

doi:10.2489/jswc.68.6.450

## Assessment of phosphorus retention in irrigation laterals

J.A. Ippolito and N.O. Nelson

**Abstract:** Irrigation laterals transport irrigation return flow, including water, sediment, and dissolved nutrients, such as phosphorus (P), back to surface water bodies. Phosphorus transformations during transport can affect both P bioavailability and the best management practices selected to minimize P inputs to waters of the United States. The objective of this study was to determine P retention in three irrigation laterals. Soluble reactive P (SRP) concentrations in lateral waters were increased from 0.08 to 0.25 mg L<sup>-1</sup> (0.08 to 0.25 ppm) by constantly injecting a phosphate (PO<sub>4</sub>) solution for 2.5 hours. Bromide (Br) was used as a conservative tracer to determine dilution effects. Water was sampled at 10-minute intervals, beginning 30 minutes prior to injection and 120 minutes following injection, at one upstream location and various downstream locations to approximately 1,550 m (~1 mi) from injection sites. When at steady state, SRP concentrations only decreased by 5% over the lengths studied, equating to P uptake lengths of over 18 km (11.2 mi), which was one to two orders of magnitude greater than natural streams; the linear SRP uptake rate was 0.011 mg L<sup>-1</sup> km<sup>-1</sup> (0.018 ppm mi<sup>-1</sup>). Longer P uptake lengths and lower uptake rates in irrigation laterals, as compared to natural streams, may be due to the elevated sediment equilibrium P concentration, greater water velocities, and removal of vegetation causing a reduction in frictional resistance. Reducing water velocities should optimize irrigation lateral conditions to reduce uptake length and maximize P uptake.

**Key words:** coulee—irrigation lateral—retention—soluble reactive phosphorus—transport

**Aquatic resources in the United States are among its most valuable assets, yet approximately 40% of assessed stream miles, 45% of assessed lake acres, and 50% of assessed estuary acres are impaired (USEPA 2003).** The transport of excess nutrients is one of the leading causes of waterway impairment (USEPA 2003). In particular, soluble reactive phosphorus (SRP) transport in aquatic systems is of interest because phosphorus (P) is commonly regarded as the limiting nutrient governing primary production in fresh water systems (Foy 2005). Specifically, P transport in return-flow irrigation laterals may be a significant P source to river systems such as the Snake River in Idaho. Thus, determining and understanding the uptake distance SRP travels prior to being removed from solution during source-to-sink water movement could help identify whether current regional soil and water management strategies are having a sig-

nificant of an impact on water quality improvement as intended.

Soluble reactive P travel distance during water transport can be quantified in terms of P spiraling length, or the distance SRP travels as it cycles through dissolved to particulate (organic or inorganic forms) and back to the dissolved state (Davis and Minshall 1999). Total spiraling length is the sum of uptake length (distance traveled in the dissolved form) and turnover length (distance traveled in the particulate form). Uptake and turnover length are functions of downstream transport and transfer of P from the dissolved form, to being bound either organically or inorganically, and released to the water body again as SRP (Davis and Minshall 1999). Under base-flow conditions, research suggests that uptake length dominates total spiraling length (Mulholland et al. 1985).

Specifically, SRP uptake length is defined as the average distance a phosphate (PO<sub>4</sub>) ion travels in solution before being removed

by biological assimilation or sorption/precipitation to particulate form (Withers and Jarvie 2008). Shorter SRP uptake lengths are indicative of greater P retention efficiency and more rapid P cycling (Withers and Jarvie 2008). Furthermore, shorter P uptake lengths suggest that P is more likely to be a limiting nutrient in the stream ecosystem (Newbold et al. 1982). Thus, determining accurate estimates of P uptake length should be helpful in further understanding the transport and fate of P in waterways.

Because P uptake length is influenced by water velocity and discharge, the P mass transfer coefficient, or uptake velocity, can be used to enable nonbiased comparisons of P uptake across different water conveyance systems (Stream Solute Workshop 1990; Webster and Valett 2006). The uptake velocity is the theoretical speed at which a P molecule moves toward the stream bed (or other location of immobilization). Higher uptake velocities indicate stream systems with greater capacity to immobilize P.

As with natural systems, P transport and transformations in canal and lateral networks affect P bioavailability and transportability. Elaborate canal and lateral networks present in irrigated agroecosystems transport water and return flows over great distances prior to returning the water to a natural waterway. For example, the Twin Falls Canal Co. (Twin Falls, Idaho, United States) diverts up to 108 m<sup>3</sup> s<sup>-1</sup> (3,810 ft<sup>3</sup> s<sup>-1</sup>) through a 180 km (111 mi) canal network and 1,600 km (994 mi) lateral network. Irrigation return flow transported through this network contributes up to 150 kg (331 lbs) of dissolved P daily to the P-limited Snake River. An improved understanding of P transport in irrigation canals will help estimate management impacts on P loading from irrigation tracts.

Soluble reactive P transport in canal networks likely differs from that in natural water systems because canals have very low slope, straight channels, and periodic removal of bank vegetation. Subsequently, increased water velocities exist and, together with the aforementioned canal attributes,

**Jim A. Ippolito is a research soil scientist with the USDA Agricultural Research Service Northwest Irrigation and Soils Research Laboratory in Kimberly, Idaho. Nathan O. Nelson is an associate professor of soil fertility and nutrient management in the Agronomy Department at Kansas State University, Manhattan, Kansas.**

may transport SRP greater distances within canal networks as compared to natural systems. However, only a few studies have targeted SRP transport and uptake lengths in irrigation canals, and thus a gap in understanding exists. In an area receiving 1,800 mm (70 in) of annual rainfall, Barlow et al. (2006) modeled SRP flow in a 180 m (590 ft) surface irrigation drain that received pasture runoff. Taking into account time, distance, water velocity, and plant and sediment uptake and release during transport, the authors found that P uptake by bed sediment dominated when plant material was inconsequential within the drain. With the buildup of plant material, SRP uptake was either enhanced or hindered depending on the balance between plant P release and SRP uptake by bed sediments. However, Barlow et al. (2006) showed that, overall, SRP loads in the drain were only reduced by ~0% to 17%. Thus, the majority of SRP remained in the water column. Smith (2009) found that SRP injections into humid region (area receives ~900 mm [35 in] annual rainfall) drainage ditches required between 40 and 1,900 m (130 and 6,230 ft) to be removed. The author showed that uptake length was a function of ditch width, sediment properties, indirect drainage, and type of crop grown on surrounding land.

Our research objective was to determine P uptake lengths and uptake velocities in representative irrigation laterals within aridic systems and compare them with values for other natural and disturbed systems reported in the scientific literature. In this preliminary study, we aimed to fill the current knowledge gap with regards to aridic irrigation return flow P transport by utilizing controlled soluble P injections into several laterals.

## Materials and Methods

The Twin Falls irrigation tract is an 820 km<sup>2</sup> (316 mi<sup>2</sup>), aridic (USGS 2007) agricultural area located on the south side of the Snake River in Idaho (Bjorneberg et al. 2008) used primarily for alfalfa (*Medicago sativa*), corn (*Zea mays* L.), sugar beet (*Beta vulgaris* L.), potato (*Solanum tuberosum* L.), and spring and winter wheat (*Triticum aestivum* L.) production. All crops grown within the tract are irrigated because average annual precipitation is only 270 mm (10.6 in) (Bjorneberg et al. 2008). As previously mentioned, the Twin Falls Canal Co. diverts up to 108 m<sup>3</sup> s<sup>-1</sup> (3,810 ft<sup>3</sup> s<sup>-1</sup>) from the Snake River via a

180 km (111 mi) canal network and 1,600 km (994 mi) lateral network. Runoff from furrow irrigated fields and unused irrigation water are returned to the Snake River.

**Phosphorus and Bromide Injection.** Phosphorus uptake length was determined using a steady-state bromide (Br)/P injection method on September 4, 2004, and July 15, 2005, in lateral 22A (L22A) and August 23, 2005, in S1 coulee (S1) within the Twin Falls irrigation tract, Twin Falls County, Idaho, United States (figure 1). A solution of K<sub>2</sub>HPO<sub>4</sub> and KBr was injected for 2 to 2.5 hours for each injection (table 1) using a peristaltic pump attached to an injection manifold. The injection manifold was constructed of polyvinyl chloride pipe with six 1.6 mm (0.1 in) diameter holes on 20 cm (7.9 in) spacing. Injection occurred upstream of a suppressed rectangular sharp-crested weir at L22A and upstream of a riffle in S1 so that turbulence downstream of the injection would facilitate mixing of the solute with the stream water. The flow rate of the injection solution declined between 3.1% and 4.8% during the experiments based on measurements before and after the injection.

For the 2004 L22A injection (figure 2), water samples were collected at 10- to 15-minute intervals at four locations downstream of the injection point with automated water samplers for the duration of the injection and up to three hours following cessation of the injection (figure 1; figure 3). At the 184 m (604 ft) location, 24 grab samples were collected at the exact same time as automated sample collection to determine if there was any sample collection methodology bias. The sample collecting intensity of subsequent injections was reduced based on observations of the 2004 L22A injection. For the 2005 L22A injection, five manual grab samples were collected at each of five locations downstream of the injection point beginning 1.25 hours after injection commenced and continuing at approximately 10-minute intervals for 45 minutes (figure 1). In addition to the grab samples, two automated samplers collected samples at 6-minute intervals beginning 24 minutes after injection began and continuing for 1.25 hours after the injection stopped. For the 2005 S1 injection, five manual grab samples were collected at each of six locations downstream of the injection point beginning 45 minutes after injection commenced and continuing at approximately 20-minute intervals

for 1.25 hours (figure 1; figure 4). In addition to the grab samples, two automated samplers collected samples at 6- to 12-minute intervals beginning at injection and continuing for 1.25 hours after the injection stopped. A minimum of five grab samples were collected 3 m (9.8 ft) upstream of the injection point at 30-minute intervals beginning 30 minutes prior to injection and continuing until the end of injection for each of the solute injection studies.

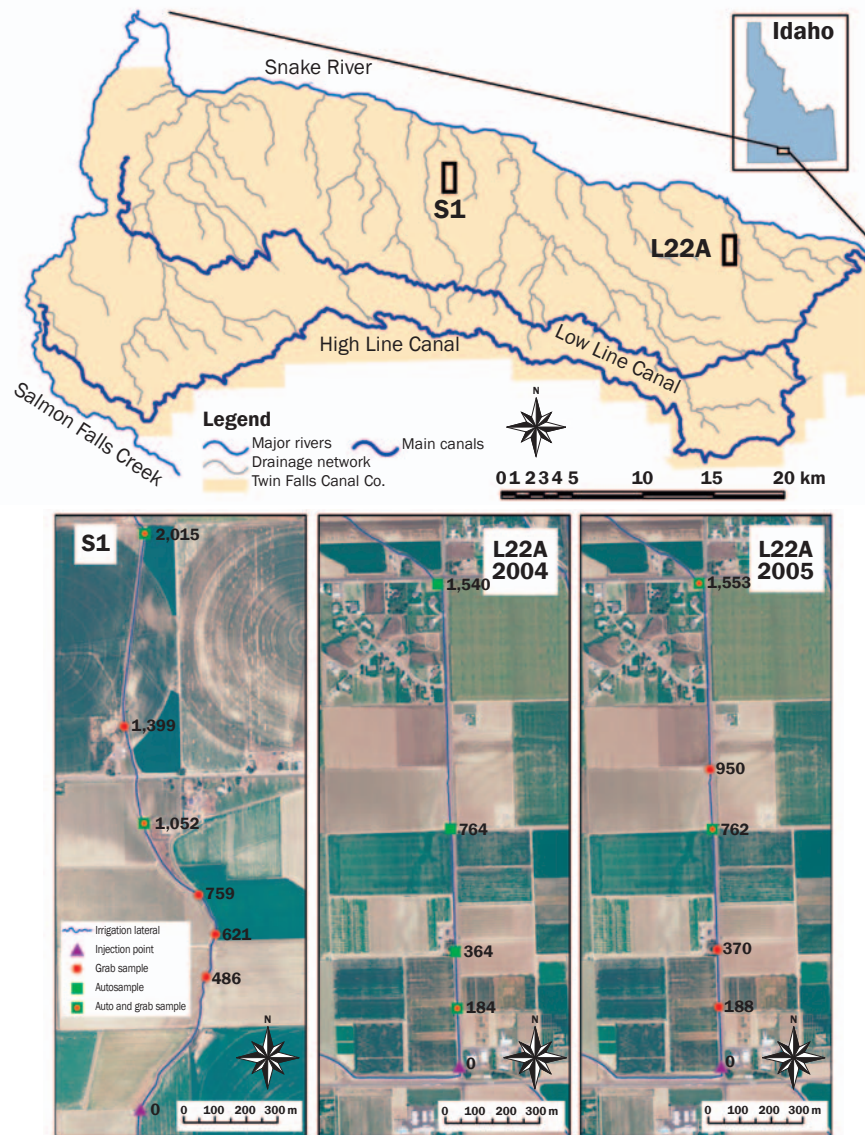
Samples from the automated samplers were filtered through a 0.2 μm (7.9 × 10<sup>-6</sup> in) filter within 6 hours of their collection. Grab samples were filtered immediately in the field through a 0.2 μm filter. Three unfiltered grab samples from each monitoring point were collected for sediment analysis. Analyses of paired samples showed no difference in dissolved P concentration between the grab samples and automated sampler samples. Background water samples were filtered through a 0.2 μm filter immediately in the field for determination of dissolved constituents, and unfiltered samples were retained for determination of electrical conductivity (APHA AWWA WEF 1992a), pH (APHA AWWA WEF 1992b), and total suspended solids (APHA AWWA WEF 1992c). Water temperature was measured in the field at 30-minute intervals.

All filtered samples were analyzed for SRP and Br using an automated flow-injection analysis system (Lachat QuikChem 8000) (Lachat Instruments 2000a, 2000b). Filtered samples from the July 15, 2005, injection in L22A were analyzed for soluble aluminum (Al), iron (Fe), calcium (Ca), magnesium (Mg), potassium (K), sodium (Na), and sulfur (S) concentrations using inductively coupled plasma optical emission spectroscopy. Water flow (table 1) during the injection period was determined by Br dilution. Flow was also monitored with a rectangular sharp crested suppressed weir at the injection point of L22A and at 759 m (2,490 ft) downstream of the injection point for S1 to verify there was not any change in flow during the studies. Additional channel characteristics are presented in table 1, and background water chemistry and sediment data are presented in table 2.

**Lateral Sediment Equilibrium Phosphorus Concentration.** Prior to the Br/P injection study, three sediment samples were collected from lateral bottoms along a 25 to 50 m (82 to 164 ft) stretch of L22A lateral at the

**Figure 1**

Location of solute injection studies including sample sites and distance from injection point to each sample location in meters (black numbers beside sample points).



1,540 m (5,053 ft) sampling location and S1 coulee at the 1,399 m (4,590 ft) sampling location. These locations were selected because they had lower water velocity and accumulated sediment. These locations contained sediments believed to be representative of the sediments that accumulate in the laterals. Sediment was allowed to settle and the supernatant removed. Six 3 g ( $6.6 \times 10^{-3}$  lb) subsamples, based on dry weight, were taken from each sample and placed in 50 mL (1.7 oz) centrifuge tubes with 25 mL (0.8 oz) of background solution containing 0.001M Ca, 0.001 M Na, 0.0006 M Cl, 0.0007 M Mg, and 0.0004 M S (similar to previously sampled L22A and S1 waters). Increasing quantities (0.000, 0.025, 0.075, 0.200, 0.500, 0.800 mL [0.000, 0.001, 0.002, 0.007, 0.017, 0.027 oz]) of a 10 mg L<sup>-1</sup> (10 ppm) P solution and three drops of toluene were added to the tubes, and the tubes were shaken end-to-end at 135 rpm for 16 hours. Tubes were then centrifuged for 10 minutes at 10,000 rpm after which a 20 mL (0.7 oz) aliquot was filtered through a 0.45 μm ( $1.77 \times 10^{-5}$  in) membrane filter. The solution was then analyzed for SRP with the molybdate-blue method (Murphy and Riley 1962) using a Spec 20 at 880 nm. The amount of P adsorbed to the sediment ( $P_{ads}$ , mg P kg soil<sup>-1</sup>) was determined by difference between the initial and final SRP concentrations, and the equilibrium P concentration at zero net P sorption ( $EPC_0$ ) was determined as the y-intercept of the regression of initial P concentration on  $P_{ads}$ . The  $EPC_0$  was used as a measure of the solution P concentration at which the sediment does not act as a P source or sink (Koski-Vähälä and Hartikainen 2001; House and Denison 2000).

**Water Chemistry—Sediment Mineral Precipitation Speciation.** In order to identify whether channel bottom sediment would

**Table 1**

Solute concentrations, injection rates, and sampling points for the solute injection experiments in lateral 22A (L22A) and S1 coulee (S1) of the Twin Falls irrigation district.

Lateral	Date	Solute concentration (mg L <sup>-1</sup> )		Solute injection rate (L s <sup>-1</sup> )	Injection duration (h)	C <sub>0</sub> (mg L <sup>-1</sup> )	Channel characteristics		
		P	Br				Flow (m <sub>3</sub> s <sup>-1</sup> )	Velocity (m s <sup>-1</sup> )	Width (m)*
L22A	Sept. 9, 2004	1,940	45,000	0.022	2.5	0.264	0.239	0.342	1.39
L22A	July 15, 2005	2,000	50,000	0.030	2.25	0.236	0.396	0.424	1.39
S1	Aug. 23, 2005	2,400	56,000	0.029	2.15	0.258	0.473	0.819	1.55

Notes: P = phosphorus. Br = bromide. C<sub>0</sub> = expected soluble reactive phosphorus (SRP) concentration in lateral water at injection point based on solute SRP, background lateral water SRP, solute injection rate, and lateral water flow.

\* Distance-weighted mean wetted channel width.



**Figure 2**

Solute injection in L22A in September of 2004.



**Figure 3**

Automated sampler placed at 364 m below the injection point in L22A during the September of 2004 injection study.



act as a source or sink of SRP, the aqueous geochemical speciation model PHREEQC (pH-Redox-Equilibrium model) for Windows was utilized (Version 2.17.00) (USGS 2010). The chemical saturation with respect to various Fe, Al, and Ca  $\text{PO}_4$  mineral phases of the L22A water collected on July 15, 2005, was evaluated using the model. Model input parameters included temperature; pH; and Al, Fe, Ca, Mg, K, Na, and S concentrations (table 2). Changes in water P chemistry were modeled prior to P injection using the background water P concentration, as well as after the injection of  $\text{K}_2\text{HPO}_4$ . Chloride (Cl) was added to the model for ion balance. The PHREEQC database was used for modeling and included strengite, vivianite, and hydroxyapatite. The database was modified to include monocalcium phosphate, octacalcium phosphate, tricalcium phosphate, and variscite based on equilibrium reactions outlined by Lindsay (1979).

**Phosphorus Uptake Length Determination.**

Phosphorus uptake length ( $S_w$ ) was calculated based on the longitudinal decrease in SRP downstream from the steady-state P injection point following computational methodologies described by Webster and Valett (2006). The  $S_w$  can be defined as

$$S_w = 1 \div k_w, \quad (1)$$

where  $k_w$  is the longitudinal uptake rate ( $\text{m}^{-1}$ ) (Webster and Valett 2006). Plateau P concentrations ( $C_x$ ) were corrected for background P concentration ( $C_b$ ) and normalized based on the conservative solute, Br ( $C_c$ ), to correct for dilution as follows:

$$C_N = \frac{(C_x - C_b)}{C_c}. \quad (2)$$

The normalized plateau P concentration ( $C_N$ ) is related to the downstream distance ( $x$ ) according to

$$C_N = C_{N0} e^{-k_w x}, \quad (3)$$

where  $C_{N0}$  is the P concentration at the injection point. The natural logarithm of Equation 3 produces the equation of a straight line with slope of  $-k_w$ :

$$\ln(C_N) = \ln(C_{N0}) - k_w x. \quad (4)$$

As shown in equation 4, a plot of the natural log of  $C_N$  at successive points in a stream channel versus  $x$  will yield a straight

**Figure 4**

View of S1 coulee looking upstream (south) at approximately 800 m downstream from the injection point.



**Table 2**

Irrigation water characteristics during solute injection experiments for lateral 22A (L22A) and S1 coulee (S1) of the Twin Falls irrigation district.

Variable	L22A	L22A	S1
Date	Sept. 7, 2004	July 15, 2005	Aug. 23, 2005
Temperature (°C)	16	25	19
EC (µS cm <sup>-1</sup> )	503	456	463
pH	8.6	8.7	8.7
SRP (mg L <sup>-1</sup> )	0.078	0.082	0.106
TSS (mg L <sup>-1</sup> )	45	220	688
EPC <sub>0</sub> (mg L <sup>-1</sup> )	ND	0.17	0.12
Al (mg L <sup>-1</sup> )	ND	0.001	ND
Fe (mg L <sup>-1</sup> )	ND	0.008	ND
Ca (mg L <sup>-1</sup> )	ND	45.92	ND
Mg (mg L <sup>-1</sup> )	ND	17.36	ND
K (mg L <sup>-1</sup> )	ND	6.286	ND
Na (mg L <sup>-1</sup> )	ND	22.39	ND
S (mg L <sup>-1</sup> )	ND	14.02	ND
P (mg L <sup>-1</sup> )	ND	0.082	ND

Notes: EC = electrical conductivity. P = phosphorus. SRP = soluble reactive P. TSS = total suspended solids. EPC<sub>0</sub> = equilibrium P concentration at zero net P sorption. Al = aluminum. Fe = iron. Ca = calcium. Mg = magnesium. K = potassium. Na = sodium. S = sulfur. ND = not determined.

line with slope of  $-k_w$  or  $-1 \div S_w$ . The average P concentration of automated sampler samples taken at the Br plateau ( $n = 11$  to  $13$ ) was used to determine  $C_x$  for the L22A 2004 injection study. The average P concentration of grab samples ( $n = 4$  to  $5$ ) was used to compute  $C_x$  for injections L22A in 2005 and S1 in 2005. Corresponding average Br concentrations were used to compute  $C_e$  and the average P concentration in grab samples collected upstream of the injection were used for  $C_b$ . Bromide concentrations in automated sampler samples were used to confirm that grab samples from L22A in 2005 and S1 in 2005 injection studies were taken during steady-state conditions and were accurate representations of plateau concentrations.

Phosphorus uptake velocity ( $v_f$ ) was calculated as

$$v_f = \frac{uz}{S_w}, \quad (5)$$

where  $u$  is the water velocity ( $m\ s^{-1}$ ) and  $z$  is the average water depth (m). Water velocity was determined as the time for Br concentration to reach half peak height divided by distance (Haggard et al. 2001a), and stream depth was determined as  $Q \div uw$ , where  $Q$  is water flow ( $m^3\ s^{-1}$ ) and  $w$  is the average channel width (m). Areal P uptake ( $U$ ) was determined as

$$U = v_f C_b. \quad (6)$$

Additional information on the derivation of  $S_w$ ,  $v_f$ , and  $U$  can be found in Webster and Valett (2006) and Stream Solute Workshop (1990).

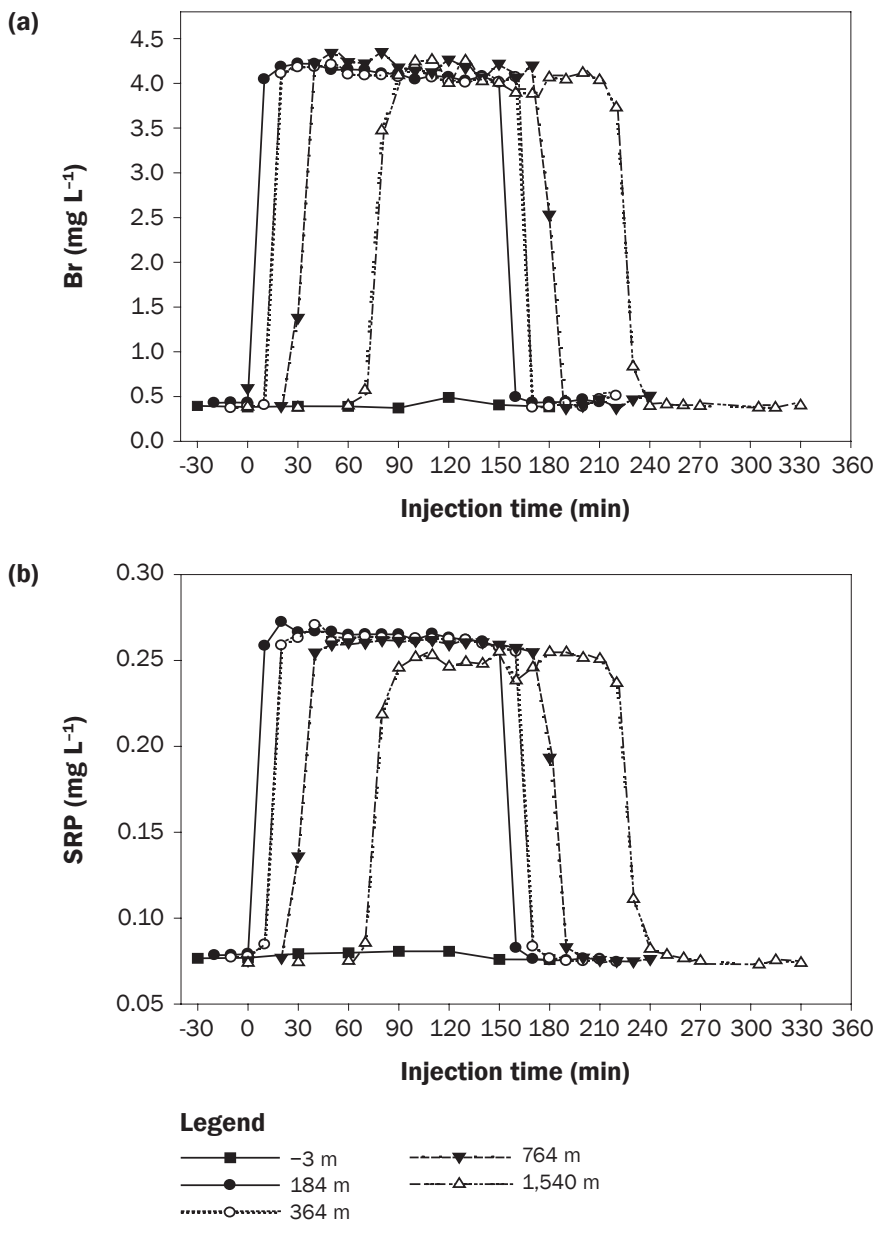
## Results and Discussion

**Phosphorus and Bromide Injection.** The Br and P pulse for all three laterals were similar. For ease of discussion, the L22A data from September 4, 2004, are presented (figures 5a and 5b). The Br tracer reached plateau concentrations in 10, 20, 40, and 90 minutes after injection at 184, 364, 764, and 1,540 m (604, 1,194, 2,506, and 5,051 ft), respectively. Longitudinal dispersion increased downstream, where Br concentrations increased from background to plateau concentrations in <10 minutes at locations <200 m (<656 ft) downstream and approximately 30 minutes at 1,540 m (5,051 ft) downstream. Findings similar to ours have been reported by others for use of Br as a conservative tracer (Harvey et al. 2003; Böhlke et al. 2004; Kellman 2004). There



**Figure 5**

(a) Bromide (Br) and (b) soluble reactive phosphorus (SRP) pulse over time downstream from solute injection, L22A lateral, September 7, 2004.



was not a significant decline in the average Br plateau concentrations with distance for the L22A 2004 injection ( $p > 0.7$ ), indicating that flow was constant with distance. This is typical because these laterals do not receive base-flow recharge as would natural streams. However, there was a significant decline in Br concentration for the L22A 2005 and S1 2005 injections ( $p < 0.05$ , data not shown). This was primarily due to tail-water inflow at 953 m (3,127 ft; L22A) and 1,200 m (3,937 ft; S1) downstream from injection points. Because of these inflows, the P concentrations were normalized based on Br concentration when computing P uptake length.

The P breakthrough with downstream distance was similar to changes in Br concentrations, showing a sharp increase at approximately the same time as Br breakthrough, followed by steady-state plateau. Based on injected solution P concentration, injection rate, and water velocity, the observed plateau P concentrations were only slightly less than the expected plateau SRP concentration of  $0.264 \text{ mg P L}^{-1}$  (0.264 ppm). The observed SRP (and Br) plateau concentration was similar for both sampling dates on L22A (data not shown), indicating a lack of hydrologic retention of waters in hyporheic storage zones.

**Lateral Sediment Equilibrium Phosphorus Concentration.** The decrease in SRP was minimal over the lateral lengths studied (figure 5), averaging  $0.011 \text{ mg L}^{-1} \text{ km}^{-1}$  ( $0.018 \text{ ppm mi}^{-1}$ ). Comparison between the SRP concentration and bed sediment  $\text{EPC}_0$  (table 2) suggests that bed sediment might possibly remove the injected SRP from the water (House and Denison 2000). The decline in SRP over distance was greatest for S1 (figure 6), which corresponded to a lower  $\text{EPC}_0$  than L22. Background P concentrations were relatively similar to  $\text{EPC}_0$  in S1. However, the background P concentrations in L22A were much less than the  $\text{EPC}_0$  of bed sediments, indicating that these sediments would likely be a source of P during undisturbed (non-enriched) conditions. It should be noted that the  $\text{EPC}_0$  of bed sediments from L22A were much more variable ( $0.13$  to  $0.23 \text{ mg L}^{-1}$  [ $0.13$  to  $0.23 \text{ ppm}$ ]) than the  $\text{EPC}_0$  measured in sediments from S1 ( $0.12$  to  $0.13 \text{ mg L}^{-1}$  [ $0.12$  to  $0.13 \text{ ppm}$ ]). Contrary to our results, Klotz (1988) found that SRP increased as stream bed sediment  $\text{EPC}_0$  values increased. However, other research in highly dynamic agricultur-

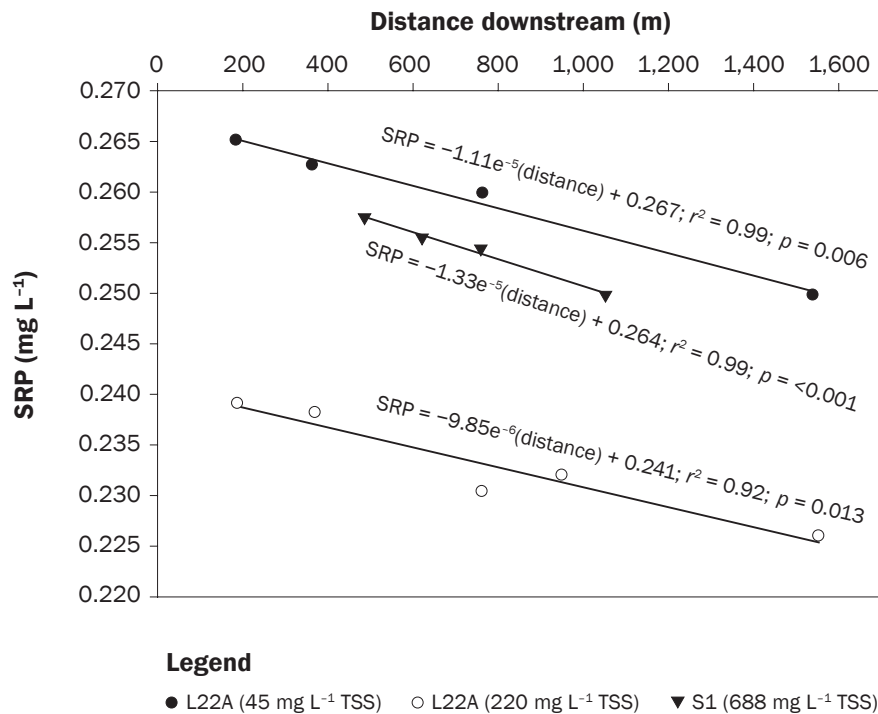
ally impacted stream systems found that stream water SRP was not always in equilibrium with sediment  $\text{EPC}_0$  (Agudelo et al. 2011).

Bed mineral species might also be controlling SRP removal from the irrigation lateral waters. Mineral saturation indices, based on PHREEQC modeling, are presented in figure 7. Prior to injection, L22A water was undersaturated (i.e., negative) with respect to most Fe, Al, and  $\text{Ca PO}_4$  mineral phases; preinjection waters were supersaturated (i.e., positive) with respect to hydroxyapatite.

The shift in saturation index following SRP injection, to include supersaturation with respect to octacalcium and tricalcium  $\text{PO}_4$ , was similar to that found by others (House and Denison 2000; Das et al. 2012). Under laboratory conditions, some researchers have suggested that  $\text{Ca PO}_4$  precursors crystallize prior to hydroxyapatite formation (Feenstra and de Bruyn 1979), while others have found rapid formation (~85 minutes) of hydroxyapatite (Wang et al. 2009). However, given the background P concentration in L22A

**Figure 6**

Soluble reactive phosphorus (SRP) concentration as a function of distance from the steady-state P injection location in the L22A lateral on September 7, 2004 (total suspended solids [TSS] = 45 mg L<sup>-1</sup>), and July 15, 2005 (TSS = 220 mg L<sup>-1</sup>), and in the S1 coulee on August 23, 2005 (TSS = 688 mg L<sup>-1</sup>).



(0.082 mg L<sup>-1</sup> [0.082 ppm]), the EPC<sub>0</sub> (0.17 mg L<sup>-1</sup> [0.17 ppm]), and the thermodynamic data found in PHREEQC, it is unlikely that background water P concentrations were controlled by bottom sediment Ca mineral species because PHREEQC predicted a P concentration of 1.104 × 10<sup>-6</sup> mg L<sup>-1</sup> (1.104 × 10<sup>-6</sup> ppm) for the preinjection model. Furthermore, Fe and Al mineral species would likely play a minor role in P precipitation given the background water pH of 8.6 to 8.7 (table 2), as at these pH values Fe and Al PO<sub>4</sub> mineral species would be assumed to be relatively unstable. Thus, bottom sediments likely do not aid in the short-term precipitation and removal of SRP.

**Phosphorus Uptake Length.** Uptake length for the three canal segments was 18.1, 47.1, and 41 km (11.2, 29.2, and 25.5 mi) for canal L22A in September of 2004, L22A in July of 2005, and S1 in August of 2005, respectively. These values were similar to that shown by Haggard et al. (2001b) and Marti et al. (2004) who studied wastewater treatment water discharge into streams and found SRP uptake lengths of 9 to 31 km (5.6 to 19.3 mi) and 0.1 to 14 km (0.06 to 8.7

mi), respectively. Doyle et al. (2003) studied P transport in a highly disturbed stream following dam rupture and found mean uptake lengths over 66 km (41 mi). Newbold et al. (1982) stated that shorter uptake lengths suggest a greater likelihood of nutrient limitation for a given stream reach, so it follows that the longer P uptake lengths in these laterals suggest that P is not a limiting nutrient within these systems. Longer SRP uptake lengths are indicative of lower P retention efficiency and slower P cycling. Lateral 22A and S1 converge with the nutrient-limited Snake River approximately 9.5 and 3.5 km (5.9 and 2.2 mi) downstream of the solute injection studies, respectively. Based on the very high uptake lengths determined in this study, P uptake would be relatively low between these reaches and the Snake River.

Because water velocity and discharge influence uptake length, uptake velocity is a preferred measure for comparison of P uptake between different stream systems. The uptake velocity of the irrigation laterals ranged from 6.1 to 9.5 × 10<sup>-6</sup> m s<sup>-1</sup> (2.0 to 3.1 × 10<sup>-5</sup> ft sec<sup>-1</sup>) (table 3). These uptake velocities are below the boundary for lower

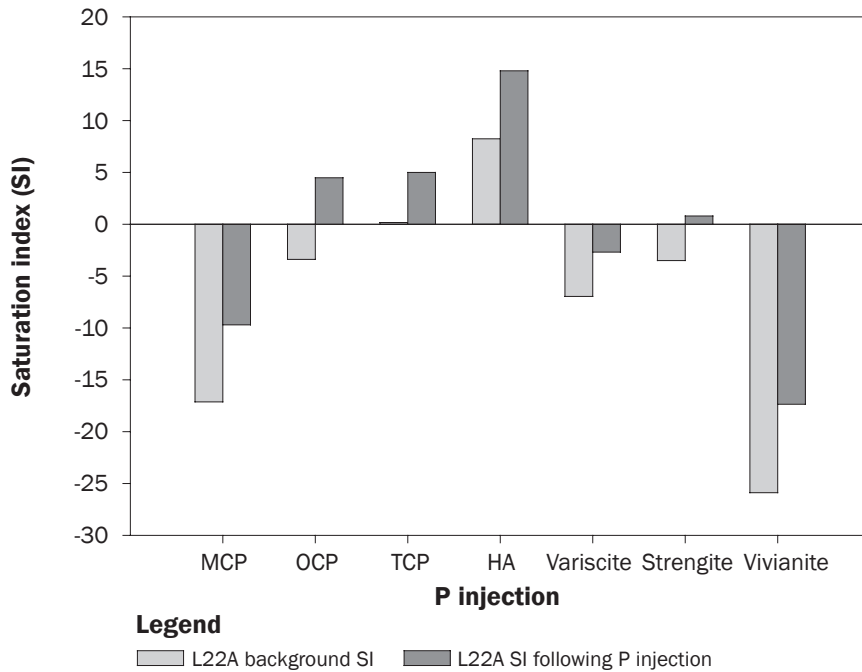
quartile of P uptake velocities from 194 published studies as compiled Ensign and Doyle (2006). Of the studies summarized by Ensign and Doyle (2006), our results compare most closely to the uptake velocities measured for a highly disturbed fifth-order stream where uptake velocities ranged from 0.3 to 18.2 × 10<sup>-6</sup> m s<sup>-1</sup> (0.1 to 6.0 × 10<sup>-5</sup> ft sec<sup>-1</sup>) (Doyle et al. 2003) (table 3). Based on these comparisons, P uptake in irrigation laterals is very low compared to natural stream systems.

Reduced P retention and cycling equates to greater SRP in irrigation return flow waters. Greater quantities of bioavailable P are exported from the tract with the potential to increase the risk of downstream eutrophication in standing waters of floodplain and estuarine ecosystems (Demars and Harper 2005). Unlike natural systems where episodic storm events periodically increase natural stream flow and thus is a period marked by elevated sediment and P transport (Ernstberger et al. 2004), the irrigation season on the Twin Falls tract is a continuous event. Ernstberger et al. (2004) describes this period in natural systems as hydrologically active with maximum P loss. This tends to occur without biological uptake and likely has a major impact on receiving waters.

The potential for irrigation return flow to have a major P impact on receiving water bodies is elevated compared to natural systems. This contention is supported by the P uptake lengths and P uptake velocities observed in the current study, which were one to two orders of magnitude greater than values reported for natural systems (table 3). Uptake lengths observed in this study were also greater than what has been reported for drainage ditches when using similar methods (table 3) (Smith 2009). Increased P uptake lengths were likely due to greater initial P concentrations, greater flow velocity, and removal of lateral vegetation causing reduced frictional resistance in the current study as compared to natural streams. McDowell et al. (2003) studied P dynamics in an eastern US catchment, showing that water P concentrations were not related to sediment P release during high stream (i.e., storm) flow rates. In our study and that of McDowell et al. (2003), greater flow rates could have removed fine bed sediment material caused by lateral-bed scouring and led to a reduced capacity for the bed sediment to adsorb P (Gainswin et al. 2006). This is supported by Klotz (1985), who showed that fine-grained streambed

**Figure 7**

PHREEQC (pH-Redox-Equilibrium model) saturation indices for L22A sampled on July 15, 2005, before and after injection of 2,000 mg phosphorus (P) L<sup>-1</sup> as monocalcium phosphate (MCP), octacalcium phosphate (OCP), tricalcium phosphate (TCP), and hydroxyapatite (HA). Negative values indicate undersaturation, while positive values indicate oversaturation with respect to the phosphate mineral phases presented.



sediments had higher P sorption indices as compared to coarser-grained stream sediments, and Klotz (1988), Davis and Minshall (1999), and Macrae et al. (2003), who suggested that streambed sediments were responsible for regulating SRP geochemically and abiotically. A greater P uptake rate, and thus reduced P uptake length, caused by smaller bed sediment particle size distributions associated with reduced flow rates has also been shown by Smith (2009).

Reduced stream flow rates have also been shown to increase average storage zone size and water retention time (Karwan and Sayers 2009; Legrand-Marcq and Laudelout 1985). Greater stream flow rates, as in our study, reduce the interaction between SRP and the hyporheic zone, reducing water retention and P storage zone size. Harvey et al. (1996) found similar results at elevated base flow rates (120 L s<sup>-1</sup> [75 gal s<sup>-1</sup>]), suggesting that the interaction between tracers were more sensitive to surface water storage processes than to hyporheic exchange.

The annual removal of vegetation from burning and mowing likely reduced lateral channel frictional resistance. Friction, due to plants, creates stagnant water zones that tem-

**Table 3**

Comparison of the phosphorus (P) uptake length determined for the current study with published results from undisturbed natural systems.

Water body type	P uptake length (m)	P uptake velocity ( $\times 10^{-6}$ m s <sup>-1</sup> )	Background DP (mg L <sup>-1</sup> )	Flow (m <sup>3</sup> s <sup>-1</sup> )	Source
Irr. lateral (L22A)*	18,090	9.5	0.078	0.24	Present study
Irr. lateral (L22A)	47,060	6.1	0.082	0.39	Present study
Irr. coulee (S1)	40,960	7.4	0.106	0.47	Present study
Ozark stream (Oklahoma)	117	NR	0.014	0.0043	Haggard and Storm 2003
Ozark stream (Oklahoma)	86	NR	0.016	0.0043	Haggard and Storm 2003
Agricultural stream (Oklahoma)	200 to 900	19.5 to 95.8†	0.029	0.13	Haggard et al. 2001a
Agricultural stream (Oklahoma)	257 to 339	75.2 to 84.0†	0.031	0.13	Haggard et al. 2001a
Mountain stream (Idaho)	370	121	0.005	0.088	Davis and Minshall 1999
Mountain stream (Idaho)	370	113	0.012	0.083	Davis and Minshall 1999
Drainage ditch (Indiana)	40 to 1,900	32 to 123	0.01 to 0.18	0.007 to 1.0	Smith 2009
Wastewater receiving creek (Arkansas)	9,000 to 31,000	NR	0.021 to 0.028	0.38 to 1.11	Haggard et al. 2001b
Wastewater receiving creek (Spain)	140 to 14,000	NR	0.01 to 1.0	0 to 0.11	Marti et al. 2004
Koshkonong River (Wisconsin)‡	4,140 to 367,000	0.3 to 18.2	0.1 to 0.32	2 to 23.5	Doyle et al. 2003

Notes: NR = not reported. DP = dissolved P.

\* Irrigation (Irr.) lateral L22A was sampled on Sept. 7, 2004, and July 15, 2005, while irrigation coulee S1 was sampled on Aug. 23, 2005.

† As reported by Ensign and Doyle 2006.

‡ Fifth order stream undergoing dynamic channel adjustment following dam removal.



porarily store solute (Harvey et al. 2003). The lack of friction likely reduces the interaction of solutes with the bottom sediments, the driving force for solute movement across the bed and temporary storage in the hyporheic zone (Packman and Bencala 2000). Thus, it is apparent that in order to reduce P uptake length the flow velocity needs to be reduced, as suggested by Barlow et al. (2003). This may be accomplished with the use of constructed ponds along channel pathways. Phosphorus uptake velocities could potentially be increased by enhancing the biological activity or altering the chemistry in the irrigation laterals (Leytem and Bjorneberg 2005).

### Summary and Conclusions

Phosphorus uptake lengths in irrigation laterals were similar to that found in other disturbed systems (>18 km [ $>11.2$  mi]) and one to two orders of magnitude greater than those observed in natural waterways. As compared to natural systems, P uptake lengths in irrigation laterals were elevated, likely due to greater initial P input concentrations and flow velocity. In addition, removal of vegetation caused reduced frictional resistance and less water interaction with streambank storage zones, further contributing to greater P uptake lengths. Dissolved P in laterals will not be greatly reduced prior to reentry into natural water bodies due to long uptake lengths and low uptake velocities. In order to reduce P in lateral waters to the maximum extent, it is suggested that remediation strategies that target a reduction in water velocity and an increase in frictional resistance be included in the Twin Falls irrigation tract.

### References

- Agudelo, S.C., N.O. Nelson, T.D. Keane, P.L. Barnes, and G.M. Pierzynski. 2011. Phosphorus adsorption and desorption potential of stream sediments and field soils in agricultural watersheds. *Journal of Environmental Quality* 40:144-152.
- APHA AWWA WEF (American Public Health Association, American Water Works Association, and Water Environment Federation). 1992a. Conductivity. *In* Standard Methods for the Examination of Water and Wastewater, ed. M.A.H. Franson, 43-2-47. Washington, DC: American Public Health Association, American Water Works Association, and Water Environment Federation.
- APHA AWWA WEF. 1992b. pH value. *In* Standard Methods for the Examination of Water and Wastewater, ed. M.A.H. Franson, 4\_65-4\_69. Washington, DC: American Public Health Association, American Water Works Association, and Water Environment Federation.
- APHA AWWA WEF. 1992c. Total suspended solids dried at 103-105°C. *In* Standard Methods for the Examination of Water and Wastewater, ed. M.A.H. Franson, 2\_56-2\_57. Washington, DC: American Public Health Association, American Water Works Association, and Water Environment Federation.
- Barlow, K., J. Fenton, D. Nash, and R. Grayson. 2006. Modelling phosphorus transport in a surface irrigation drain. *Advances in Water Resources* 29:1383-1398.
- Barlow, K., D. Nash, H. Turrall, and R. Grayson. 2003. Phosphorus uptake and release in surface drains. *Agricultural Water Management* 63:109-123.
- Bjorneberg, D.L., D.T. Westerman, N.O. Nelson, and J.H. Kendrick. 2008. Conservation practice effectiveness in the irrigated Upper Snake River/Rock Creek watershed. *Journal of Soil and Water Conservation* 63(6):487-495, doi:10.2489/jswc.63.6.487.
- Böhlke, J.K., J.W. Harvey, and M.A. Voytek. 2004. Reach-scale isotope tracer experiment to quantify denitrification and related processes in a nitrate-rich stream, midcontinent United States. *Limnology and Oceanography* 49:821-838.
- Das, J., S.H. Daroub, J.H. Bhadha, T.A. Lang, and M. Josan. 2012. Phosphorus release and equilibrium dynamics of canal sediments within the Everglades agricultural area, Florida. *Water Air Soil Pollution* 223:2865-2879.
- Davis, J.C., and G.W. Minshall. 1999. Nitrogen and phosphorus uptake in two Idaho (USA) headwater wilderness streams. *Oecologia* 119:247-255.
- Demars, B.O.L., and D.M. Harper. 2005. Water column and sediment phosphorus in a calcareous lowland river and their differential response to point source control measures. *Water Air Soil Pollution* 167:273-293.
- Doyle, M., E. Stanley, and J. Harbor. 2003. Hydrogeomorphic controls on phosphorus retention in streams. *Water Resources Research* 39:1147.
- Ensign, S.H., and M.W. Doyle. 2006. Nutrient spiraling in streams and river networks. *Journal of Geophysical Research-Biogeosciences* 111:G04009.
- Ernstberger, H., A.C. Edwards, and P.W. Balls. 2004. The distribution of phosphorus between soluble and particulate phases for seven Scottish East Coast rivers. *Biogeochemistry* 67:93-111.
- Feenstra, T.P., and P.L. de Bruyn. 1979. Formation of calcium phosphates in moderately supersaturated solution. *Journal of Physical Chemistry* 83:475-479.
- Foy, R.H. 2005. The return of the phosphorus paradigm: Agricultural phosphorus and eutrophication. *In* Phosphorus: Agriculture and the Environment, eds. J.T. Sims and A.N. Sharpley, 911-939. Madison, WI: American Society of Agronomy, Crop Science Society of America, and Soil Science Society of America.
- Gainswin, B.E., W.A. House, B.S.C. Leadbeater, P.D. Armitage, and J. Patten. 2006. The effects of sediment size fraction and associated algal biofilms on the kinetics of phosphorus release. *Science of the Total Environment* 360:142-157.
- Haggard, B.E., D.E. Storm, and E.H. Stanley. 2001b. Effect of a point source input on stream nutrient retention. *Journal of American Water Resources Association* 37:1291-1299.
- Haggard, B.E., D.E. Storm, R.D. Tejral, Y.A. Popova, V.G. Keyworth, and E.H. Stanley. 2001a. Stream nutrient retention in three northeastern Oklahoma agricultural catchments. *Transactions of the American Society of Agricultural Engineers* 44:597-605.
- Harvey, J.W., M.H. Conklin, and R.S. Koelsch. 2003. Predicting changes in hydrologic retention in an evolving semi-arid alluvial stream. *Advances in Water Resources* 26:939-950.
- Harvey, J.W., B.J. Wagner, and K.E. Bencala. 1996. Evaluation the reliability of the stream tracer approach to characterize stream-subsurface water exchange. *Water Resources Research* 32:2441-2451.
- House, W.A., and F.H. Denison. 2000. Factors influencing the measurement of equilibrium phosphate concentrations in river sediments. *Water Research* 34:1187-1200.
- Karwan, D.L., and J.E. Saiers. 2009. Influences of seasonal flow regime on the fate and transport of fine particles and a dissolved solute in a New England stream. *Water Resources Research* 45:W11423.
- Kellman, L. 2004. Nitrate removal in a first-order stream: Reconciling laboratory and field measurements. *Biogeochemistry* 71:89-105.
- Klotz, R.L. 1985. Factors controlling phosphorus limitation in stream sediments. *Limnology and Oceanography* 30:543-553.
- Klotz, R.L. 1988. Sediment control of soluble reactive phosphorus in Hoxie Gorge Creek, New York. *Canadian Journal of Fisheries and Aquatic Sciences* 45:2026-2034.
- Koski-Vähälä, J., and H. Hartikainen. 2001. Assessment of the risk of phosphorus loading due to resuspended sediment. *Journal of Environmental Quality* 30:960-966.
- Lachat Instruments. 2000a. Determination of orthophosphate by flow injection analysis colorimetry. Quikchem Method 10-115-01-1-B. Loveland, CO: Lachat Instruments.
- Lachat Instruments. 2000b. Bromide in water. Quikchem Method 10-135-21-2-A. Loveland, CO: Lachat Instruments.
- Legrand-Marcq, C., and H. Laudelout. 1985. Longitudinal dispersion in a forest stream. *Journal of Hydrology* 78:317-324.
- Leytem, A., and D. Bjorneberg. 2005. Removing soluble phosphorus in irrigation return flows with alum additions. *Journal of Soil and Water Conservation* 60(4):200-208.
- Lindsay, W.L. 1979. *Chemical Equilibria in Soils*. New York, New York: John Wiley & Sons.
- Macrae, M.L., M.C. English, S.L. Schiff, and M.A. Stone. 2003. Phosphate retention in an agricultural stream

- using experimental additions of phosphate. *Hydrological Processes* 17:3649-3663.
- Marti, E., J. Aumatell, L. Gode, M. Poch, and F. Sabater. 2004. Nutrient retention efficiency in streams receiving inputs from wastewater treatment plants. *Journal of Environmental Quality* 33:285-293.
- McDowell, R.W., A.N. Sharpley, and G. Folmar. 2003. Modification of phosphorus export from an eastern USA catchment by fluvial sediment and phosphorus inputs. *Agriculture, Ecosystems & Environment* 99:187-199.
- Mulholland, P.J., J.D. Newbold, J.W. Elwood, and L.A. Ferren. 1985. Phosphorus spiraling in a woodland stream: Seasonal variations. *Ecology* 66:1012-1023.
- Murphy, J., and J. P. Riley. 1962. A modified single solution method for the determination of phosphate in natural waters. *Analytica Chimica Acta* 27:31-36.
- Newbold, J.D., J.W. Elwood, R.V. O'Neill, and A.L. Sheldon. 1982. Nutrient spiraling in streams: Implications for nutrient limitation and invertebrate activity. *American Naturalist* 120:628-652.
- Packman, A.I., and K.B. Bencala. 2000. Modeling surface-subsurface hydrologic interactions. In *Streams and Ground Waters*, eds. J.A. Jones and P.J. Mulholland, 45-80. San Diego, CA: Academic Press.
- Smith, D.R. 2009. Assessment of in-stream phosphorus dynamics in agricultural drainage ditches. *Science of the Total Environment* 407:3883-3889.
- Stream Solute Workshop. 1990. Concepts and methods for assessing solute dynamics in stream ecosystems. *Journal of the North American Benthological Society* 9:95-119.
- USEPA (US Environmental Protection Agency). 2003. National management measures to control nonpoint source pollution from agriculture. EPA 841-B-03-004. Washington, DC: US Environmental Protection Agency. <http://www.epa.gov/owow/nps/agmm/index.html>.
- USGS (US Geological Survey). 2007. Drought watch. Where do droughts usually occur? Sioux Falls, SD: US Geological Survey. <http://nd.water.usgs.gov/drought/faqs/faq4.html>.
- USGS. 2010. PHREEQC (version 2) – A computer program for speciation, batch-reaction, one-dimensional transport, and inverse geochemical calculations. Version 2.17.0. [http://wwwbrr.cr.usgs.gov/projects/GWC\\_coupled/phreeqc/index.html](http://wwwbrr.cr.usgs.gov/projects/GWC_coupled/phreeqc/index.html).
- Wang, C.G., J.W. Liao, B.D. Gou, J. Huang, R.K. Tang, J.H. Tao, T.L. Zhang, and K. Wang. 2009. Crystallization at multiple sites inside particles of amorphous calcium phosphate. *Crystal Growth & Design* 9:2620-2626.
- Webster, J.R., and H.M. Valett. 2006. Solute dynamics. In *Methods in Stream Ecology*, Second edition, eds. F.R. Hauer and G.A. Lamberti, 169-185. Burlington, MA: Academic Press.
- Withers, P.J.A., and H.P. Jarvie. 2008. Delivery and cycling of phosphorus in rivers: A review. *Science of the Total Environment* 400:379-395.