Nutrients in Runoff from a Furrow-Irrigated Field after Incorporating Inorganic Fertilizer or Manure

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Use of dairy manure to supply crop nutrients is gaining broader acceptance as the cost of fertilizer rises. However, there are concerns regarding manure's effect on water quality. In 2003 and 2004, we measured sediment, NO_3–N, NH_4–N, K, dissolved reactive P (DRP), and total P (TP) concentrations in runoff from furrow irrigated field plots (6–7 irrigations yr^{-1}) cropped to corn (*Zea mays* L.) in the semiarid climate of southern Idaho. Annual treatments included 13 (Year 1) and 34 Mg ha^{-1} (Year 2) stockpiled dairy manure (M); 78 (Year 1) and 195 kg N ha^{-1} (Year 2) inorganic N fertilizer (F); or control–no amendment (C). Available N in manure applied each year was similar to amounts applied in fertilizer. Constituent concentrations (mg L^{-1}) in runoff ranged widely among all treatments: sediment, 10 to 50,000; NO_3–N, 0 to 4.07; NH_4–N, 0 to 2.28; K, 3.6 to 46.4; DRP, 0.02 to 14.3; and TP, 0.03 to 41.5. Over both years, fertilizer and manure treatments increased irrigation mean values (averaged across irrigations) for NO_3–N runoff concentrations (M = 0.30, F = 0.26, C = 0.21 mg L^{-1}) and mass losses (M = 0.50, F = 0.42, C = 0.33 kg ha^{-1}) relative to the control. Over both years, the manure treatment also increased mean irrigation runoff concentrations of DRP (M = 0.19, F = 0.09, C = 0.08 mg L^{-1}) and K (M = 1.13, F = 0.79, C = 0.62 mg L^{-1}) compared with fertilizer and control plots. Average DRP and K runoff mass losses were 2.0 to 2.4 times greater in manure treatments than in control plots. Neither F or M affected season-long cumulative infiltration. Runoff DRP and inorganic-N losses appeared to be influenced more by the timing of the amendment application and environmental conditions than by the quantity of nutrients applied. Nutrient additions to furrow irrigated soils, whether from fertilizer or manure, can potentially increase nutrient losses in irrigation runoff, with the degree of impact depending on the nutrient, amount, and timing of application and whether inorganic fertilizer or manure was applied.

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Irrigated cropland produces a large share of the total crop value in the United States. Of the U.S. irrigated acreage, furrow irrigation is used on about one-quarter, or 5 million ha (USDA, 1998). Although furrow irrigation provides several important advantages over other irrigation methods, an important consequence is that surface runoff, which is commonly tolerated to improve water application uniformity along the furrows, is permitted to leave the field (Lehrsch et al., 2005). The water discharged from surface-irrigated fields can enter natural surface waters via return flow and is a potential source of contamination and diffuse (nonpoint source) pollution. In the United States, the water quality of return flows was recognized as a management concern early in 1970 (Bondurant, 1971; Law and Skogerboe, 1972), but more recently it has come to the fore in other surface-irrigated regions of the world (McHugh et al., 2008; Monaghan et al., 2009). The water quality issue related to agricultural irrigation and drainage remains one of the most difficult challenges facing agricultural and engineering professionals (Tanji and Keyes, 2002).

Irrigation runoff from cropped fields can transport beneficial materials applied onsite to offsite environments where they may generate negative ecological consequences. These materials include sediment, organic carbon, salts, nutrients such as nitrate nitrogen (NO_3–N), ammonium nitrogen (NH_4–N), potassium (K), and phosphorus (P), trace elements, pesticides, and microorganisms (Bondurant, 1971; Turner et al., 1980; Bjorneberg et al., 2002; Tanji and Keyes, 2002; Causapé et al., 2004).

Sediment concentrations in runoff from recently tilled furrow-irrigated fields commonly are 1000 to 10,000 mg L^{-1} (Berg and Carter, 1980). The transported sediment and associated organic matter are an important source of N and P (Heathwaite and Johnes, 1996), which play a dominant role in the eutrophication of both freshwater and ocean ecosystems (Correll, 1998). Bjorneberg et al. (2006) reported that runoff from furrow-irrigated fields contained mean dissolved reactive P (DRP) concentrations ranging from 0.04 to 0.10 mg L^{-1} and total P (TP) ranging from 0.3 to 12.5 mg L^{-1}, with the latter being linearly related to runoff suspended sediment.

Runoff from a flood-irrigated hay meadow contained median reactive P concentrations of 0.53 to 18.12 mg L^{-1} and NH_4–N

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Abbreviations: DRP, dissolved reactive phosphorus; EC, electrical conductivity; ICP–OES, inductively couple plasma–optical emission spectrometry TMDL, total maximum daily load; TP, total phosphorus.

concentrations of 0.11 to 3.46 mg L^{-1}, depending on the timing of a broadcast monoammonium phosphate fertilizer application (White et al., 2003). Reactive P in runoff from these flood-irrigated fields was greater than for furrow because in the former, irrigation water flows over the entire field area (and hence exposure to broadcast P fertilizer is maximized), whereas in furrow only a fraction of the soil surface is contacted by water. Ultimately, 1.1 to 18% of applied reactive P and 0.1 to 3.3% of applied ammonium N was lost from the fertilized experimental plots in irrigation runoff, with greater amounts lost on more recently fertilized plots (White et al., 2003). Flood-irrigated pastures fertilized with superphosphate produced the greatest P losses in the first irrigation (Austin et al., 1996), and runoff losses decreased with time between fertilizer application and the first irrigation (Bush and Austin, 2001).

Cessna et al. (2001) reported that nutrient concentrations in major surface drainage ditches of a Saskatchewan surface irrigation district averaged (i) 0.03 to 0.93 mg L^{-1} for TP when the irrigation source water included only 0.017 mg L^{-1}; (ii) 0.007 to 0.035 mg L^{-1} reactive P relative to 0.003 mg L^{-1} in source water; and (iii) 0.012 to 0.044 mg L^{-1} nitrate N compared with 0.031 mg L^{-1} present in source water. The researchers concluded that 2.2% of TP and 1.9% of inorganic N (ammonium and nitrate) applied as fertilizer was lost in irrigation runoff.

Little research has evaluated the effect of manure amendments on nutrient losses in runoff from surface-irrigated crops, although manure effects on nutrients in runoff resulting from actual or simulated rainfall events are well documented (Cabrera et al., 2009; Kleinman et al., 2002; Little et al., 2005; Smith et al., 2007; Gilley et al., 2007). Mundy et al. (2003) evaluated the effect of defoliation and cow stocking density on P and N in runoff from a flood-irrigated perennial pasture. One day after defoliation (cutting or grazing), flow-weighted filterable P and TP concentrations in runoff were 1.7 and 2.1 mg L^{-1}, respectively, for a 375 cows ha^{-1} stocking-rate (short-term) treatment, while those for the lowest stocking-rate treatment (0 cows) were 1.5 and 1.5 mg L^{-1} (Mundy et al., 2003). No treatment differences in runoff P were observed by 8 d after defoliation–grazing. Westermann et al. (2001) measured P in runoff from furrow-irrigated plots having varying fertilizer and manure treatments; however, their data was limited in that (i) they monitored only a single irrigation in each of 2 yr; (ii) furrow length, inflow rates, and irrigation periods used were not typical for irrigation in the area; (iii) manure treatments were applied to plots at least 4 yr before their study; and (iv) they did not represent fields with growing crops. Neither Bjorneberg et al. (2006) nor Westermann et al. (2001) monitored inorganic N or K in furrow streams.

During furrow irrigation, the applied water streams interact with a fraction of the soil surface that would be exposed to a rainfall or flood-irrigation event. The objective of this study was to measure season-long runoff nutrient and sediment losses from furrow-irrigated corn (*Zea mays* L.) fields in semiarid southern Idaho, which had been treated with either inorganic fertilizer or manure. We hypothesized that the disposition of nutrient losses in furrow-irrigation runoff would differ depending on the nutrient source used.

Materials and Methods

Site and Soils

The experimental site was established in fall 2002 on furrow-irrigated Portneuf silt loam (coarse-silty, mixed, superactive, mesic Durinodic Xeric Haplocalcids) with 1.5% slopes near Kimberly, ID. The surface soil is a silt loam and contains on average 100 g kg^{-1} clay, 700 g kg^{-1} silt, 10 to 13 g kg^{-1} organic carbon, and 5% calcium carbonate equivalent. The soil has a saturated-paste-extract electrical conductivity (EC) of 0.07 S m^{-1}; exchangeable sodium percentage of 1.5; pH of 7.7 (saturated paste); and a cation exchange capacity of 19 $cmol_c$ kg^{-1}. Cropping on the site consisted of an alfalfa (*Medicago sativa* L.)–corn–bean (*Phaseolus* L.)–grain rotation for the previous 33 yr. Between 1969 and 1986, the field received dairy manure applications every 3 yr (40–75 Mg ha^{-1} dry wt.), but none was applied between 1986 and 2002. In the cropping year before this study (2002), the field was fertilized with 135 kg N ha^{-1} as urea and planted to silage corn. Plots were planted to silage corn in each season during the 2-yr study.

We aligned the experimental plots with previously installed soil water percolation samplers. This permitted us to make simultaneous leaching observations, which will be reported in a separate article.

Experimental Design

The experimental design was a randomized complete block with three replicates. The three treatments included (i) a control (no nutrient additions); (ii) dairy manure applied at a locally typical rate (13 or 34 Mg ha^{-1} dry wt.); and (iii) conventional inorganic fertilizer, applied at a rate that provided an amount of N similar to that of the manure amendment (described below). Manure and fertilizer amendments are described in detail in Table 1. Each experimental unit (i.e., plot) was 4 m wide by 57 m long and was separated from adjacent plots by a 1.3-m-wide buffer strip (Fig. 1). Each plot included five rows of corn planted on a 0.76-m-row spacing and four irrigation furrows. In a typical irrigation event, every other furrow was watered. However, during periods of exceptionally high potential evapotranspiration, all furrows were watered to maintain adequate soil moisture. The buffer strip included one irrigated furrow. We normally monitored and sampled runoff water from one furrow in each plot. On days when all four furrows per plot were irrigated, however, two furrows in each plot were monitored and results were reported as an average of the two.

Field Operations

After silage corn harvest in fall 2002, the field was disked to 0.1-m depth. Stockpiled solid manure from dairy cattle (*Bos* sp.) was applied at 13 Mg ha^{-1} (dry wt.) to designated plots on 10 Oct. 2002 using a commercial spreader truck equipped with rooster-comb beaters. Three 1.6- by 2.4-m tarps were placed at random locations in each block to collect manure and quantify its application rate. We weighed, mixed, and subsampled the manure intercepted by each tarp, then returned it to the soil surface where it had been collected. All applied manure was immediately incorporated with a disk to 0.1-m depth, and later that fall the entire field was chisel plowed. Preemergence herbicide

Table 1. Amendment nutrient concentrations, bulk and nutrient application (appl.) rates (all on a dry wt. basis), and time of application.

Crop year		Stockpiled dairy manure										Inorganic fertilizer			
		Amendment properties and nutrient concentration													
		Solids	C:N	C	N†	Ca	K	Mg	P	Mn	Zn	Form	C	N	Na
		kg kg^{-1}		g kg^{-1}						mg kg^{-1}			g kg^{-1}		
2003	Manure	0.56	16.2	302	18.6	34.9	27.2	10.8	7.8	260	193	Urea	200	460	–
2004	Manure	0.40	16.0	160	10.0	55.7	14.1	8.9	3.8	248	85	NaNO3	–	160	270
		Field bulk amendment or nutrient application rates													
		Appl. date	Manure (solids)	C	N†	Ca	K	Mg	P	Mn	Zn	Form	Appl. date	Appl. Rate Fertilizer	Appl. Rate N
			Mg ha^{-1}		kg ha^{-1}									kg ha^{-1}	
2003	Manure	10 Oct. 2002	13	3.92	242	454	354	140	102	3.4	2.5	Urea	6 May 2003	169	78
2004	Manure	24 Mar. 2004	34	5.44	340	1894	479	302	129	8.4	2.9	$NaNO_3$	12 May 2004	1219	195

† N = Total N.

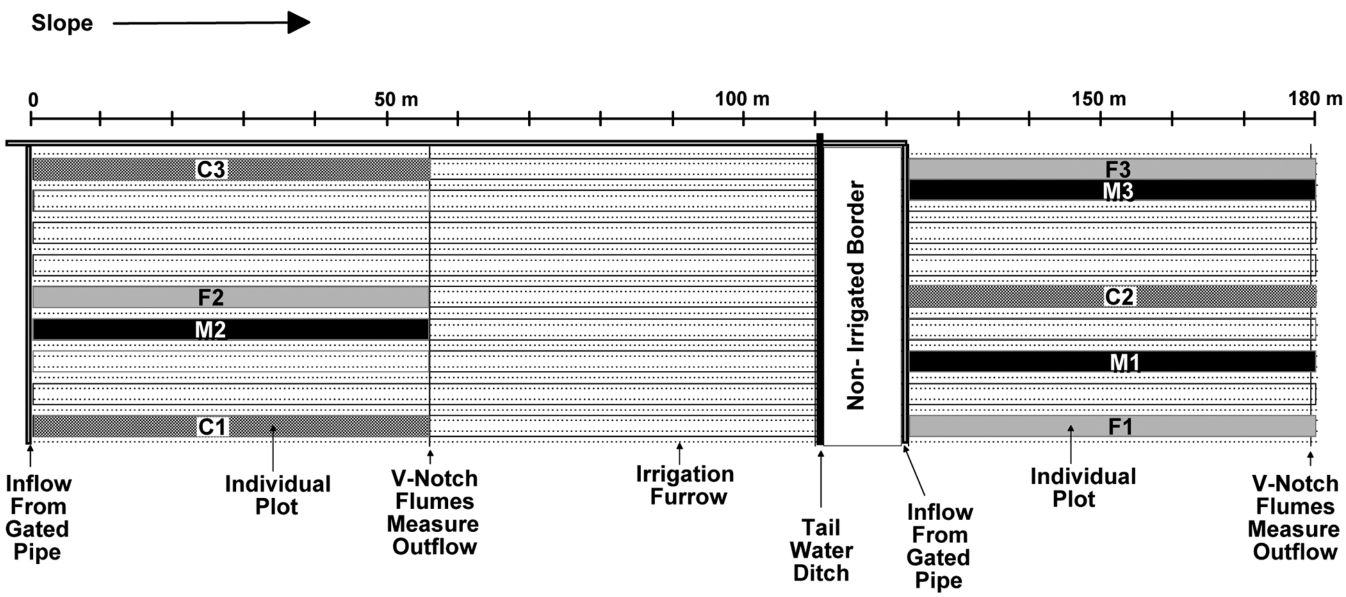

Fig. 1. Diagram of experimental plot layout. (C1 = control, Rep 1; F2 = fertilizer, Rep 2; M3 = manure, Rep 3; etc.)

was applied to the entire area in spring 2003, followed by roller harrow tillage to incorporate. On 6 May 2003, urea fertilizer at 78 kg N ha^{-1} was applied with a drop spreader on designated plots and incorporated with a roller harrow.

Silage corn was planted on 15 May 2003. The V-shaped, 0.1-m-deep furrows were formed as an integral part of the planting operation. Furrows were formed using a weighted V-shaped tool attached to the toolbar and aligned with the tractor wheels. All monitored irrigation furrows were wheel trafficked when formed in the field to reduce furrow infiltration variability (Yoder et al., 1996). The field was cultivated in the first week of July during each growing season to control weeds and simultaneously reshape the irrigation furrows. After silage harvest in mid-September, the remaining corn stover (15- to 30-cm-tall stems with leaves) was incorporated by disking to 0.1 m.

In early March 2004, the field was moldboard plowed to 0.25-m depth, followed by two roller harrow passes. Manure was applied to designated plots on 24 March 2004 at a rate of 34 Mg ha^{-1} (dry wt.) and incorporated with an offset disk and roller harrow. The manure application rate was increased in 2004 because we expected that the spring manure would contain less solids and have a higher C-to-N ratio (due to higher straw content) than the fall manure and because the 2002 fall manure application was slightly smaller than was typical for the area. Preemergence herbicide was applied on 31 March and incorporated with a roller harrow. On 12 May 2004, sodium nitrate at a rate of 195 kg N ha^{-1} was applied with a drop spreader over fertilizer plots and incorporated using a roller harrow. The sodium nitrate fertilizer was derived from a Chilean source and had a unique isotopic signature, which we used to track nitrate leaching in the soils (unpublished data). Silage corn was planted on 13 May 2004. Other field operations were the same in 2004 as in 2003.

Standing corn silage yields were measured in both years from a midfield location in each plot. Two 3-m lengths of the planted corn row were collected at each location, one from either side of a treated irrigation furrow.

Irrigation

The Snake River water used for irrigation had an average EC of 0.04 S m^{-1}, sodium adsorption ratio of 0.6, and carried little sediment (<500 mg L^{-1}). A gated pipe with adjustable spigots conveyed irrigation water across the plots at the head, or inflow-end, of the furrows. At the head of each plot, a manifold made from 0.15-m-diameter polyvinyl chloride pipe withdrew water from the gated pipe and directed it under equal hydrostatic pressure into irrigation furrows of each experimental plot. Irrigation outflows from each furrow entered a tail-water ditch that ran perpendicular to the furrows at the bottom of the plots. Seven irrigations on 14-d intervals were applied to plots each year beginning on 10 June 2003 and 15 June 2004.

Irrigations began between 0730 and 0830 h and ran for 24 h. Inflows typically were set to 13.25 L min^{-1}. Furrow inflow rates, furrow stream outflow rates, and sediment concentrations were measured during each irrigation (described below). Outflow rates were measured and runoff water samples were collected to determine sediment concentrations at half-hour intervals early in the irrigation, every hour during the midirrigation period, and every 2 h thereafter, when irrigation outflows and sediment loads had stabilized (typically after 7 h or more into the set). Approximately 11 to 12 h into the irrigation set (~1930 h), a final irrigation measurement was made for the day. Monitoring was resumed at 0600 to 0730 h the next morning. Previous monitoring experience has shown that the late evening and early morning readings provided a reasonably accurate mean flow and infiltration measurements for the overnight period. Inflows were measured by timing the filling rate of a known volume, and outflows were measured with long-throated V-notch flumes.

Water Quality Sampling and Analyses

In addition to the water samples taken for sediment determinations, three to four additional runoff water samples per irrigation were collected for nutrient analysis. The water was collected from the runoff monitoring flumes located at the outflow end of the furrow. Runoff nutrient concentrations were monitored in all but the seventh (last) irrigation in 2004. Since runoff nutrient concentrations tend to be lowest in the last irrigations of the season (Brown, 1985), there was little likelihood that a significant nutrient loss event would go unrecorded by omitting sampling in irrigation 7. Four runoff samples per furrow were collected during irrigations 1 and 2 in 2003, at 5 min, 1 h, 4 h, and either 10 or 22 h after furrow advance (i.e., after runoff began). In the remaining irrigations in 2003 and 2004, runoff samples were collected at 5 min, 1 h, and 5 h after furrow advance. Sediment and nutrient concentrations in furrow streams commonly peaked between 1 and 3 h after furrow advance and generally declined to a lower level by 5 h after advance (Lentz et al., 2001). Irrigation inflow samples were also collected periodically during irrigations to determine nutrient background concentrations. A subsample of inflow and runoff samples was taken and filtered through a 0.45-μm Millipore membrane (Billerica, MA). Water samples were stabilized with a saturated H_3BO_3 solution (1 mL per 100-mL sample) and stored at 4°C until analyzed. We determined TP in the unfiltered samples by persulfate digestion (American Public Health Association, 1992) and analyzed for DRP (Watanabe and Olsen, 1965); nitrate-nitrogen (NO_3–N) and ammonium-nitrogen (NH_4–N) using flow injection analysis and colorimetric methods (Mulvaney, 1996); and K by inductively couple plasma–optical emission spectrometry (ICP–OES).

The collected manure samples were subsampled to determine solids content with the remainder air dried at about 30°C. A composite sample of the air-dry manure mass was ground in a Thomas Wiley mill (Swedesboro, NJ) to pass an 865-μm screen, freeze dried to remove moisture, and then analyzed on a Thermo-Finnigan FlashEA1112 CNS analyzer (CE Elantech Inc., Lakewood, NJ) to determine total C and N. An elemental analysis was conducted on a portion of each freeze-dried sample, which was dry ashed, digested with nitric acid, and analyzed on an Optima 4300 DV ICP–OES (PerkinElmer Instruments, Waltham, MA).

Calculations and Statistical Analysis

The inorganic fertilizer amounts added each year were intended to furnish an N-rate equivalent to that of the supplied manure. We calculated the available N from manure assuming that the manure's total N equaled 1.5% of the total manure dry mass, and that 40% of the manure total N became available during the growing season. These percentage values were average values for manures in south-central Idaho (A. Leytem, personal communication, 2002). Subsequent chemical analyses (Table 1) indicated that the manure N value calculated as described above represented a reasonable average for the manures we applied each year.

The mass of sediment per liter of sampled furrow runoff was determined from the settled volume of sediment in an Imhoff cone, which was converted to a mass value via a calibration function (Lentz et al., 1992). The computer program WASHOUT fitted calibration functions and calculated net infiltration and runoff sediment losses for furrows (Lentz and Sojka, 1995). Individual calibration functions were developed for each year of irrigation, type of furrow (freshly cultivated or previously irrigated), and treatment. WASHOUT computed the net infiltration volume for individual furrows by subtracting the total outflow volume from the total inflow volume, where inflow and outflow volumes were computed by integrating the inflow- and outflow-rate curves over time. The net infiltration depth (i.e., infiltration volume per unit area) was then calculated by dividing the net infiltration volume by the field area watered by the irrigation furrow. The watered area was the product of irrigation furrow spacing and the furrow length. Infiltration as a fraction of irrigation inflow (infiltration fraction) was calculated as 100 times the ratio of net furrow infiltration divided by net inflow.

Reported sediment and nutrient concentrations and values used in mass-loss computations were adjusted for inflow concentrations, so furrow losses represented only those losses resulting from treatments. The exception to this rule was when we reported median, mode, and range values for nutrient concentrations in collected inflow and outflow samples. Furrow sediment and nutrient losses were computed by WASHOUT, which calculated sediment and nutrient loads in furrow stream outflows and integrated component losses over the duration of the irrigation. Cumulative TP, DRP, NO_3–N and NH_4–N mass losses per irrigation were computed with the assumption that runoff constituent concentrations remained constant between sampling intervals. Mean sediment and nutrient concentrations for a given furrow and irrigation were computed as the ratio of total mass loss divided by total outflow volume. Season-long cumulative values for irrigation parameters and component losses were also computed.

Data from each irrigation in 2003 and 2004 were analyzed via analysis of variance (ANOVA), PROC Mixed (SAS Institute, 2008) using a repeated measures approach, which accounted for correlations between a response variable's values measured from one irrigation to the next. The model included treatment, year, and irrigation as fixed effects, and block with its associated interactions as random effects. Response variables (runoff nutrient concentrations and losses for individual irrigations) were transformed using common log or square root to stabilize variances and improve normality. Means and 95% confidence intervals were back transformed to original units for reporting. Studywide median, mode, minimum, and maximum values for sediment and nutrient concentrations (nontransformed) in inflow and runoff samples were determined using PROC Univariate (SAS Institute, 2008). An ANOVA on season-long cumulative values and crop yields was conducted using PROC Mixed to determine the effect of treatment and year on each parameter. The model included treatment and year as fixed effects with block and its associated interactions as random effects. Analyses of season-long cumulative values did not require transformation of responses.

Finally, stepwise multiple regression analyses using the PROC REG procedure (SAS Institute, 2008) described the relationships between transformed runoff nutrient concentration values and predictor variables: mean runoff sediment concentration, treatment, irrigation date (day of year), infiltration fraction, and mean outflow rate. Predictor variable values used in the regression analyses were the means for each irrigation in each year.

All analyses were conducted using a $P = 0.05$ significance level.

Results and Discussion

Water applied during irrigation events in 2003 and 2004 far exceeded any that occurred as a result of rainfall (Fig. 2). Cumulative precipitation at the study site was only 249 and 232 mm in 2003 and 2004, compared with cumulative applied irrigations of 1520 and 1621 mm. Most of the precipitation occurred during the fall, winter, and spring months and was received at intensities <7 mm h^{-1}. In 2003 and 2004, only one rainfall event exceeded 7 mm h^{-1}, a 24 mm h^{-1} event that occurred on 3 Aug. 2003. We observed no evidence that the precipitation events produced runoff from the plots, even for the 24 mm h^{-1} event. During this high-intensity rainfall event, runoff was unlikely because the corn crop had attained full canopy cover, surface soils were dry before the event, and water intake rates of dry Portneuf soil typically exceed 25 mm h^{-1} during the first 30 min of a water infiltration event (Rasmussen and Cary, 1979).

The ANOVA reported in Table 2 examined data for individual irrigations across both years. Overall, treatment main effects did not significantly influence furrow infiltration or runoff concentrations of sediment, NH_4–N, or TP. However, treatments did significantly affect K, NO_3–N, and DRP runoff concentrations and mass losses (Table 2) and significantly influenced sediment and TP runoff mass losses for specific irrigations, as indicated by the significant treatment × irrigation interaction shown in Table 2. All hydraulic and nearly all runoff component concentration and mass loss variables were affected by a significant interaction between irrigation number and year (Table 2). Furthermore, while the treatment × year interaction had little effect on response variables (except K), the treatment × irrigation × year interaction effects were significant for sediment, K, NO_3–N, and DRP regarding both runoff concentrations and mass losses.

Furrow Inflows, Outflows, and Infiltration Fractions

Total furrow inflow per irrigation did not differ among treatments (Table 2) and consistently contributed about 220 mm irrigation water per set. A few irrigation inflows varied from this, most notably, irrigations 1 (250 mm) and 2 (165 mm) in 2003 and irrigation 5 (278 mm) in 2004. Irrigation infiltration fraction was not affected by treatment (Table 2). However, it did vary between years, being greater in 2003 than in 2004 (32% vs. 25%). Furthermore, the manner in which infiltration fraction changed from irrigation to irrigation differed between years (Table 2), as a result of varying soil and weather conditions and inflow amounts for individual irrigations. Since mean furrow outflow rates were inversely related to

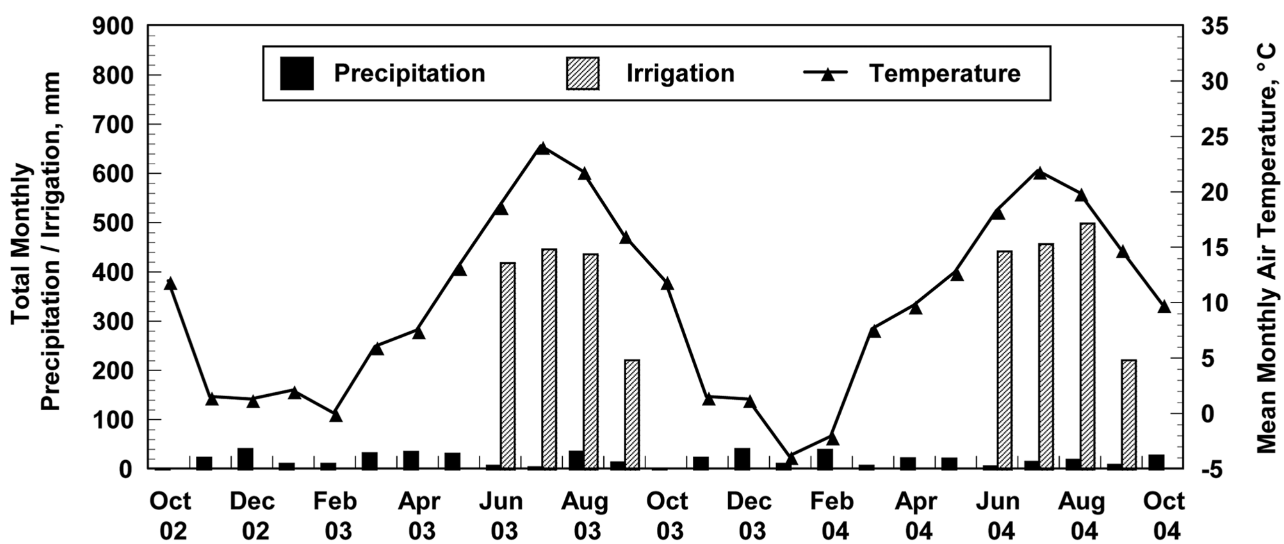

Fig. 2. Total monthly precipitation and irrigation amounts and mean monthly air temperature at the study site from Oct. 2002 through Oct. 2004.

Table 2. The influence of fertilizer treatment, irrigation number, and year on furrow hydraulic parameters, component concentrations, and cumulative component losses in irrigation furrow runoff. The table gives *P* values for main effect and interaction terms derived from an analysis of variance.

Source of variation	*P* values															
	Total inflow	Inf. frac†	Outflow rate	Advance time	Mean irrigation runoff concentrations‡						Cumulative mass losses in each irrigation‡					
					Sed	K	NO_3–N	NH_4–N	DRP	TP	Sed	K	NO_3–N	NH_4–N	DRP	TP
Treatment (TRT)	0.7	0.6	0.7	0.5	0.4	*	*	0.9	***	0.9	0.4	*	*	0.8	***	0.8
Irrigation no. (IRR)	***	**	***	***	***	***	***	0.07	***	***	***	***	***	0.06	***	***
Year (Y)	***	**	***	***	0.1	***	***	*	***	0.5	*	**	***	**	*	*
TRT × IRR	0.6	0.9	0.8	0.9	0.07	***	0.5	0.7	***	0.06	*	***	0.2	0.8	**	*
TRT × Y	0.4	0.2	0.3	0.8	0.11	**	0.8	0.7	0.3	0.06	0.2	*	0.7	0.9	0.2	0.08
IRR × Y	***	**	***	***	***	***	***	***	***	***	***	***	***	***	***	***
TRT × IRR × Y	0.8	0.2	0.1	0.2	*	***	*	0.6	**	0.1	*	***	**	0.4	**	0.1

* $P < 0.05$.

** $P < 0.01$.

*** $P < 0.001$.

† Inf. frac. = infiltration fraction, calculated as 100 times the ratio of net furrow infiltration divided by net inflow.

‡ DRP = dissolved reactive phosphorus; Sed = sediment; TP = total phosphorus.

infiltration, outflow data tracked those of the infiltration fraction. Furrow outflow rates ranged from 6.8 to 11.2 L min^{-1}. This variation resulted in part from deviations in irrigation inflow amounts but primarily was due to changes in furrow infiltration caused by differences in furrow stream-wetted perimeter, flow velocity, and maximum sediment concentration (Trout, 1992; Trout et al., 1995). Since treatments did not alter irrigation runoff (i.e., mean outflow rate, Table 2), any treatment effects on sediment and nutrient mass losses observed in this study largely resulted from the influence of treatment on runoff sediment and nutrient concentrations.

Sediment and Nutrient Losses

In general, all treatments produced similar runoff sediment concentrations and mass losses. The exceptions occurred in irrigations 2, 4, and 6 in 2004. In these few instances, fertilizer and/or control treatments produced greater runoff sediment concentrations (Fig. 3B) and mass losses (Fig. 4B) than the manure treatments.

When evaluated on a per-irrigation basis across both years, NO_3–N concentrations in furrow runoff were 1.3 times greater in the fertilizer and manure than in control plots (Table 3). Accordingly, in any given irrigation, cumulative NO_3–N mass losses for fertilizer and manure were on average 1.4 times control values (Table 3). Furthermore, in runoff the manure DRP concentrations were 2.4 times and manure K concentrations were 1.8 times those of the control, which resulted in similar increases in DRP and K mass losses compared to the control (Table 3). Our NO_3–N findings and results from other studies (Austin et al., 1996; White et al., 2003; Sharpley and Smith, 1995) suggest that adding inorganic P or K nutrients to plots at rates similar to that provided by the manure, would result in similar increases in P or K runoff concentrations and mass losses for both treatments relative to the control.

Despite the greater fertilizer and manure application rates in 2004, component runoff concentrations and mass losses were generally greater in 2003 (except for K) (Table 3). Two factors may have contributed to this result. First, fall 2002 manure contained greater N than did spring 2004 manure (Table 1); therefore, even though 2.6 times more 2004 manure was applied than 2002 manure, the former contributed only 1.4 times more total N. Second, summer temperatures in 2004

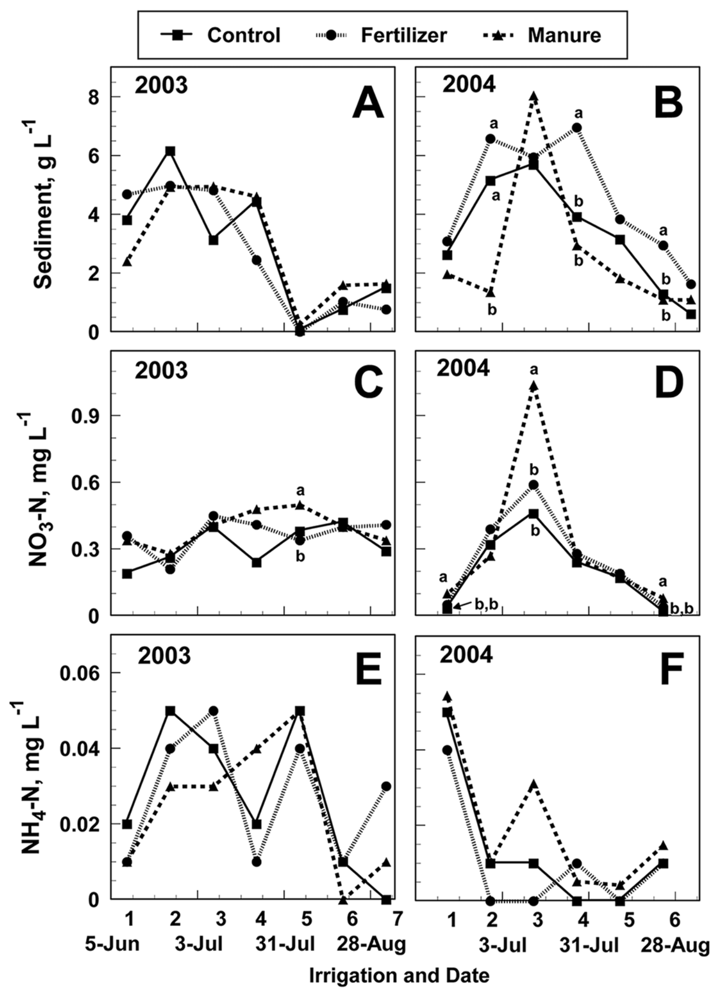

Fig. 3. Runoff (A, B) sediment, (C, D) NO_3–N, and (E, F) NH_4–N as affected by year (2003 on left vs. 2004 on the right), irrigation number, and treatment. Values are derived from irrigation means. Treatment means for a given irrigation and year are significantly different at $P < 0.05$ if labeled with a different letter. Absence of letters indicates no significant difference among means.

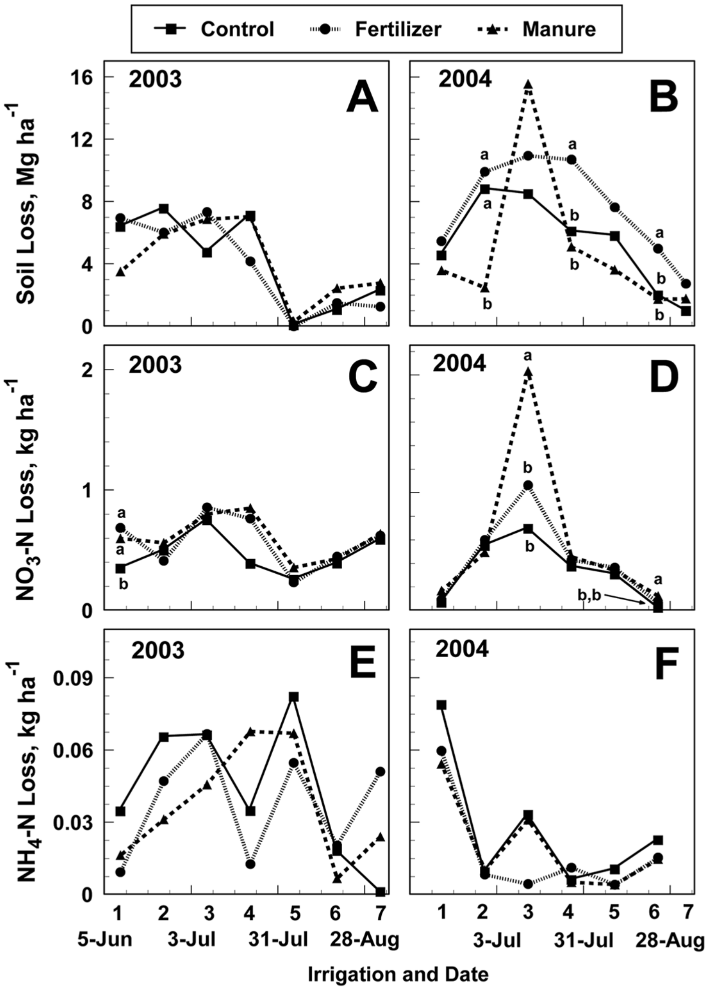

Fig. 4. Runoff mass losses of (A, B) sediment, (C, D) NO_3–N, and (E, F) NH_4–N as affected by year (2003 on left vs. 2004 on the right), irrigation number, and treatment. Values are derived from irrigation means. Treatment means for a given irrigation and year are significantly different at *P* < 0.05 if labeled with a different letter. Absence of letters indicates no significant difference among means.

averaged nearly 1.7°C. cooler than in 2003 (Fig. 2), which may have inhibited N and P mineralization or P desorption in 2004 soils (Yli-Halla and Hartikainen, 1996).

The effect of irrigation number on runoff component concentrations is illustrated in Fig. 3 and 5. Likewise, its effect on runoff component mass losses is evident in Fig. 4 and 6. In general, furrow runoff concentrations and mass losses for all components except NH_4–N were relatively small early in the irrigation season, peaked during midseason, and then declined to minimum or near minimum values during the last two or three irrigations. Brown et al. (1995) reported a similar pattern for runoff sediment in their furrow irrigation study, and Gilley et al. (2007) observed the same pattern in runoff from plots amended with incorporated cattle manure under simulated rainfall. These findings contrast with those of Austin et al. (1996), who found that 80% of the TP runoff losses from fertilized, flood-irrigated pastures occurred in the first irrigation. Fertilizers broadcast applied in the Austin et al. (1996) study were not incorporated into the soil with tillage.

Despite the similarities in the pattern among irrigations in this study, the significant irrigation × year interaction observed for all response variables (Table 2) indicates that the pattern of runoff component concentrations and mass losses among irrigations differed from one year to the next. See, for example, NO_3–N runoff concentrations (Fig. 3C, 3D), which exhibited a broad peak during midseason irrigations in 2003, yet a sharp midseason peak in 2004. It is not fully clear why these differences occurred between irrigations and years because there are a number of factors that may influence sediment and nutrient runoff losses (see later discussion).

Sediment and nutrient concentrations in individual inflow and runoff samples varied substantially during the study (Table 4). When values from control plots were averaged over each irrigation in 2003 and 2004, ranges for sediment concentrations (0.01–6 g L^{-1}), DRP (0.03–0.2 mg L^{-1}), and TP (1.5–8.5 mg L^{-1}) in furrow outflows (Fig. 3A, 3B, 5A–D) were comparable to those reported by Bjorneberg et al. (2006) for similar soils, but longer furrows. However, DRP concentrations in our recently manured plots averaged at least two times greater than those for Bjorneberg et al. (2006), which had no recent manure. By contrast, runoff from irrigation furrows established in a clay soil only produced TP concentrations ranging from 0.01 to 0.38 mg L^{-1} (McHugh et al., 2008), far lower than our TP values. Runoff DRP concentrations in this experiment were greater than those reported by Westermann et al. (2001) on similar, manured soils. Westermann et al.'s (2001) lower values resulted because (i) water flowing in the furrows was exposed to soil for a more limited time owing to their short furrows and relatively high flow rates, (ii) their irrigations were conducted either early in spring or late in fall when water temperatures were cooler than average (Yli-Halla and Hartikainen, 1996), and (iii) manure added to Westermann et al.'s (2001) plots was applied at least 4 yr previous to his study.

The Upper Snake River receives irrigation return flow originating from these and other irrigated fields. The mean sediment concentrations in furrow runoff exceeded the total maximum daily load (TMDL) target for the Snake River (0.052 g L^{-1}, Buhidar, 1997) for all but two irrigations of control furrows (Fig. 3A, 3B). Similarly, the mean TP concentrations in furrow runoff exceeded the TP TMDL for the Snake River (0.075 mg L^{-1}, Buhidar, 1997) for all irrigations (Fig. 5C, 5D).

In the present study, median furrow runoff concentrations for DRP ranged from 0.21 to 0.32 mg L^{-1} depending on treatment, and for NH_4–N from 0.07 to 0.08 mg L^{-1} (Table 4). These values were 1/2 to 1/100 of those found in runoff from a fertilized, flood-irrigated meadow, depending on how recently the meadow had been fertilized (White et al., 2003). The much greater DRP and NH_4–N values of the study of White et al. (2003) probably result from two factors. First, monoammonium phosphate fertilizer applied to the meadow was broadcast without incorporation. Second, irrigation waters came in direct contact with the entire soil surface area. Neither of these factors were characteristic of irrigations in this study. In one of the few surface irrigation investigations that included cow manure among their treatments, Monaghan et al. (2009) monitored runoff from border dike-irrigated dairy pastures littered with cow feces and fertilized

Table 3. Furrow runoff component concentrations and component mass losses in each irrigation (minus inflow contributions).

Year	Irrigation runoff component concentrations†						Cumulative mass losses in each irrigation†					
Treatment	Sed	K	NO_3–N	NH_4–N	DRP	TP	SED	K	NO_3–N	NH_4–N	DRP	TP
	g L^{-1}	mg L^{-1}					Mg ha^{-1}	kg ha^{-1}				
2003												
Control	2.3	0.60	0.31	0.02	0.11b ‡	2.4	3.4	0.87	0.45	0. 037	0.16b	3.5b
Fertilizer	2.1	0.67	0.36	0.02	0.12b	2.6	3.1	0.96	0.56	0.034	0.18b	3.8b
Manure	2.6	0.78	0.39	0.02	0.21a	3.6	3.7	1.11	0.60	0.033	0.31a	5.2a
Avg.	2.8	0.68B§	0.35A	0.02A	0.14A	2.8	3.4	0.98B	0.53A	0.035A	0.20A	4.1
2004												
Control	2.9	0.63b	0.14	0.01	0.07b	3.3	4.8ab	1.04b	0.22b	0.022	0.11b	5.4
Fertilizer	4.2	0.92b	0.18	0.01	0.07b	3.5	7.2a	1.59b	0.31ab	0.014	0.12b	5.9
Manure	2.3	1.55a	0.23	0.01	0.17a	2.5	4.1b	2.84a	0.42a	0.016	0.31a	4.4
Avg.	3.1	1.00A	0.18B	0.01B	0.09B	3.1	5.3	1.75A	0.31B	0.017B	0.16B	5.2
2-yr avg.												
Control	2.6	0.62b	0.21b	0.02	0.08b	2.8	4.1	0.95b	0.33b	0.029	0.13b	4.3
Fertilizer	3.1	0.79b	0.26a	0.01	0.09b	2.3	4.9	1.26b	0.42a	0.023	0.15b	4.8
Manure	2.4	1.13a	0.30a	0.01	0.19a	3.0	3.9	1.88a	0.50a	0.024	0.31a	4.8

† DRP = dissolved reactive phosphorus (filtered sample); Sed = sediment; TP = total phosphorus (unfiltered sample).

‡ Within a given variable and year, treatment means followed by the same lowercase letter are not significantly different ($P < 0.05$). Not displayed if effect was not significant in the ANOVA (Table 2).

§ Within a given component, yearly means followed by the same uppercase letter are not significantly different ($P < 0.05$). Not displayed if effect was not significant in the ANOVA (Table 2).

with superphosphate and urea (without incorporation). The dairy pasture's runoff typically contained 0.7 mg L^{-1} DRP and 0.35 mg L^{-1} NO_3–N + NH_4–N, which spiked to 4.6 and 3.4 mg L^{-1}, respectively, when irrigation occurred within 10 d of fertilizer application. These results were substantially greater than our values, verifying that incorporation of fertilizer or manure with tillage is an effective method for reducing nutrient runoff losses.

During a typical irrigation, sediment and nutrient concentrations in furrow runoff peaked shortly after initiation of runoff, declined to a moderate level for 4 to 5 h, then decreased still further where it often stabilized for the remainder of the irrigation. Data presented for each year in Fig. 7 demonstrate how runoff DRP concentrations varied during the initial irrigation, when DRP mass losses were small, and for the third or fourth irrigation, when DRP losses were greatest. The data show how strongly the DRP concentration in runoff from the manure treatment diverged from those in runoff from the control and fertilizer treatments (Fig. 7B, 7C, 7D), even though mean sediment concentrations did not differ (Fig. 3A, 3B). The furrow outflow rate data for the irrigations of Fig. 7 are presented in Fig. 8. Note that irrigation 3 in 2004 (Fig. 8D) was the only irrigation for which mean furrow outflow values of manure and fertilizer treatments were substantially greater than that of control plots (manure = 11.2 L min^{-1}, fertilizer = 10.4 L min^{-1} vs. control = 8.7 L min^{-1}). Thus the cumulative DRP losses in runoff from manured plots in this irrigation likely resulted from the combined effect of both increased concentration and increased outflow rate, relative to control plots.

Season-long Cumulative Mass Losses

When seasonal cumulative sediment and nutrient mass-loss values were analyzed (Table 5), the pattern of treatment effects was similar to that for individual irrigations (Table 2). In general, the influence of year was less pronounced when mass losses were summed over the irrigation season. Treatment effects on mean seasonal cumulative DRP mass losses were still significant overall, with losses from manure being 2.7 times greater than that for the control and 2.3 times greater than the fertilizer treatment (Table 6). However, while season-long NO_3–N mass-loss means for manure were nearly 1.5 times that of the control across both years (Table 6), the differences were no longer significant when analyzed for individual years, 2003 ($P = 0.33$) or 2004 ($P = 0.07$).

Previous research has shown that the shorter the period between fertilizer or manure application and the first irrigation, the greater the observed NO_3–N, NH_4–N, and DRP runoff losses (Monaghan et al., 2009; White et al., 2003; Smith et al., 2007). Accordingly, our 2003 crop season (fall manure application with immediate incorporation) should have resulted in smaller nutrient losses in runoff than the 2004 season (spring application with immediate incorporation). This is especially true since the 2004 spring manure application also supplied greater N and P than did the fall application (Table 1). However, our season-long mass-loss data appear to corroborate the above effect only for K and DRP, and not for NO_3–N, NH_4–N, or TP (Table 6). Even the case for DRP is somewhat questionable because the seasonal manure DRP losses for 2004 were only slightly greater than for 2003, and a majority of the difference was likely caused by the increased furrow outflow that occurred in 2004 relative to that in 2003. Season-long K mass losses for the 2004 manure treatment were more than 3.2 times those of the 2003 manure treatment. Thus, increasing the lag between manure application and first irrigation did decrease K runoff losses. During the period between the fall 2002 manure application and the first irrigation in 2003, the plots received 160 mm rainfall, while spring 2004 manure received only 43 mm between application and the first irrigation. Much of the K in 2003 manure plots apparently had been leached from the surface soil and thus did not interact with the furrow stream.

Because the quantity of NO_3–N and NH_4–N compared to organic N can be relatively low in manure (Gilley et al., 2007), the main source of inorganic N in runoff was likely mineralized from organic N. Hence, availability of inorganic N was governed by biocycling processes. The longer period from manure application to the first irrigation may have permitted greater mineralization to occur in 2003 than in 2004. We also observed more straw in the spring 2004 manure than was present in the previous manure application (although the two manure's C-to-N ratios were very similar; Table 1). The straw probably led to inorganic N being immobilized in microbial tissues in 2004, reducing the amount available for removal in runoff.

The 2004 fertilizer treatment also increased season-long cumulative K runoff losses compared with control plots (Table 6), even though the fertilizer we applied did not contain K. Sodium in the added fertilizer, however, may have increased the availability of soil K through mass action, with the increase in soil Na causing the release of K from the soil exchange complex.

In our study, 2.7% of the total inorganic N added as fertilizer in 2003 and 2004 and 1.5% of the total N added as manure were lost in runoff (Tables 1, 6). These results are similar to that reported for flood-irrigated cropland in southern Saskatchewan, Canada, where 1.9% of applied N was lost in surface runoff (Cessna et al., 2001). The manure treatment added a total of 231 kg P ha^{-1} during the 2 yr, while 33.4% of the added P was lost in furrow runoff. We calculated the value of the lost nutrients from our furrow irrigated plots based on fertilizer replacement costs for urea (US\$1.32 kg^{-1} N^{-1}), super phosphate (\$1.96 kg^{-1} P^{-1}), and KCl (\$1.03 kg^{-1} K^{-1}). Mean dissolved nutrient losses amounted to \$8.36 ha^{-1} yr^{-1} for control, \$11.24 ha^{-1} yr^{-1} for fertilizer, and \$17.63 ha^{-1} yr^{-1} for manure treatments. Total seasonal losses of TP, which include particulate P losses, amounted to \$88.11 ha^{-1} yr^{-1} for control, \$101.00 ha^{-1} yr^{-1} for fertilizer, and \$103.83 ha^{-1} yr^{-1} for manure treatments. If the nutrients lost in runoff were replaced with nutrients derived from manure, the costs would likely be lower.

Factors Affecting Runoff Nutrient Concentrations

Results from the stepwise regression analyses presented in Table 7 show that chosen independent variables explained a majority of the variability associated with mean DRP, TP, and K runoff concentrations per irrigation (Model R^2 = 0.52, 0.73, and 0.52), but only a small portion of that for NO_3–N (Model R^2 = 0.16). The pool of independent variables likely did not include good measures of NO_3–N availability or quantity in furrow soils, or the factors that influence the biologic cycling of N. Of the factors considered in component regressions, sediment concentration and treatment typically explained a major portion of the variance associated with the dependent variable and were positively related to DRP, TP, NO_3–N, and K runoff concentrations. In addition, DRP, TP, and K concentrations decreased as the season progressed (increasing day of year). Sediment concentration more strongly influenced TP loads in furrow runoff than DRP because unfiltered runoff samples included substantial quantities of P-bearing minerals and organic solids (Berg and Carter, 1980; Bjorneberg et al., 2006). The TP, NO_3–N, and K models included infiltration fraction as a predictor, whereas the DRP incorporated the inversely correlated mean outflow rate factor. This suggests subtle differences in the processes responsible for introducing these two component groups into the furrow stream.

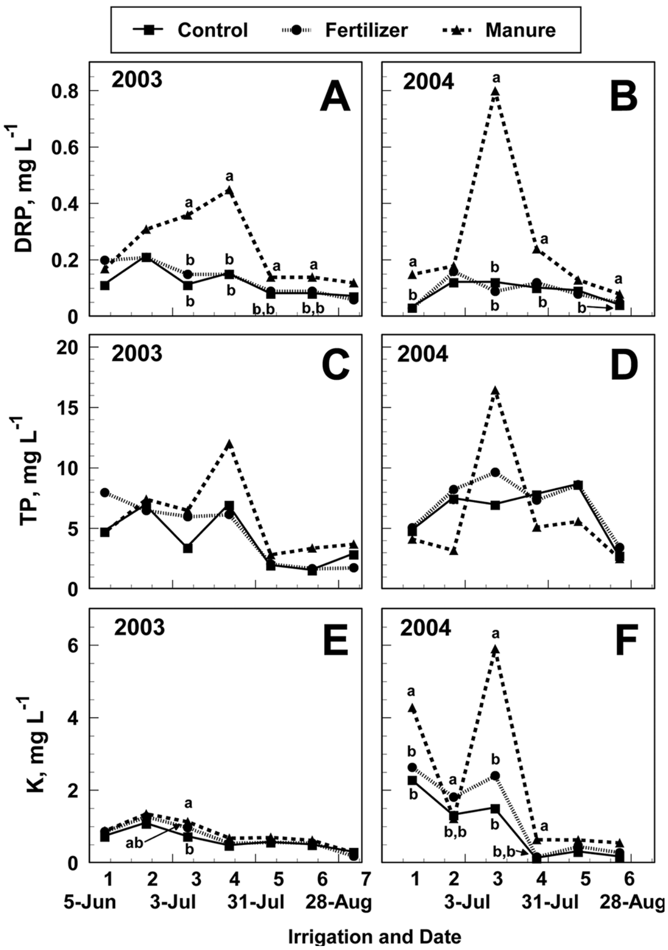

Fig. 5. Runoff (A, B) dissolved reactive phosphorus (DRP), (C, D) total phosphorus (TP), and (E, F) dissolved K as affected by year (2003 on left vs. 2004 on the right), irrigation number, and treatment. Values are derived from irrigation means. Treatment means for a given irrigation and year are significantly different at *P* < 0.05 if labeled with a different letter. Absence of letters indicates no significant difference among means.

Table 4. Median, mode, and range of component concentrations as measured in all inflow and runoff water samples (runoff n = 160; inflow n = 32). Note: median and mode values were included instead of means because nontransformed component concentration values were not normally distributed.

Runoff component / Treatment	Median	Mode	Range
	g L^{-1}		
Sediment			
Control	4.82	9.9	0.01– 29.9
Fertilizer	4.99	5.1	0.01– 27.1
Manure	3.95	3.7	0.01– 50.0
Inflow	negligible	negligible	negligible
	mg L^{-1}		
Potassium			
Control	5.33	4.05	3.75–14.7
Fertilizer	5.56	6.00	3.84–16.8
Manure	5.98	10.3	3.80–46.4
Inflow	4.0	3.6	3.6–5.0
NO_3–N			
Control	0.44	0.0	0.0–4.07
Fertilizer	0.51	0.0	0.0–2.85
Manure	0.52	0.0	0.0–3.11
Inflow	0.04	0.01	0.0–0.15
NH_4–N			
Control	0.08	0.02	0.003–2.25
Fertilizer	0.07	0.03	0.0–0.37
Manure	0.07	0.02	0.0–2.28
Inflow	0.06	0.05	0.03–0.09
Dissolved reactive P			
Control	0.21	0.16	0.02–0.71
Fertilizer	0.23	0.02	0.02–2.90
Manure	0.32	0.15	0.03–14.3
Inflow	0.03	0.03	0.02–0.86
Total P			
Control	5.39	5.5	0.33–20.7
Fertilizer	5.56	11.5	0.32–24.4
Manure	4.85	11.3	0.11–41.5
Inflow	0.077	0.08	0.03–0.14

Runoff sediment concentration represented a measure of the mass and, by extension, surface area of soil particles present in the water stream. Increasing this mass and surface area enhanced diffusion, dissolution, desorption processes, and hence the transfer of soluble nutrients from soil to water (Yli-Halla and Hartikainen, 1996; Muukkonen et al., 2009). As an independent variable in the regression, treatment was coded as control = 0, fertilizer = 1, and manure = 2. Presumably this variable represented a crude measure of the quantity of nutrients in the soil, increasing from control through manure (Robbins et al., 2000). Day of year also represented a measure of nutrient quantity in the following way. As the number of applied irrigations increased through the irrigation season, the amount of easily erodible soil in furrows declined, as did soluble and desorbable nutrients associated with the soil lining the furrow wetted perimeter (Oloya and Logan, 1980).

The infiltration fraction and mean outflow rate are strongly correlated, yet they were selected separately for individual regression models. These two factors explained a relatively large amount of variation in DRP, NO_3–N, and K concentrations but were less influential for TP (Table 7). Outflow rate in absolute terms provided a measure of "mixing opportunity time," where a decrease in outflow rate increased the residence time of water in the furrow, and extended interaction opportunities between the water and furrow soils (Bjorneberg et al., 2006). Infiltration fraction is a relative value that may represent a measure of the interaction potential between surface flow and subsurface soils. Increasing infiltration fraction increased subsurface water contents and the potential for soil to become saturated above near-surface restrictive layers such as plow pans or cemented horizons. This could lead to local occurrences of increased hyporheic flow, greater water exchanges between subsurface zones and surface water, and extended water-sediment contact times (Harvey and Bencala, 1993; Marion et al., 2008). Whereas NO_3–N concentration in furrow streams increased with enhanced hyporheic water exchange (as indicated by the positive correlation of infiltration fraction with NO_3–N, Table 7), K concentrations decreased with enhanced hyporheic water exchange (as indicated by the negative correlation of infiltration fraction with K, Table 7). Unlike NO_3–N concentrations, which were more dependent on biocycling processes active in soil sublayers, stream K concentrations may have resulted, for the most part, from the simple dissolution of salts present in surface soils. Increasing infiltration fraction decreased stream flow and reduced the soil surface area exposed to flows (i.e., the furrow's wetted perimeter) and also leached a greater proportion of K cations into soils where they were less available for surface transport. Both effects would tend to decrease stream K concentrations.

Apparently, the processes responsible for increasing NO_3–N and K concentrations in the furrow stream were not as dependent on sediment interactions as those for DRP and TP. Moreover, in-stream mixing opportunity time (as indicated by mean outflow rate) appeared to affect DRP more than other components. This suggests that DRP concentrations depended more on desorption and diffusion processes than other nutrients.

A substantial portion of variability was unexplained in all of the predictive models. Clearly, our pool of independent variables did not encompass all factors affecting component concentrations. For example, water ionic strength and temperature are known to influence DRP desorption from soils (Yli-Halla and Hartikainen, 1996). Potentially, the crops themselves may influence runoff water quality in response to variations in biomass production and litterfall. Nutrients in senescent leaves that collect in furrows late in the growing season can be solubilized during an irrigation event. In this study, corn yields did not differ among treatments (Table 5), suggesting that differences in litterfall were not important. The lack of treatment

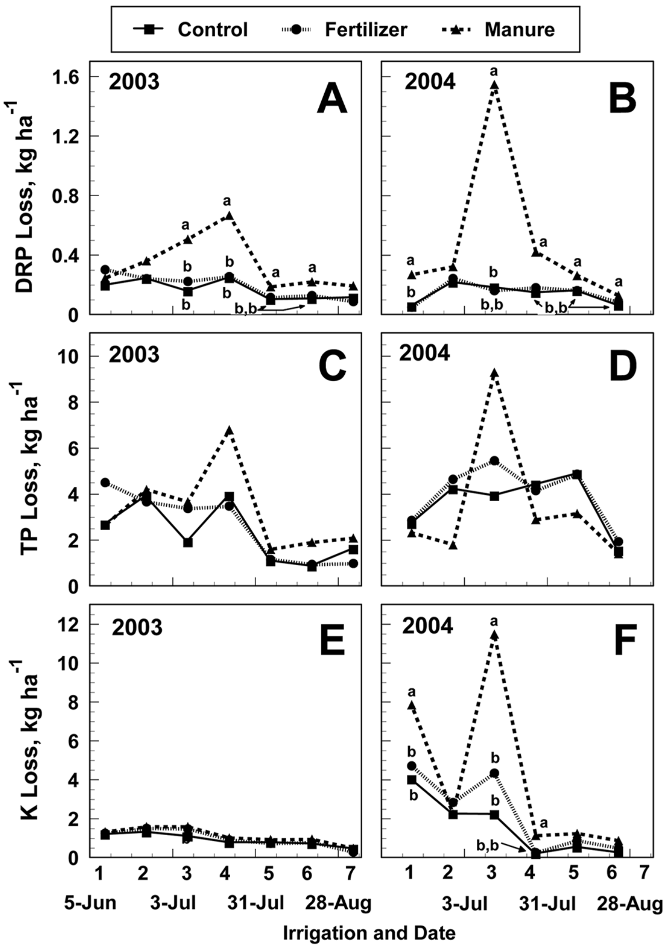

Fig. 6. Runoff mass losses of (A, B) dissolved reactive phosphorus (DRP), (C, D) total phosphorus (TP), and (E, F) dissolved K as affected by year (2003 on left vs. 2004 on the right), irrigation number, and treatment. Values are derived from irrigation means. Treatment means for a given irrigation and year are significantly different at *P* < 0.05 if labeled with a different letter. Absence of letters indicates no significant difference among means.

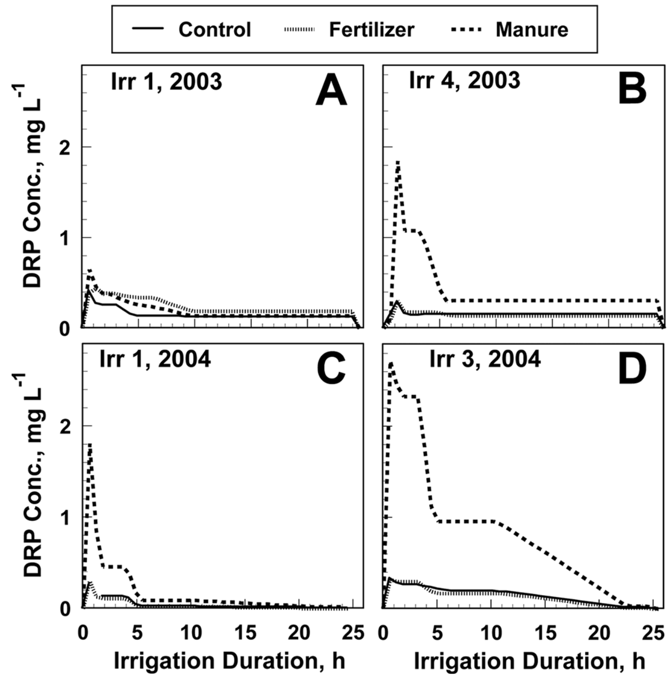

Fig. 7. Treatment dissolved reactive phosphorus (DRP) concentrations in furrow runoff for 2003 during (A) irrigation 1 and (B) irrigation 4 and for 2004 during (C) irrigation 1 and (D) irrigation 3. Values are means of three replicates.

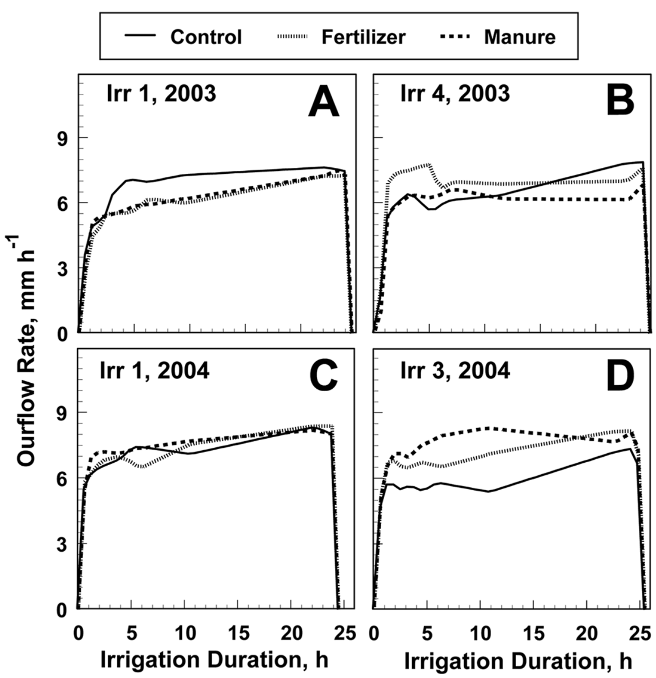

Fig. 8. Treatment furrow outflow rates during (A) irrigation 1 and (B) irrigation 4 in 2003 and during (C) irrigation 1 and (D) irrigation 3 in 2004. Values are means of three replicates.

effect on crop yield also suggests that the uptake of water and soil nutrients by crops was similar among treatments.

Synopsis of Findings

1. Inorganic fertilizer and manure treatments did not influence water infiltration rates into irrigation furrows, and except for a few irrigations, sediment concentrations and mass losses in irrigation furrows were similar for all treatments (Table 2).
2. The addition of N to soils, whether from fertilizer or manure, increased mean NO_3–N losses per irrigation (2-yr average) from furrows but had no effect on NH_4–N (Table 3).
3. Relative to control plots, manure additions increased runoff concentration and mass losses of NO_3–N, DRP, and K in each irrigation (2-yr average), but not TP (Table 3).
4. The magnitude of DRP and inorganic-N losses in runoff varied from year to year for all treatments and appeared to be influenced more by the timing of the amendment application and environmental conditions than by the quantity of nutrients applied.
5. Manure added large amounts of K to the soil, which was susceptible to leaching and available for transport and export in runoff waters. Sizable leaching events that occurred after manure application and before the first irrigation probably reduced K runoff losses appreciably during the subsequent irrigation season.

Table 5. The influence of treatment and year on dry corn silage yield, cumulative seasonal furrow flows, and cumulative season-long runoff component losses (minus inflow contribution). Table gives *P* values for main effect and interaction terms derived from an analysis of variance.

Source of variation	*P* values									
	Corn yield	Cumulative seasonal flows			Cumulative seasonal mass losses†					
		Inflow	Outflow	Infiltration	Sed.	K	NO_3–N	NH_4–N	DRP	TP
Treatment (TRT)	0.9	0.5	0.6	0.5	0.5	**	*	0.7	***	0.8
Year (Y)	0.2	*	*	0.2	0.1	**	0.06	**	0.9	0.3
TRT × Y	0.3	*	**	*	0.3	**	0.4	0.9	**	0.4

* $P < 0.05$.

** $P < 0.01$.

*** $P < 0.001$.

† DRP = dissolved reactive phosphorus; Sed., sediment; TP = total phosphorus.

Table 6. Crop yields and cumulative season-long values for furrow inflow, outflow, infiltration, and component mass losses (minus inflow contributions).

Year Treatment	Corn yield dry wt.	Seasonal total flows			Cumulative seasonal mass losses†					
		Inflow	Outflow	Infiltration	Sed.	K	NO_3–N	NH_4–N	DRP	TP
	Mg ha^{-1}	mm			Mg ha^{-1}	kg ha^{-1}				
2003										
Control	20.4	1526b‡	1032b	494ab	34.0	6.41d	3.44	0.36	1.21cd	33.2
Fertilizer	18.8	1503b	1037b	467a	29.2	7.44cd	4.18	0.30	1.47c	35.9
Manure	21.2	1531b	1028b	504ab	29.4	8.08cd	4.30	0.29	2.49b	41.8
2004										
Control	16.8	1623a	1153ab	470ab	38.0	9.8c	2.11	0.24	0.89d	44.0
Fertilizer	20.1	1624a	1211a	413b	54.6	14.4b	2.75	0.16	1.06cd	50.7
Manure	17.4	1617a	1252a	365ab	34.2	26.1a	3.81	0.15	3.20a	40.4
2-yr avg.										
Control	18.6	1575	1093	483	36.0	8.11B§	2.78B	0.31	1.05B	38.6
Fertilizer	19.5	1563	1124	440	41.9	10.9 B	3.47AB	0.23	1.26B	43.3
Manure	19.3	1574	1140	434	31.8	17.1A	4.05A	0.22	2.84A	41.1

† DRP = dissolved reactive phosphorus (filtered sample); Sed., sediment; TP = total phosphorus (unfiltered sample).

‡ For a given component, individual treatment-by-year means followed by the same lowercase letter are not significantly different. ($P < 0.05$). Not displayed if effect was not significant in the ANOVA (Table 5).

§ For a given component, 2-y treatment averages followed by the same uppercase letter are not significantly different ($P < 0.05$). Not displayed if effect was not significant in the ANOVA (Table 5).

Table 7. Models derived from stepwise regressions fitting average irrigation runoff concentrations (adjusted for inflow contributions), Log_{10} [dissolved reactive P mg L^{-1}], Log_{10} [unfilt. total P mg L^{-1}], [NO_3–N mg L^{-1}]$^{1/2}$, [potassium mg L^{-1}]$^{1/2}$, to predictor variables including treatment, day of year, and irrigation averages for infiltration fraction, runoff sediment concentration (mg L^{-1}) and furrow outflow rate (L min^{-1}). (n = 226)

Independent variable†	Dependent variable‡											
	Log_{10} [DRP (mg L^{-1})]			Log_{10} [TP (mg L^{-1})]			[NO_3–N (mg L^{-1})]$^{1/2}$			[Potassium (mg L^{-1})]$^{1/2}$		
	Par. coef.§	Model¶		Par. coef.	Model		Par. coef.	Model		Par. coef.	Model	
		R^2	Pr > F		R^2	Pr > F		R^2	Pr > F		R^2	Pr > F
Intercept	–0.5397	–	0.03	0.0292	–	0.04	0.2444	–	<0.001	2.7593	–	<0.0001
Sediment	0.0555	0.24	<0.0001	0.0861	0.70	0.05	0.0248	0.09	<0.001	0.0303	0.02	0.011
Treatment	0.1991	0.47	<0.0001	0.0385	0.71	0.03	0.0559	0.13	0.01	0.1318	0.07	0.0003
Day of year	–0.0018	0.49	0.04	–0.0011	0.72	0.05				–0.0085	0.44	<0.0001
Infiltration fraction#				0.3035	0.73	0.05	0.4403	0.16	0.03	–1.1758	0.52	0.0003
Outflow rate	–0.0445	0.52	0.005									

† The order of independent variable shown here is not necessarily the order they were select in the stepwise regression analysis.

‡ DRP = dissolved reactive phosphorus; Par., coef. = parameter coefficients. TP = total phosphorus.

§ These parameter coefficients were those of the final statistical model that included the independent variables listed below for which a coefficient is shown.

¶ The model's statistics shown are those that resulted after the corresponding independent variable was added to the statistical model.

Computed as the ratio of net infiltration divided by total inflow.

6. Mean irrigation values for runoff component concentrations and mass losses were comparatively small early in the season, peaked in midseason, then declined to minimal values in the last two or three irrigations, a pattern unlike that for TP runoff losses from fertilized, flood-irrigated pastures (Fig. 3, 4, 5, and 6).
7. The DRP and inorganic-N concentrations in runoff from a these furrow-irrigated fields were substantially smaller than published values from fertilized flood-irrigated meadows (White et al., 2003) or border-dyke-irrigated dairy pastures (Monaghan et al., 2009), suggesting that incorporation of fertilizer or manure with tillage is an effective method for reducing nutrient losses in irrigation furrow runoff.
8. The cost to replace NO_3–N, NH_4–N, K, and TP lost in runoff from even nonfertilized plot soils was high, averaging \$88 ha^{-1} yr^{-1} (assuming replacement using inorganic fertilizers). After fertilization with either inorganic fertilizer or manure, the replacement costs were 15% greater, being more than \$101 ha^{-1} yr^{-1}.

Conclusions

Before this work, little documentation was available describing the effect of fertilizer or manure application on season-long nutrient losses in runoff from furrow-irrigated fields. Our study conducted in the semiarid Intermountain West indicates that the addition of inorganic fertilizer or manure to soils can double or triple nutrient losses from furrow-irrigated fields. This increased nutrient loss from these cropped fields is a substantial and direct financial cost to the farmer and increases the potential for offsite ecological damage. This research points toward a need to develop management practices that can reduce nutrient losses from amended, furrow-irrigated soils.

Year-to-year variation in nutrient runoff losses from control and treated soil suggests that the magnitude of annual nutrient losses in irrigation runoff depends not only on the type and amount of nutrient added to the soil but also on the timing of the application and attendant environmental factors. Results suggest that when amendments were incorporated into soil, runoff losses of soil DRP and inorganic N were substantially influenced by biocycling processes. These processes determine the amounts and forms of nutrients that can be transported in furrow streams at any given time. Further study over more extended periods may be needed to better understand environmental effects on nutrient runoff potential.

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