsoil plots averaged 5.6 g kg^{-1}, and manured subsoil plots, 9.0 g kg^{-1}. Lime content for the topsoil averaged 108 g kg^{-1}, and for the subsoil, 243 g kg^{-1}.

We collected field-dry (ca. 14% by weight) surface soil (ca. 8 cm depth) from duplicate plots of selected treatments (Table 1) in the fall of 1998 following harvest of dry beans (*Phaseolus vulgaris* L.). The unsieved bulk soil was stored in covered 114-L plastic containers until we performed the simulated sprinkler irrigation and runoff tests in spring of 1999. For the tests we used six steel boxes 1.5 m long, 1.2 m wide, and 0.2 m deep at a slope of 2.4%. The 1.2-m-wide downslope edge of each box was reduced to a 0.15-m height to provide for the attachment of a runoff trough to collect runoff. Each box had a drainage manifold. However, none of the treatments ever saturated and drainage was not necessary.

To minimize the soil requirement for each test we placed a dry substrate of leveled clay loam in each box. Each test soil was friable and no screening or mixing was done prior to placing it on top of the substrate at about a 5-cm depth. We smoothed the soil surface by screeding (moving a flat wood board edgewise back and forth across the soil surface), leaving the surface crumbly and loose. Bulk density was about 1 Mg m^{-3} in all tests.

There was a total of 32 complete runoff tests: two each of four topsoil and four subsoil samples and two water sources. Each test soil was irrigated sequentially three times with either reverse osmosis (RO) water or a mix of half RO water and half well water (RO/Tap). The first irrigation was when the soil surface was air-dry and crumbly, followed by a second irrigation 2 d later when the soil surface was still visibly wet. The third irrigation followed about 7 to 10 d later when the soil surface had dried and formed a crust with resultant shallow 1- to 2-mm-wide cracks. Each irrigation was restricted to 15 min. Following three irrigation runs with either RO water or RO/Tap water, the soil was allowed to dry. The surface soil was then removed and replaced with the next set of sample soils and irrigated. To maintain similar conditions for each irrigation series, the moist substrate was loosened prior to adding new test soil. Soil water content, by oven-dry weight, in the surface 2.5 cm ranged from 9 to 13% for the first, 22 to 27% for the second, and 10 to 13% for the third irrigation. These soil water contents compare to about 24% at field capacity, and about 12% at the permanent wilting point for this soil (Robbins, 1977).

We used the RO/Tap water to simulate surface irrigation water. The RO/Tap water had pH = 7.5, electrical conductivity (EC) = 0.4 dS m^{-1}, sodium adsorption ratio (SAR) = 1.3, total P < 0.02 mg L^{-1}, and DRP < 0.01 mg L^{-1}. The RO water had pH = 5.3, EC = 0.02 dS m^{-1}, total P < 0.02 mg L^{-1}, and DRP < 0.002 mg L^{-1}; SAR was nonapplicable. For comparisons, Snake River source-irrigation water for the area averages pH = 8.2, EC = 0.5 dS m^{-1}, SAR = 0.7, and total P and DRP ≤ 0.07 mg L^{-1} (Carter et al., 1973, 1974). Average reported rainwater analysis for Idaho locations is pH ≈ 5.4, EC ≈ 5.7×10^{-3} dS m^{-1}, and SAR ≈ 0.8 (National Atmospheric Deposition Program, National Trends Network, 2000).

Water was pumped from 210-L mixing barrels and applied through an oscillating sprinkler similar to one described by Meyer and Harmon (1979). A Veejet nozzle (8070; Spraying Systems Co., Wheaton, IL) was mounted 3 m above the soil surface and water was applied at 80 mm h^{-1} with a nozzle pressure of 76 kPa, providing a 1.2-mm diam. median drop size and a 20-mm water depth per irrigation. The resultant Christiansen's uniformity coefficient was 88% (Cuenca, 1989). Droplet energy striking the soil surface was about 25 J kg^{-1} (Kincaid, 1996). The configuration was chosen to simulate water application rates at the outer end of center pivot irrigation systems.

Runoff was determined following each irrigation run by weighing the gross runoff and subtracting the weight of the filtered and dried sediment. Filtration was done by using Whatman (Maidstone, UK) No. 5 filter paper placed in a buchner funnel subjected to house vacuum. Sediment and filter paper were oven-dried at 105°C and weighed; filter paper weight was subtracted to obtain sediment weight. We took two 60-mL water samples for phosphorus analyses as soon as each irrigation run was completed. One sample was filtered through a 0.45-μm filter and stabilized with 0.6 mL saturated boric acid for later DRP analysis. The second, unfiltered sam-

ple was used to determine total P after persulfate digestion (American Public Health Association, 1992) and biologically available P by the iron-oxide impregnated filter paper strip method (Sharpley, 1993).

Prior to the first irrigation, four surface (ca. 2 cm) soil samples from each soil box were collected, composited, and analyzed for inorganic Olsen P (P_i) (Olsen et al., 1954), organic Olsen P following digestion with persulfate, iron-oxide impregnated paper–extractable phosphorus in soil (FeO-P_s), and water-soluble phosphorus (P_{ws}) (Pote et al., 1996). We determined a single-point phosphorus sorption index (PSI) based on the procedure developed by Bache and Williams (1971). All phosphorus concentrations were determined using the molybdenum-blue method (Murphy and Riley, 1962). We also determined acid equivalent lime (Allison and Moodie, 1965) and organic carbon (Nelson and Sommers, 1982) on each soil sample.

The data were analyzed with statistical regression techniques using a general linear test approach described by Neter and Wasserman (1974) and by stepwise regression. All statistical comparisons are reported at $P \leq 0.05$.

RESULTS AND DISCUSSION

Olsen P concentrations in our soil samples ranged from 20 to 141 mg kg^{-1}. Manured subsoils had the highest P_i concentration. None of the soil samples were below the range (20 to 30 mg kg^{-1}) normally considered P deficient for field crop production. There were statistically significant ($P \leq 0.05$) linear relationships among soil test P concentrations. The strongest relationship was between FeO-P_s and P_i, described by FeO-P_s = $0.72 \times P_i + 27.13$, $r^2 = 0.89$. The relationship between P_{ws} and P_i was the weakest with $r^2 = 0.49$. The relationship between P_{ws} and FeO-P_s was intermediate with $r^2 = 0.57$.

Whether the soil surface was initially loose and dry (first irrigation), wet (second irrigation), or dry and crusted (third irrigation) made no difference on any of the relationships developed except for differences in runoff quantities (Fig. 1A). These differences were statistically significant with the order of runoff from wet surface > dry crusted > dry loose. Comparisons of soil loss and sediment concentration between the two water sources were essentially random with r^2 values less than 0.1. Therefore, although irrigation sequences are identified when data are presented in figure form, no consideration will be given to soil surface conditions when reporting or discussing results.

Linear regression calculations between three forms of P in runoff water are shown in Table 2. The relationships followed the same pattern in both RO and RO/Tap runoff water with the better relationship between DRP and iron-oxide impregnated paper–extractable phosphorus (FeO-P_w). The better relationships were in RO/Tap runoff water, the weakest being between total P and DRP ($r^2 = 0.35$). In RO runoff water, there was a very weak relationship between total P and FeO-P_w ($r^2 = 0.11$), and no relationship between total P and DRP.

Simple regression analyses showed that runoff DRP was significantly related to all three soil P tests whether runoff was from RO or RO/Tap water (Fig. 2). There was no difference in regressions ($P \leq 0.05$) between RO

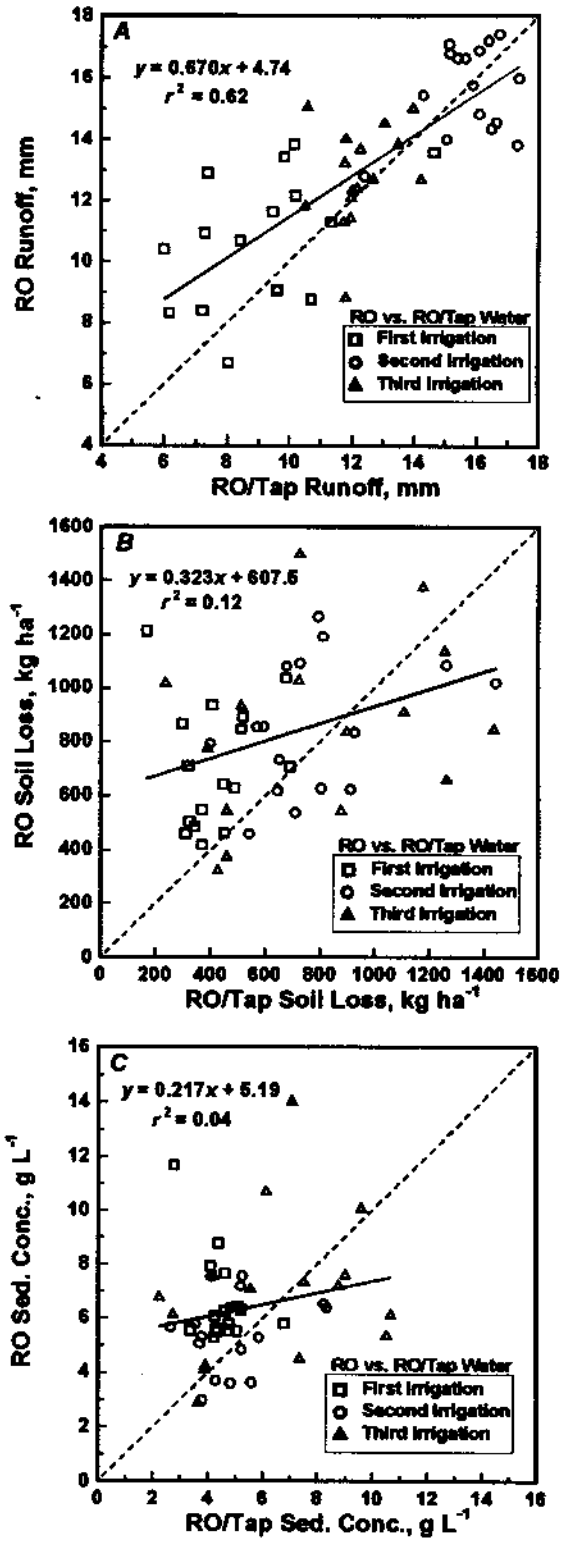

Fig. 1. Comparisons of runoff between reverse osmosis water (RO) and a 50:50 mix of RO water and tap water (RO/Tap) and between soil loss and sediment concentrations in the runoff waters. Regressions, except for sediment concentration, are significant; runoff differed significantly among the three irrigations ($P \leq 0.05$). Dashes signify 1:1 lines.

Table 2. Linear relationships among three forms of phosphorus in runoff from soils irrigated with either reverse osmosis water (RO) or a 50:50 mix of RO and well water (RO/Tap).

RO water		RO/Tap water	
DRP = 0.20 × FeO-P_w − 0.022†	r^2 = 0.52	DRP = 0.21 × FeO-P_w − 0.036	r^2 = 0.63*
Total P = 1.51 × FeO-P_w + 5.03	r^2 = 0.11*	Total P = 0.34 × FeO-P_w + 2.40	r^2 = 0.61*
Total P = 4.26 × DRP + 5.69	r^2 = 0.07ns‡	Total P = 9.80 × DRP + 3.79	r^2 = 0.35*

* Significant at the 0.05 probability level.
† DRP, dissolved reactive phosphorus; FeO-P_w, iron-oxide impregnated paper–extractable phosphorus in runoff water.
‡ ns, not significant.

and RO/Tap water for soil P_i and FeO-P_s concentrations, indicating that the respective relationships belong to the same population. The relation between DRP and P_{ws} was significantly different (r^2 and slope) between the two water sources, having a much stronger predictive value for RO than for RO/Tap water, and was the best of any calculated. By adding lime, organic carbon, and runoff in a stepwise regression analysis for RO water there was a slight improvement to $R^2 = 0.93$ from $r^2 = 0.90$; by reducing the added independent variables to only lime, R^2 became 0.91. Thus, there was not much to be gained by adding variables. For RO/Tap water nothing was gained through stepwise regression.

Similar results to those shown for DRP in Fig. 2 were obtained for FeO-P_w versus soil test concentrations (Fig. 3). In this instance all regressions were statistically the same between RO and RO/Tap waters. However, simple linear relations between FeO-P_w and soil test concentrations were generally better than those between DRP and soil test concentrations. Another difference between the two was a consistently stronger relationship (greater r^2) in all instances between FeO-P_w and soil tests using RO/Tap water than there was using RO water, contrary to that for DRP. There was no advantage in using stepwise regression analysis for RO water. However, for RO/Tap water, again using all soil P tests, lime, organic carbon, and sediment concentrations and runoff as independent variables in a stepwise regression analysis, FeO-P_s, P_i, sediment concentration, and runoff were selected, resulting in an R^2 of 0.90. That is probably not enough of an improvement beyond the simple regression r^2 values of 0.85 and 0.82 found in the relationships of FeO-P_w to P_i or FeO-P_s, respectively (Fig. 3), to warrant the extra time and labor to sample and analyze for the added variables needed.

Total P in RO runoff was not related to any soil test (Table 3). However, in RO/Tap runoff water, total P was related to all three soil tests, albeit the relations were not strong, the strongest being with P_i ($r^2 = 0.48$). Total P in runoff is generally related to sediment concentration, and, in our study, total P was in fact statistically related to sediment concentration in runoff from both RO and RO/Tap water (Fig. 4). Although the simple regression relations are statistically significant, and differ from each other, with $r^2 = 0.53$ for total P in RO/Tap runoff water versus sediment concentration, there was little predictive value, particularly between total P in RO runoff water versus sediment concentration, with $r^2 = 0.17$.

Employing stepwise regression procedures with total P as the dependent variable and all soil test P, lime, organic carbon, and sediment concentrations and runoff as independent variables returned no selections for improvements in relationships shown in Table 3 and Fig. 4 for RO water. For RO/Tap water there was an improvement by combining P_i and sediment concentration (SC) as follows: total P = 0.022 × P_i + 0.55 × SC + 1.01, $R^2 = 0.74$. The P_i would generally not be expected to be related to runoff total P concentration since P_i is a relatively small portion of the soil's total P concentration. Apparently in the RO/Tap runoff water, P_i accounted for some of the variability in total P runoff. It was not related to the total P concentration of the sediment in the runoff but was weakly related to the bulk soil's total P concentration (data not shown).

In none of the stepwise regression analyses was soil $NaHCO_3$–extractable organic P selected as contributing to the regression expressions, contrary to findings from a furrow erosion field study on the same plots from which soil was taken for our laboratory study (Westermann et al., 2001). The possibility exists, during the time we stored the soil, that some oxidation of organic matter took place, accounting for the difference.

Soluble P in runoff is related to P saturation of the sorption complex (Pote et al., 1999; Tunney et al., 1997). The sorption maximum may be estimated from the single-point phosphorus sorption index, PSI (Bache and Williams, 1971). We used stepwise regression to obtain the following significant relationships among PSI and other soil variables: for RO water PSI = 5.74 × lime − 2.13 × P_{ws} + 147, $R^2 = 0.82$; and for RO/Tap water PSI = 6.26 × lime − 0.818 × FeO-P_s + 158, $R^2 = 0.88$.

We calculated ratios between soil test P concentrations and the PSI, similar to the approach used by Pote et al. (1999), except that we did not adjust PSI for maximum sorption. We used these ratios as estimates of P saturation in regressing DRP against them. For illustration, with RO water we obtained the following statistically significant relationships with P_i: DRP = 0.648 × (P_i/PSI) + 0.00036, $r^2 = 0.71$, and with FeO-P_s: DRP = 0.755 × (FeO-P_s/PSI) − 0.066, $r^2 = 0.84$, improving the regression relationships from $r^2 = 0.40$ and $r^2 = 0.53$, respectively, when only soil test P concentrations were used in the regression calculations (Fig. 2).

Using RO/Tap water, corresponding relationships were: DRP = 0.384 × (P_i/PSI) + 0.0213, $r^2 = 0.54$, and DRP = 0.489 × (FeO-P_s/PSI) − 0.023, $r^2 = 0.54$. These regression relationships were worse than for RO water and were similar to those when only soil test P concentration was used in the regression calculation (Fig. 2).

We divided each runoff DRP and FeO-P_w concentration by its corresponding runoff volume (or depth of

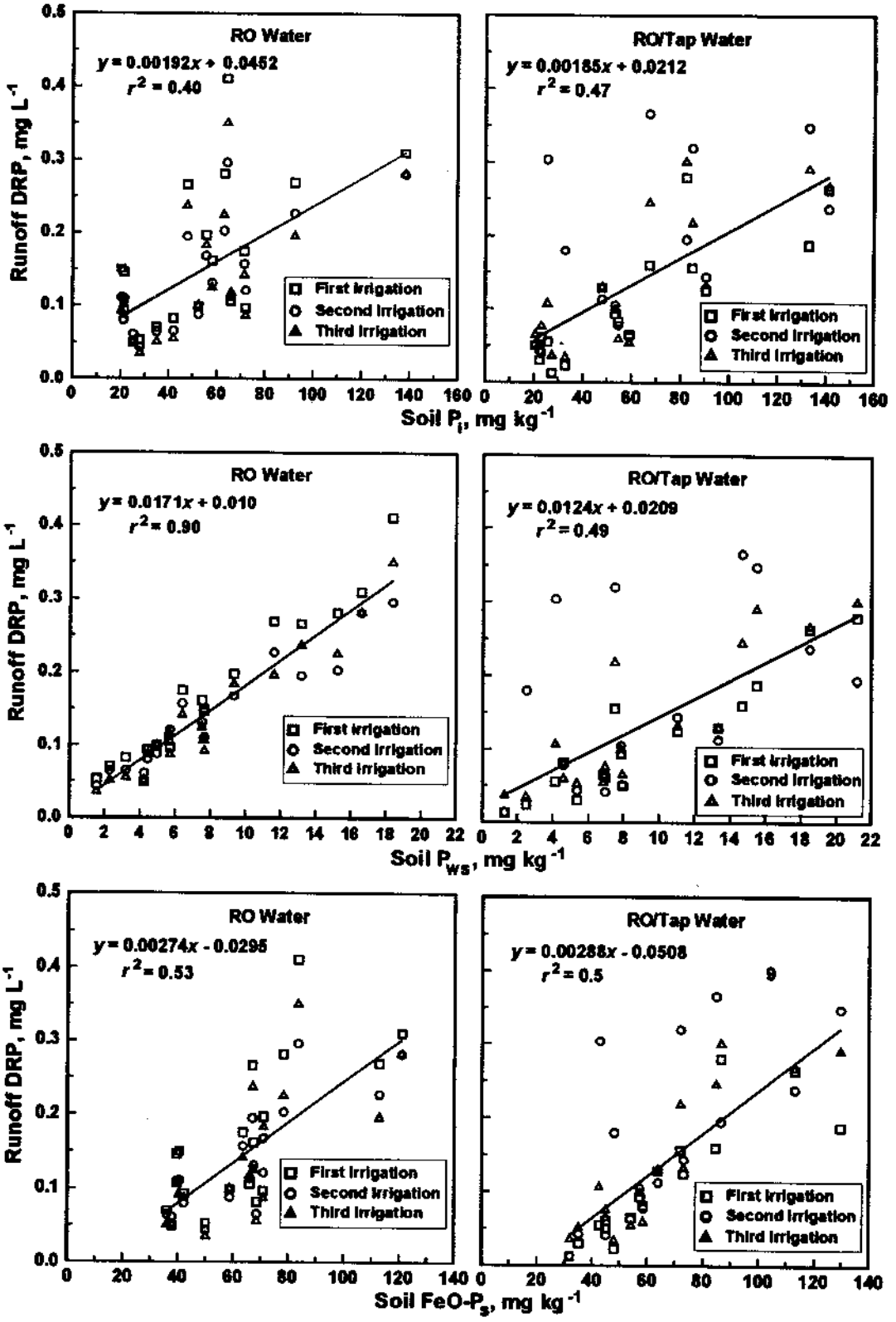

Fig. 2. Comparisons of dissolved reactive phosphorus (DRP) in reverse osmosis water (RO) and a 50:50 mix of RO water and tap water (RO/Tap) versus Olsen P (P_l), water-soluble P (P_{ws}), and iron-oxide impregnated paper–extractable P (FeO-P_s) soil tests. Regressions differ between RO and RO/Tap runoff waters for P_{ws}, but not for P_l or FeO-P_s. All regressions are significant ($P \leq 0.05$).

runoff). By so doing, in every case in our experiment, relationships between P concentration in runoff versus soil P tests deteriorated contrary to results obtained by Pote et al. (1999). Their field study was done on different soils and with different ranges of runoff and infiltration conditions compared with our box study. Their study

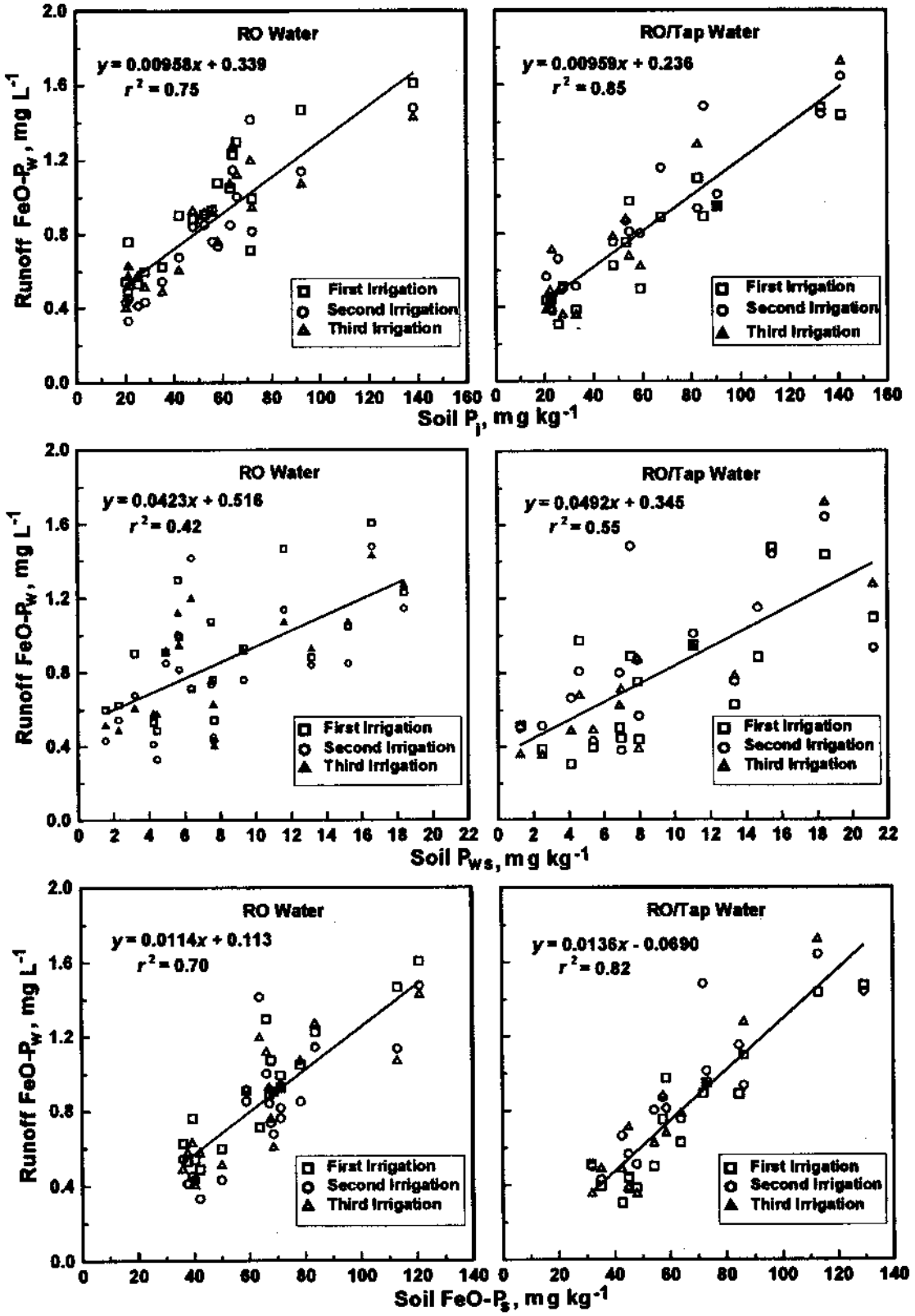

Fig. 3. Comparisons of iron-oxide impregnated paper–extractable phosphorus in runoff water (FeO-P_w) in reverse osmosis water (RO) and a 50:50 mix of RO water and tap water (RO/Tap) versus Olsen P (P_l), water-soluble P (P_{ws}), and iron-oxide impregnated paper–extractable P (FeO-P_s) soil tests. Regressions do not differ among RO and RO/Tap waters, except intercepts differ between RO and RO/Tap runoff waters for P_l. All regressions are significant ($P \leq 0.05$).

Table 3. Runoff total P versus three soil P tests. Water source was either reverse osmosis water (RO) or a 50:50 mix of RO and well water (RO/Tap).

Soil test	RO water		RO/Tap water	
Olsen P (P_i)	Total P = 0.006 × P_i + 6.00	r^2 = 0.02ns†	Total P = 0.030 × P_i + 3.45	r^2 = 0.48*
Water-soluble P (P_w)	Total P = 0.061 × P_w + 5.83	r^2 = 0.04ns	Total P = 0.155 × P_w + 3.71	r^2 = 0.29*
FeO-P_t‡	Total P = 0.005 × FeO-P_t + 6.00	r^2 = 0.01ns	Total P = 0.037 × FeO-P_t + 2.72	r^2 = 0.38*

* Significant at the 0.05 probability level.
† ns, not significant.
‡ Iron-oxide impregnated paper–extractable phosphorus in soil.

would be affected by site hydraulic factors whereas an attempt was made to minimize hydraulic factors in our laboratory study.

We also related P mass loss in runoff to soil P test concentrations, with mixed results. Relationships (r^2) for RO water were about the same as for DRP and FeO-P_w concentrations versus soil tests shown in Fig. 2 and 3. However, for RO/Tap water, P mass versus soil P test concentration relationships were worse than even those for normalized P concentration relationships (data not shown). The simple regression relationships displayed in our paper, therefore, generally stand as the best manner in which to report our results.

A moderate EC (ca. 2 dS m^{-1}) and low SAR are preferable in irrigation water to prevent soil dispersion, reduce soil erosion, and increase infiltration because of strengthened soil bonds and heightened retention of soil aggregates (Lentz et al., 1996). There was a trace of Na^+ (0.6 mg L^{-1}) in the RO water and no evidence of Ca^{++} and Mg^{++}, and therefore a low EC (0.02 dS m^{-1}) and an undefined SAR. By mixing RO and well water we added all three ions, with a resultant EC of 0.4 dS m^{-1}, a 20-fold increase compared with RO water, but nevertheless a low EC for irrigation water. By adding the divalent Ca^{++} (55 mg L^{-1}) and Mg^{++} (33 mg L^{-1}) ions contained in the well water, we obtained a SAR of 1.3. These numbers compare with a Snake River irrigation water EC of about 0.5 dS m^{-1} and a SAR of about 0.9. (Suitable irrigation water should have a SAR no higher than about 5.)

It appears that RO water caused greater soil dispersion than RO/Tap water since average soil loss and sediment concentration in runoff were greater in RO than in RO/Tap water runoff ($P = 0.05$), and average runoff across all three irrigations did not differ. This is indicated by the greater number of points falling above the 1:1 lines in the sediment relationships (Fig. 1B,C). Overall average runoff sediment concentrations for RO and RO/Tap runoff were 6.33 and 5.01 g L^{-1}, respectively.

The dispersive action on soil aggregates by RO water may affect particle size distribution in runoff sediment (Kim and Miller, 1996). We did not measure particle size distribution in runoff; however, it is conceivable that finer particle-size fractions were more prominent in RO runoff than in RO/Tap runoff. More divalent cations in RO/Tap than in RO water would encourage larger aggregates in the runoff. If this occurred, P losses could be potentially greater in RO than in RO/Tap water since finer soil particles and aggregates have higher P concentrations than larger soil particles and aggregates in sediments from Snake River irrigation water (Carter et al., 1974). Average total P was greater ($P = 0.05$) in RO than in RO/Tap water runoff, largely because of greater sediment concentration in RO runoff, because the sediment's total P concentration was the same for both water sources (data not shown). There were also no differences in sediment total P concentrations between subsequent irrigations for either RO or RO/Tap water (Fig. 4), implying that sufficient particle

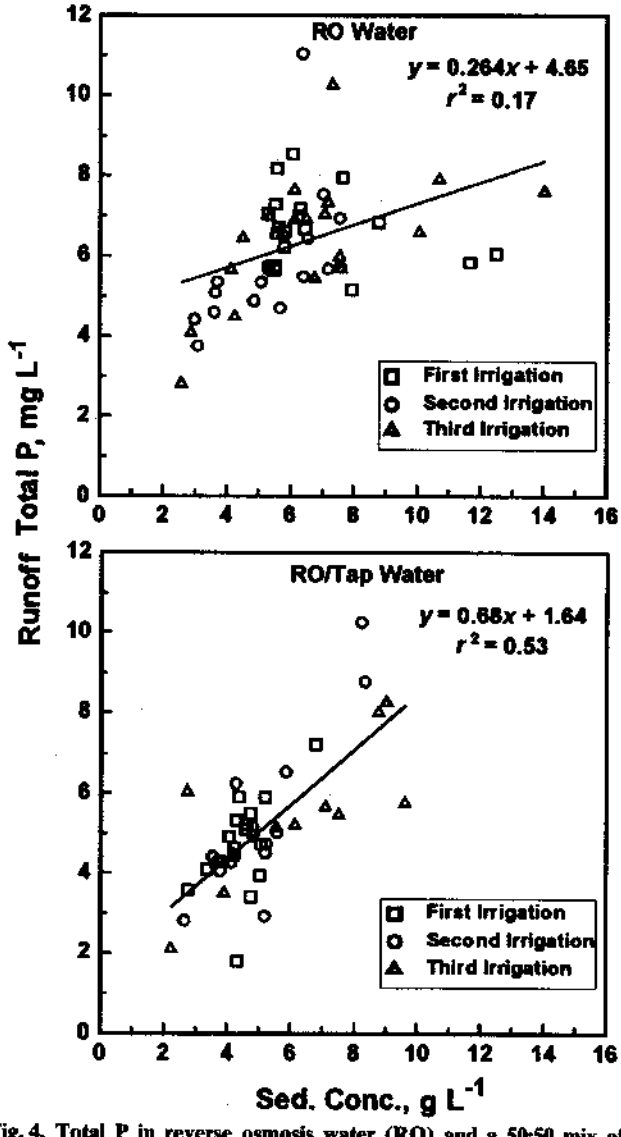

Fig. 4. Total P in reverse osmosis water (RO) and a 50:50 mix of RO water and tap water (RO/Tap) versus sediment concentration. Regressions are significant and they differ from each other ($P \leq 0.05$).

size sorting with repeated irrigations did not occur as reported in other studies (Sharpley et al., 1981). Droplet energy striking the soil surface was also the same for both water sources and irrigations because experimental conditions were identical. The better simple linear regression relationship between average total P and sediment concentration in runoff for RO/Tap than for RO water (Fig. 4) could imply that the soil in RO/Tap runoff was similar to the bulk soil.

Increasing ionic strength increases solubility of a slightly soluble salt via reduction in ion activity coefficients for a system at chemical equilibrium. However, the desorption and/or dissolution of P in soil–solution systems is considered to be predominantly controlled by kinetics rather than driven by chemical equilibria (Sharpley, 1983). Dissolved reactive P concentrations in runoff for both RO and RO/Tap water (Fig. 2) were much higher than those obtained from the furrow irrigation runoff study (0.007 to 0.045 mg P L^{-1}) on plots where soil for the laboratory study was taken (Westermann et al., 2001). Soil–water contact time in the laboratory study was limited to application duration (15 min) plus a few minutes to complete runoff and filtering (<5 min), suggesting that dissolution and/or desorption is very rapid. Exploratory laboratory studies showed that about 70% of the final DRP concentration was achieved in the first 15 min of soil–water contact (Westermann and Aase, unpublished data, 2000).

Increasing the EC (ionic strength) of the extracting solution decreases P desorption (Barrow and Shaw, 1979; Lehr and van Wesemael, 1952; Yli-Halla et al., 1995), particularly in acid soils. Increasing cation charge decreases desorption whereas increasing hydrated radii increases desorption (Barrow and Shaw, 1979). Based on chemical differences between RO and RO/Tap water, larger DRP concentrations in RO runoff might be expected. Even though the average DRP concentrations in RO and RO/Tap runoff were 0.149 and 0.129 mg L^{-1}, respectively, they were equivalent ($P = 0.05$). The saturated paste extract from the soil used in our study had an EC of 0.5 to 0.9 dS m^{-1} (Table 1). This is slightly higher than that of RO/Tap water (i.e., 0.4 dS m^{-1}). The dissolution of soluble Ca and Mg salts when RO water was added to the soil probably increased the EC of the applied RO water to approach that of RO/Tap water. These properties would tend to negate potential DRP runoff differences caused by initial ionic differences between RO and RO/Tap water. This implies that the specific response to water source depends on amount and kind of soluble salts present in the soil as well as on chemical characteristics of the applied water.

The best regression between average runoff DRP concentration and water soluble soil P (Fig. 2) probably occurred because the soil extractant (RO water) better simulated soil surface conditions during RO water application and runoff. Applying RO irrigation water would be similar to extracting the soil with RO water but at a higher solution to soil ratio. Substituting the ratio of soil test P concentration divided by PSI for the soil test concentration improved the regression relationships for RO runoff because the ratio does not depend on chemical properties of the soil extractant and because RO water is relatively free of soluble ions. Conversely, using the ratio in the DRP relationship for RO/Tap runoff did not improve the relationships because it does not account for possible chemical interactions of the RO/Tap water with the soils.

We compared extractable soil P from the test soils with either RO/Tap or RO water using the procedure given by Pote et al. (1996). The extractable P concentrations from the topsoil samples were similar (one-to-one) with both extractants; however, the extractable P was consistently greater (2 to 5 mg L^{-1}) with the RO/Tap extractant for the subsoil samples (data not shown). Runoff DRP in RO/Tap water was slightly less than or equivalent to runoff DRP in RO water (Fig. 2) in opposition to that found in our laboratory extraction test for the subsoils. We also could not successfully separate our runoff data (Fig. 2 and 3) into topsoil and subsoil, or into manure, whey, and conventional data groups. Both RO and RO/TAP water had similar effects on visible soil dispersion. This illustrates the difficulty of relating P extracted by laboratory procedures to P in runoff and emphasizes the need for further studies to resolve this predicament.

CONCLUSIONS

Average runoff was the same from RO and RO/Tap water, although the relationship between the two was not one-to-one. There was no predictive relationship between the two water sources for either soil loss or sediment concentration, nor were there differences between means for DRP and FeO-P_w in the runoff from the two water sources, although total P means differed. Neither DRP nor FeO-P_w were related to sediment concentration for either source of water, whereas for total P there was a reasonable relationship with sediment concentration for RO/Tap water but not for RO water, indicating two different populations (Fig. 4). The RO/Tap water total P relationship was significantly improved by adding both sediment and P_i ($NaHCO_3$ soil test P) concentrations via stepwise regression (R^2 = 0.74).

All relationships between runoff P concentrations and soil test P concentrations appear linear in our study. Dissolved reactive P concentrations at the lowest soil test P concentration were nearly 0.1 mg P L^{-1} for RO runoff and 0.05 mg P L^{-1} for RO/Tap runoff. These concentrations are considered sufficient to affect the eutrophication of receiving waters. Average FeO-P_w runoff concentrations generally exceeded 0.4 mg P L^{-1}, regardless of water source. Total P concentrations were all above 1 mg P L^{-1} in the runoff.

Water source and antecedent soil surface condition in our study had little effect on P in runoff from a calcareous soil. Water quality (chemistry) and possibly soil particle dispersion should be determined and considered before a decision is made about what water source to use in field studies of P runoff relationships. It may be impractical and expensive to use RO or distilled water as a source of water for field determinations of

P relationships to runoff and erosion. Therefore, if reasonably clean, acceptable water is available it may be used because it appears that a water source with similar chemical constituents to that of the soil will yield equivalent, reliable, and comparable results with that of RO water. Whether this conclusion holds for a wider range of soil and water conditions remains to be determined.

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