Efficacy of Natural Wetlands to Retain Nutrient, Sediment and Microbial Pollutants

A. K. Knox,* R. A. Dahlgren, K. W. Tate, and E. R. Atwill University of California

Wetlands can improve water quality through natural processes including sedimentation, nutrient transformations, and microbial and plant uptake. Tailwater from irrigated pastures may contribute to nonpoint source water pollution in the form of sediments, nutrients, and pathogens that degrade downstream water quality. We examined benefits to water quality provided by a natural, flow-through wetland and a degraded, channelized wetland situated within the flood-irrigation agricultural landscape of the Sierra Nevada foothills of Northern California. The non-degraded, reference wetland significantly improved water quality by reducing loads of total suspended sediments, nitrate, and *Escherichia coli* on average by 77, 60, and 68%, respectively. Retention of total N, total P, and soluble reactive P (SRP) was between 35 and 42% of loads entering the reference wetland. Retention of pollutant loads by the channelized wetland was significantly lower than by the reference wetland for all pollutants except SRP. A net export of sediment and nitrate was observed from the channelized wetland. Decreased irrigation inflow rates significantly improved retention efficiencies for nitrate, *E. coli*, and sediments in the reference wetland. We suggest that maintenance of these natural wetlands and regulation of inflow rates can be important aspects of a best management plan to improve water quality as water runs off of irrigated pastures.

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*Corresponding author (akknox132@yahoo.com).
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677 S. Segoe Rd., Madison, WI 53711 USA

Runoff from grazed, irrigated pastures can contain high concentrations of nutrients, sediments and microbial pollutants, which contribute to the degradation of downstream water quality. In California, irrigated pastures maintain more than 445,000 ha of green forage throughout dry summer months. Flood irrigation is a common and inexpensive method of supplying irrigation water, but a substantial portion of the applied water can become runoff at the base of the pasture (Tate et al., 2000). Irrigation runoff enlarges natural wetlands during the dry summers, and therefore wetlands are commonly found downslope from irrigated pastures. As these naturally occurring wetlands are in various stages of hydrologic and vegetative degradation, there are opportunities within the range landscape to utilize, manage, and restore the wetlands to improve water quality.

Wetlands can provide important benefits to water quality by retaining or transforming pollutants such as nutrients, sediments, pathogens, pesticides, and trace metals (Blahnik and Day, 2000; Fisher and Acreman, 2004; Jordan et al., 2003; Mitsch and Gosselink, 1993; Woltemade, 2000). Riparian buffer zones and vegetated filter strips have been accepted as important management practices to improve the quality of rangeland runoff before it enters streams or rivers (Dillaha et al., 1989; Haycock et al., 1996; Atwill et al., 2006). Wetlands may improve on the benefits provided by vegetated buffer zones through additional processes that occur under highly productive, flooded, and anaerobic conditions. Wetlands can also provide critical aquatic and terrestrial wildlife habitat.

Factors controlling the efficiency of a wetland to retain contaminants include the contaminant loading rate, the hydraulic residence time of water within the wetland, concentrations of microbially-labile organic matter, and the available surface area of plants and other substrates for the growth of microbes (Phipps and Crumpton, 1994; Woltemade, 2000; Fisher and Acreman, 2004). Excessive pollutant loading to wetlands and alteration of either inflow sources or inflow rates to previously unimpacted natural wetlands can degrade natural wetlands and reduce biodiversity (Cooke et al., 1990; van der Valk and Jolly, 1992; Gopal, 1999). However, by applying appropriate management practices to existing wetlands, and by restoring or enhancing wetlands, important benefits to water quality can be realized while also increasing habitat diversity and maintaining ecological sustainability (Zedler, 2003).

A summary of the literature by Kadlec and Knight (1996) showed mean retention percentages for contaminants by constructed wetlands receiving municipal wastewaters as follows: total P (34%), ortho-PO_4 (41%), total N (55%), NO_3 (51%), and total

A.K. Knox and R.A. Dahlgren, Dep. of Land Air and Water Resources, Univ. of California, Davis, CA. K.W. Tate, Dep. of Plant Sciences, Univ. of California, Davis, CA. E.R. Atwill, School of Veterinary Medicine, Univ. of California, Davis, CA.

suspended solids (TSS) (68%). In a natural wetland, Blahnik and Day (2000) observed similar, although highly variable, retention of phosphate (3–50%), NO_3 (32–95%), and TSS (48–91%). Retentions of 80 to 99% have been seen for *Escherichia coli* (*E. coli*) and fecal coliforms in constructed surface flow wetlands treating municipal and livestock wastewater (Gerba et al., 1999; Quinonez-Diaz et al., 2001; Hill, 2003). Two constructed wetlands in an agricultural (dairy) watershed provided reductions in median NO_3–N concentrations of 34 and 94% and median TN concentration reductions of 56 and 33% (Tanner et al., 2003). However, it is not certain whether similar levels of retention should be expected in natural wetlands.

Extensive research has been conducted to ascertain water quality benefits of constructed wetlands and to identify principles for their optimum design (Kadlec and Knight, 1996). However, knowledge gained from these constructed wetland systems is not readily transferable to naturally occurring wetlands receiving agricultural runoff. In natural wetlands, flow paths often become channelized through alterations to microtopography. Since channelization generally decreases the hydraulic residence time and reduces the connectivity of the water with the wetland, the pollutant removal efficiency of the wetland is often reduced in channelized wetlands (Hammer and Kadlec, 1986; Wetzel, 2001; Jenkins and Greenway, 2005). In addition, agricultural runoff tends to have more pulsed flows, less organic matter, and more sediment than does wastewater from urban sources, which is the typical source for retention studies in constructed wetlands (Kadlec and Knight, 1996; Woltemade, 2000; Jordan et al., 2003).

In this study, we examined changes in water quality through an intact, flow-through wetland and a degraded, channelized, flow-through wetland situated within a flood-irrigated, agricultural landscape in the Sierra Nevada foothills. The range landscape is a patchwork of irrigated, improved grass hillslopes interspersed within a matrix of annual grassland and oak woodland pastures. Due to the topography of this region, natural wetlands are interspersed throughout the landscape as part of the natural drainage system. We quantified and compared the efficacy of two wetlands within the flood irrigation landscape to enhance water quality of tailwaters by removing nutrients (nitrate, total N, soluble reactive PO_4, total P), sediments (TSS, volatile and nonvolatile SS), and *E. coli*. Irrigation inflow rates were varied to determine the effects of hydraulic loading rate on pollutant removal efficiencies. We suggest that maintenance of these natural wetlands can be an important aspect of a best management plan to improve water quality of discharge from irrigated pastures.

Methods

Study Site

Research was conducted on two natural, flow-through wetlands within the University of California's Sierra Foothill Research and Extension Center (SFREC) located on the western slope of the Sierra Nevada foothills (elevation ∼350 m) in Yuba County, CA. The climate is Mediterranean with cool, wet winters and hot, dry summers. Average annual precipitation is 65 cm, with approximately 90% falling between October and March (Lewis et al., 2000). Mean maximum daily summer temperature (June–August) is 31°C. Flood or sprinkler irrigation is required to maintain summer forage for grazing between April and October.

Water quality was monitored from two wetlands which receive irrigation tailwaters. The "reference wetland" was composed of a broad basin that had no distinct downcut, or entrenched, region for the length of the wetland. In the second wetland, the "channelized wetland," a primary channel (one to three meters wide and 30–70 cm lower than the surrounding topography) transported most of the irrigation runoff water. This incised channel was sparsely covered by vegetation relative to the dense wetland vegetation in the surrounding area of this wetland and throughout the reference wetland. Each of the two wetlands was situated at the base of a small drainage basin that collected runoff from 6.1 (reference) and 9.3 ha (channelized) of flood irrigated pastures (Fig. 1). Each wetland had a surface area of about 0.2 ha, with minimum longitudinal flowpaths of 135 m (reference) and 115 m (channelized). The longitudinal flowpaths were 1.5 and 3 times greater than the wetland widths. Typical water depths within the wetlands ranged from 0 to about 60 cm.

Dominant vegetation within the wetlands was composed of *Typha* spp., *Polygonum punctatum* Ell., *Veronica catenata* Pennell, and *Leersia oryzoides* (L.) Sw. Dominant soils in the pastures were classified as fine-loamy, mixed, thermic, Mollic Haploxeralfs (Herbert and Begg, 1969). The irrigated pastures upslope from the wetlands were rotationally grazed by beef cattle about one time per month throughout the irrigation season with grazing events ranging from 8 to 16 d. Livestock numbers were between 56 and 102 head resulting in mean stocking rates per grazing event of 3.1 to 4.5 animal unit months (AUM) per ha. Grazing was permanently excluded from both wetlands beginning more than 1 mo before the first irrigation trial in May 2004.

Irrigation Trials and Sample Collection

The pasture upstream of each wetland consisted of five irrigated sections each covering 1 to 2 ha (Fig. 1). Sections were irrigated sequentially over several consecutive days as water application on each section lasted from 3.5 to 25 h depending on application rate. To maintain green forage throughout the summer, re-irrigation of each section occurred within 9 to 14 d. For all trials in this study, runoff was collected from only one irrigated section (i.e., Section 4 shown in Fig. 1) from each pasture. Irrigation of this section was randomly assigned within each rotation so that some collection events were at the beginning of the irrigation sequence (directly following a 9 to14 d dry down period) while other collections directly followed one or more flow events through the wetland.

Eleven irrigation events were sampled for each wetland over two summer irrigation seasons in 2004 (n = 7) and 2005 (n = 4). Streamflow samples (400 mL) were collected at 15- to 30-min intervals upstream and downstream from the wetlands using ISCO 6700 autosamplers (ISCO, Lincoln, NE) for the duration of each event. Samples were collected during both the rising and falling limb of the runoff hydrograph for each irrigation event. Inlet and outlet flows were continuously recorded at 0.25-h intervals within a channelized reach immediately upstream and downstream from each wetland via a 1 ft, 90°

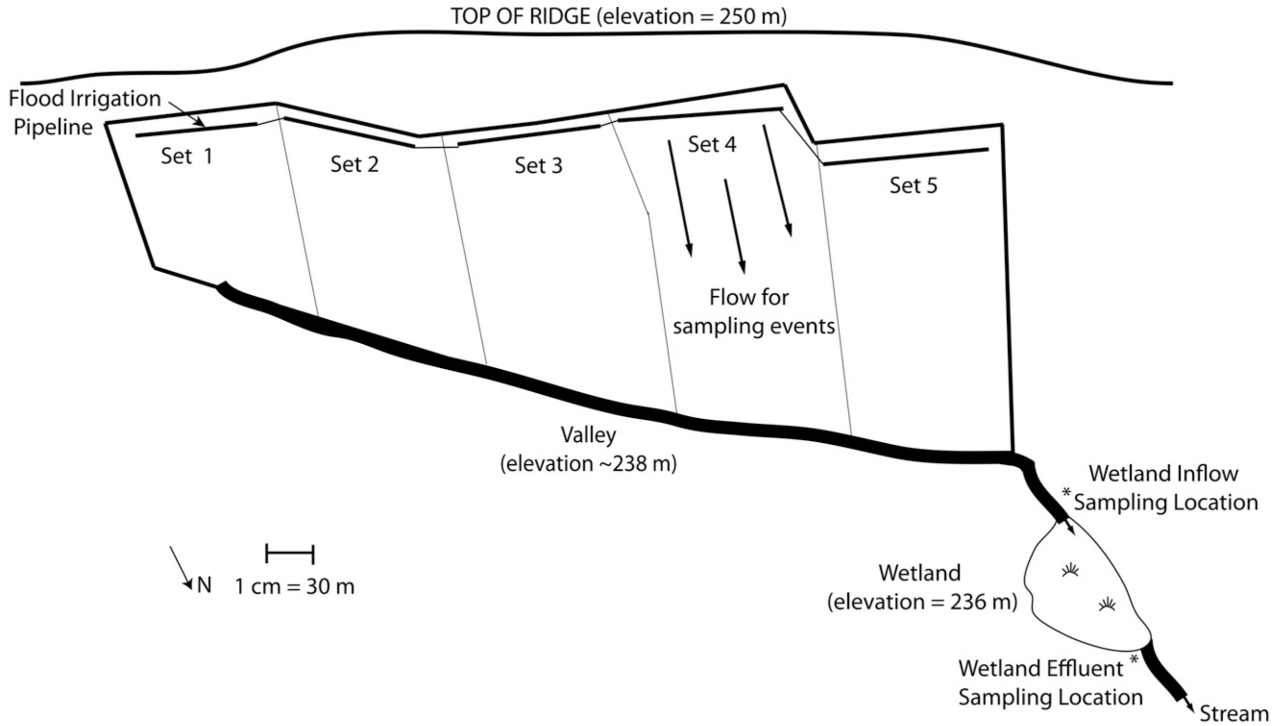

Fig. 1. Site layout showing the hillslope pasture, five sections in the irrigated pasture, and the wetland, including the inflow and effluent sampling locations.

v-notch weir with an automatic depth recorder (Metritape Type AGS, Metritape, Littleton, MA). To ascertain relationships between hydraulic loading rate and wetland efficiency for nutrient removal, flood irrigation application rates were varied to attain a range of maximum inflow rates from 0.006 to 0.06 $m^3 s^{-1}$.

Water samples were refrigerated (3°C) on collection and through completion of sample analyses. Water pH was determined using Acumet AB15 pH meter (Fischer Scientific). A 70-mL subsample was immediately filtered through a 0.2 μm polycarbonate membrane (Nuclepore) for analysis of dissolved components: nitrate N (NO_3–N) and soluble reactive phosphorus (PO_4–P or SRP). Total nitrogen (TN) and total phosphorus (TP) were determined on non-filtered samples following a 30 min digestion with 13.3% persulfate (Yu et al., 1994). The Carlson conductance method (Carlson, 1978) was used to quantify nitrate, TN, and ammonium concentrations for samples in 2004. Nitrate and TN were quantified using the vanadium chloride reduction method (Miranda et al., 2001; Doane and Horwath, 2003) in 2005. For both procedures, the minimum detection limit (MDL) was 10 μg L^{-1}. In 2005, ammonium was quantified colorimetrically using the phenate method (MDL = 20 μg L^{-1}) (Forster, 1995). Total phosphorus and SRP were measured with the ammonium molybdate method (Clesceri et al., 1998) using a PerkinElmer Lambda 3 spectrophotometer (PerkinElmer, Norwalk, CT) (MDL = 5 μg L^{-1}). Quantification of *E. coli* was conducted within 24 h after sample collection. *E. coli* concentrations (cfu 100 mL^{-1}) were determined by direct membrane filtration and culturing the membrane onto CHROMagar EC (Chromagar Microbiology, Paris, France) at 44.5°C for 24 h (Clesceri et al., 1998).

Total suspended solids (TSS) were measured from a 150- to 650-mL subsample. The subsample was filtered through a preweighed glass fiber filter (0.45 μm Millipore); the filter was dried at 60°C for 24 h and weighed again, the difference being the mass of sediment in the water sample. Nonvolatile suspended solids (NVSS) were determined by reweighing the same sample after ashing at 550°C for 3 h. Volatile suspended solids (VSS) were calculated as the difference between TSS and NVSS. The MDL for TSS and VSS was ~0.5 mg L^{-1}; values that fell below the MDL were set to 0.1 mg L^{-1} for flux calculations.

Bromide-Nitrate Injections

Continuous bromide (a conservative tracer) and nitrate co-injections were conducted at peak, steady-state flow to quantify hydraulic residence times (HRT) and to confirm nitrate removal within the wetland. A solution of known Br (NaBr) and nitrate ($NaNO_3$) concentrations was premixed in a 20 L carboy and injected at a known rate (20–25 mL min^{-1}) into the channel upstream from the wetland using a fluid metering pump. During injections, water samples were collected at short time intervals (3 to 20 minutes) to collect rising limb, peak, and falling limb concentrations both upstream and downstream from the wetland. Bromide and nitrate concentrations were quantified using ion chromatography (Dionex 500x; AS4A columns, Sunnyvale, CA).

Bromide-nitrate co-injections were conducted at a range of inflow rates in each wetland. Direct flow measurements using the V-notch weir and stage recorder were compared with calculated discharge rates (bromide injectate concentration multiplied by the injection rate divided by the plateau concentration) (Webster and Ehrman, 1996). In the reference wetland, injec-

Table 1. Minimum, maximum and median pollutant concentrations in runoff from irrigated pastures as it entered the reference (n = 195) and channelized (n = 320) wetlands.

	Reference wetland			Channelized wetland		
	Min.	Median	Max.	Min.	Median	Max.
E. coli (cfu/100mL)	420	5400	157,800	450	10,000	121,555
NO_3–N (mg L^{-1})	0.04	0.20	0.70	0.02	0.31	2.44
TN (mg L^{-1})	0.39	0.90	8.32	0.42	1.74	10.27
NVSS (mg L^{-1})	<0.5	10	405	<0.5	7.1	274
VSS (mg L^{-1})	1	28	251	0.9	4.7	115
TSS (mg L^{-1})	1	41	501	1	11	389
SRP (mg L^{-1})	0.01	0.05	0.28	0.01	0.07	0.24
TP (mg L^{-1})	0.02	0.11	0.40	0.04	0.15	0.57

tions were completed at flows of 13, 31, and 43 L sec^{-1}. Three co-injections were completed on the channelized wetland at a range of inflow rates (8, 28, and 34 L sec^{-1}). The target concentration for bromide was 1 mg L^{-1} in all events. Target nitrate concentrations were 0.5 to 1 mg L^{-1} above background levels.

The HRT was calculated for each wetland at a variety of inflow rates using the time it took for half of the Br mass introduced at the upstream site to pass the downstream site (Webster and Ehrman, 1996). By comparing the Br/NO_3 ratio (conservative tracer/biologically reactive nitrate), it was possible to distinguish nitrate retention (uptake or denitrification) from hydrologic variance (Triska et al., 1989). In addition, the difference between NO_3–N input and output during a co-injection experiment can be used to estimate nitrogen removal (Rutherford and Nguyen, 2004).

Data Analysis

Instantaneous pollutant load curves for each sample location (above, below) for both wetlands (reference, channelized) were calculated as the product of concentration and instantaneous flow for each sample collection throughout each irrigation event. Total irrigation event loads (g or kg) entering and leaving each wetland were then determined by summing instantaneous loads across all time intervals for each event (i.e., integration under the instantaneous flux curve). This resulted in a total of 44 observations of total event load for analysis (11 irrigation events at two wetlands with two sample locations per wetland).

General linear analysis was used to test if (i) the efficacy of the channelized wetland to reduce pollutant load during individual irrigation events was different than the reference wetland, and (ii) water inflow rate (L sec^{-1}) and cattle grazing on the pasture during the irrigation event (present v. absent) impacted the overall magnitude of pollutant load realized per event. A separate analysis was conducted for each pollutant. The dependent variable in each analysis was the event load for the pollutant of concern (e.g., sediment, *E. coli*). Independent variables included in analysis for each pollutant were wetland (reference v. channelized), sample site (upstream v. downstream), cattle grazing during irrigation (present v. absent), average water inflow rate to the wetland for each event (L sec^{-1}), and the two-way interaction between wetland and sampling location. Inclusion of the interaction between wetland (reference v. channelized) and sample site (upstream v. downstream) in the analysis provided a direct test to determine if the load reduction by the channelized wetland was significantly different (greater or smaller) than by the reference wetland. Cattle presence or absence in the pasture and average water inflow rate into the wetland during each irrigation event were included in the analysis as covariates to examine the impacts of pasture irrigation and grazing management on overall event pollutant load, as well as to account for event-to-event variation due to these factors while simultaneously testing for differences in wetland filtration efficacy (interaction between wetland and sample location). A backward-stepwise approach was used to arrive at a final linear model for each pollutant with a criterion for inclusion in the model of $p < 0.1$. Standard residual plots were used to check that assumptions of linearity, normality and constant variance were met for each analysis. Sediment (TSS, VSS, NVSS) and *E. coli* event loads were $\log_{10}$ transformed to achieve normality.

To illustrate wetland efficacy and facilitate extrapolation of results to similar wetlands in the region, the percent load retention was calculated for each pollutant for each wetland as the difference between event inflow and outflow load. This resulted in a total of 22 observations of percent load reduction (11 irrigation events at two wetlands). The significance of differences in percent reduction was determined by the linear analysis described above. We also examined the possibility that wetland efficacy decreased as water flow rate increased (i.e., increased pollutant transport capacity) and inflow pollutant load increased (i.e., exceed wetland assimilation capacity). Specifically, we used regression to determine if percent event load reduction was dependent on average water inflow rate and pollutant inflow load. A separate analysis was conducted for each pollutant at each wetland. A backward-stepwise approach was used to arrive at a final regression model for each pollutant with a criterion for inclusion in the model of $p < 0.1$. Standard residual plots were used to check that assumptions of linearity, normality, and constant variance were met for each analysis.

Results and Discussion

Characterization of Irrigation Tailwater

Due to a combination of dilution and flushing effects, differences in irrigation onflow rates, and variable antecedent conditions (i.e., patterns of cattle grazing and time since last irrigation event), the concentrations of sediments, nutrients, and *E. coli* in tailwater entering the wetlands varied extensively (Table 1). Broad variation was observed in wetland inflow concentrations both within a given sampling event and between events (Fig. 2 and 3). Table 1 presents the range of concentrations observed for each pollutant in irrigation runoff (onflow) entering the wetland.

Median inflow concentrations for nitrate, TN, *E. coli*, TP, and SRP in the reference wetland were lower than median inflow concentrations in the channelized wetland. However, the median inflow TSS concentration was higher in the reference wetland than the channelized wetland (Table 1). Inflow TSS was primarily composed of NVSS averaging 77% for the reference wetland and 61% for the channelized wetland. Water pH ranged from 6.6 to 7.8 with a mean pH of 7.3 for inflow to both wetlands. Ammonium concentrations were generally at or near the detection limit (10 µg L^{-1}) in all samples, so this constituent was not considered in any further analyses.

To illustrate the raw data structure from the reference and channelized wetlands, influent and effluent flow and concentration profiles for representative high and low inflow events are shown for *E. coli* (Fig. 2) and nitrate (Fig. 3). Irrigation water collection events ranged in duration from 4.5 to 13.5 h. Events with lower inflow rates typically had longer collection times. At each wetland inflow sampling location, the beginning of irrigation runoff events was characterized by a sharp concentration peak. Following this peak, concentrations gradually declined throughout remainder of the event (Fig. 2 and 3). This concentration profile is indicative of a flushing effect for pollutants in runoff from irrigated pastures (Tate et al., 2000; Gannon et al., 2005).

In most events, the wetlands effectively mediated this flushing effect for most constituents by greatly decreasing initial peak pollutant concentrations, in addition to reducing concentrations throughout the sampling event. For example, in the representative events shown in Fig. 2, initial peak concentrations for *E. coli* were 67 and 94% lower in wetland effluent than influent in the high and low flow event, respectively. In the channelized wetland, the initial peak concentration for *E. coli* diminished by slightly more than 50% in each of these two events (Fig. 2). Similarly, the peak nitrate concentration decreased by 68 and 95% in the reference wetland at high and low inflow rates, respectively in the reference wetland (Fig. 3). On the other hand, peak nitrate concentrations were affected minimally in the two example events shown for the channelized wetland (Fig. 3).

Wetland Efficacy and Trends of Pollutant Retention

Biogeochemical processes that occur within wetlands can effectively remove a variety of pollutants from the water column. These processes include (i) sedimentation and burial (adsorbed P, pesticides, suspended sediments, particulate organic carbon, pathogens), (ii) microbial transformations to gaseous forms (denitrification, methanogenesis), (iii) plant and microbial uptake of nutrients, (iv) microbial degradation (oxidation–reduction) of pesticides and other organic compounds, and (v) predation within the food web (pathogen consumption). Quantification and comparison of influent and effluent event loads assessed the combined impacts of these wetland processes.

Event loads for pollutants entering and leaving the wetland during irrigation inflow events are shown for the reference (Fig. 4) and channelized wetland (Fig. 5). In the reference wetland, total pollutant loads in wetland effluent were smaller than influent loads for all 11 sampled events (with the

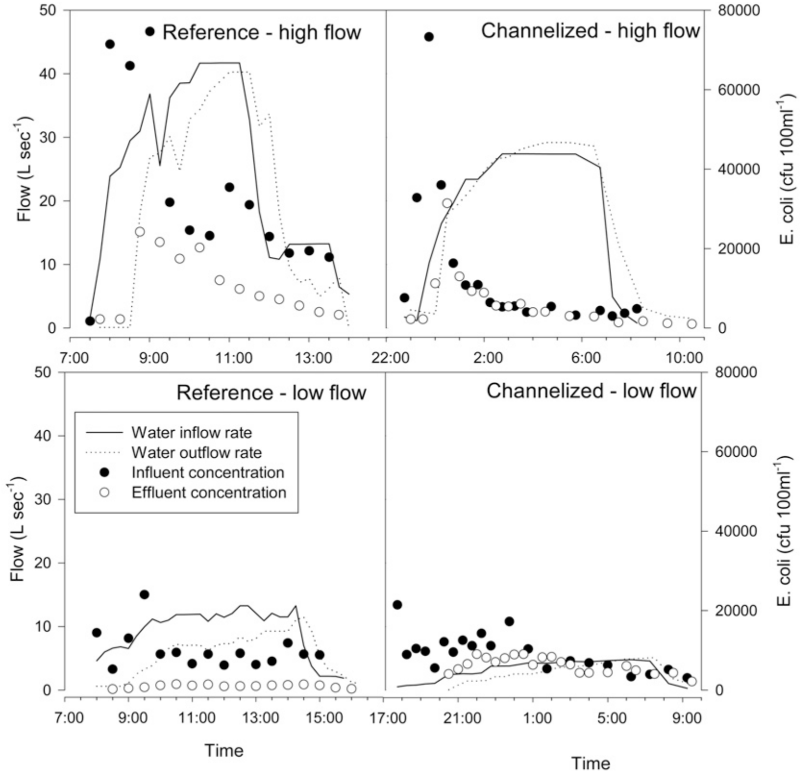

Fig. 2. ***E. coli*** **concentration and flow profiles for representative irrigation events from a high and low irrigation inflow event to the reference and channelized wetlands.**

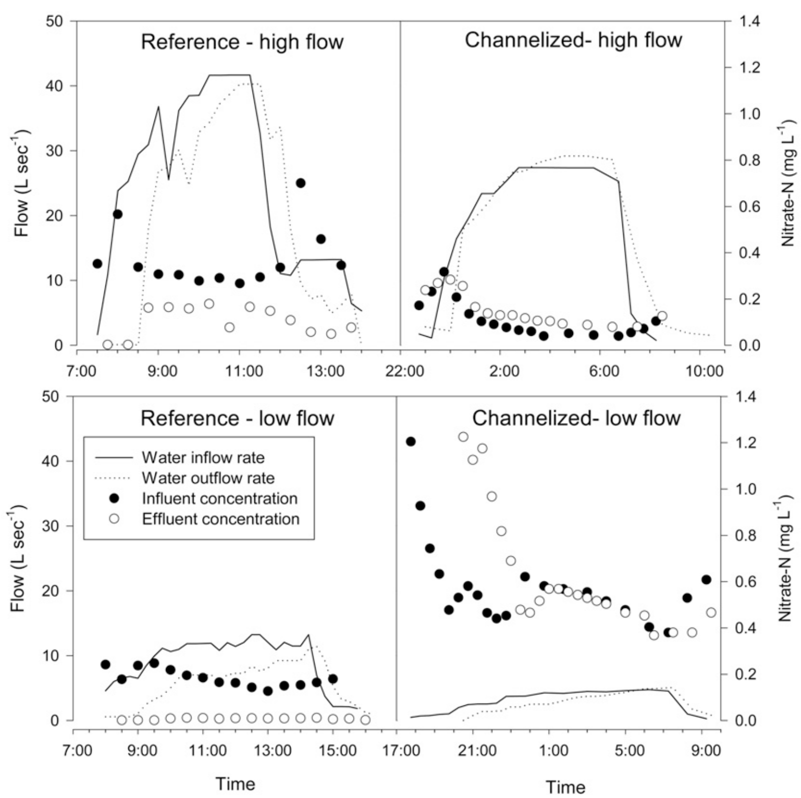

Fig. 3. Nitrate-N concentration and flow profiles for two representative irrigation events representative of high and low inflow events to the reference and channelized wetlands.

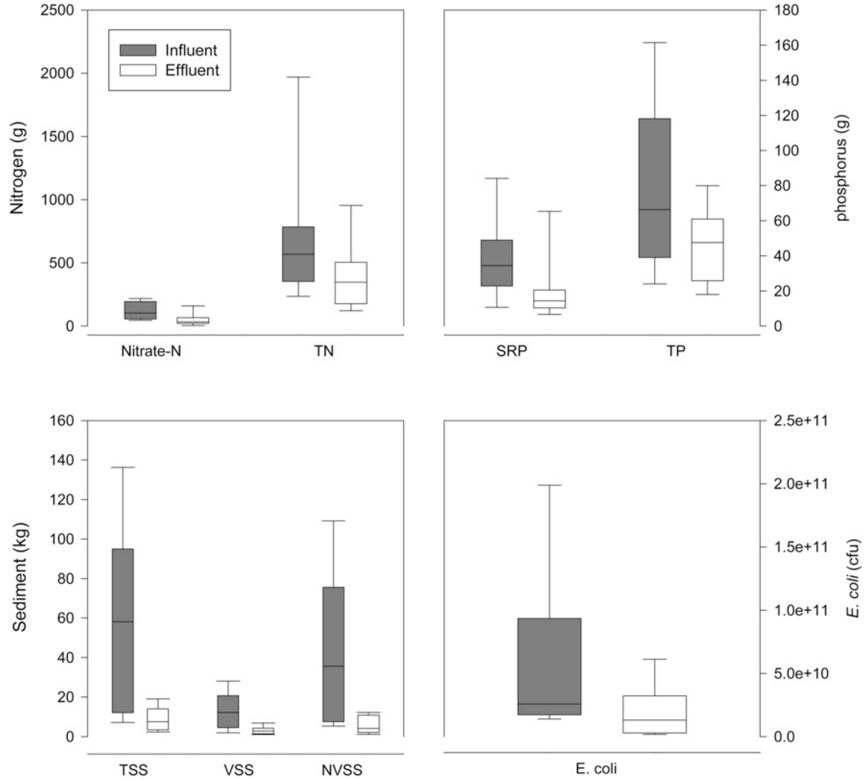

Fig. 4. Comparison of mean inflow and outflow loads for pollutants in the reference wetland (*n* = 11). Box plots show median and 25th and 75th percentiles. Whiskers represent 10th and 90th percentile.

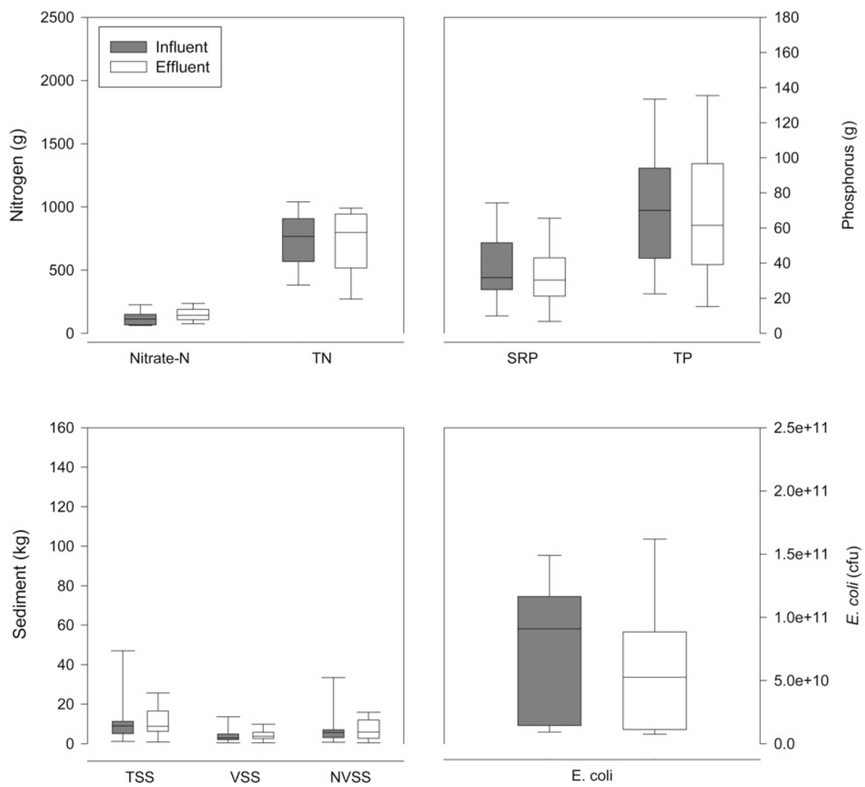

Fig. 5. Comparison of mean inflow and outflow loads of pollutants in the channelized wetland (*n* = 11). Box plots show median and 25th and 75th percentiles. Whiskers represent 10th and 90th percentile.

exception of TP in a single event and SRP in a single event). The channelized wetland was a net source for each pollutant in at least one, and up to six, of the 11 sampling events. In the channelized wetland, only *E. coli* loads consistently decreased from flow through the wetland (Fig. 5). The magnitude of event loads was positively correlated with the irrigation onflow rate; higher onflow rates generally resulted in greater pollutant loads.

Differences between water inflow and outflow volumes during events were generally small for both wetlands. In the reference wetland, on average, the surface inflow volume was slightly larger than the surface outflow volume during irrigation events (11.9 ± 7.7%; mean ± SD). In the channelized wetland, surface flow water volumes entering and leaving the wetland were very similar (0.0 ± 9.2%).

Table 2 reports the results of the final general linear analysis for each pollutant, testing for significant differences in wetland filtration efficiency, and the impact of irrigation and grazing management on overall event loads. Figure 6 illustrates the percent reduction of each pollutant by each wetland, as well as significant differences between wetlands based on general linear analysis reported in Table 2.

Wetlands in this study were naturally occurring rather than constructed. These wetlands were small and had short HRT (<2 h). However, sediment and nitrate retention efficiencies, particularly in the reference wetland, were similar to efficiencies observed from surface flow constructed wetlands receiving municipal wastewater (Kadlec and Knight, 1996). In the reference wetland, mean percent retentions (calculated as the difference between the pollutant load entering and exiting the wetland from events of all inflow rates) were between 35 and 77% (Fig. 6). Specifically, during irrigation events the reference wetland decreased loads of total suspended sediments, nitrate, and *E. coli* on average by 77, 60, and 68%, respectively. Percent retentions provided by the channelized wetland were widely variable with the mean values ranging from −37 (net source) to 25% (Fig. 6). On average, the channelized wetland was a net source for both sediment and nitrate while mean retention for *E. coli*, TP, and SRP was 25, 7, and 11%, respectively (Fig. 6).

The significant wetland by sample location interaction for *E. coli*, NO_3, TN, TP, TSS, VSS, and NVSS reported in Table 2 indicates that the relative difference between upstream and downstream sample location at the channelized wetland was significantly different compared to the reference wetland ($p < 0.10$). Figure 6 illustrates these significant differences as percent pollutant load reduction. The channelized wetland removed less of these pollutants, and in the case of sediment and nitrate the channelized wetland was on average a source of these pollutants. While both wetlands decreased SRP loads in general, they did not differ significantly in their efficiencies to decrease the SRP load ($p = 0.13$).

Pollutant Retention Processes

The largest retention efficiencies were observed in the reference wetland for suspended solids followed by *E. coli* and nitrate (Fig. 6, Table 2). Sedimentation, or settling, that occurred from decreased flow rates as water passed through the wetlands may have been responsible for retention of sediment and adsorbed pollutants including P and *E. coli* (Kay et al., 2005). Sediment retention remained high as inflow rates increased in the reference wetland. However, in the channelized wetland, a net export of sediment occurred during some events, but *E. coli* retention was still observed indicating that sedimentation may not have been the primary method of *E. coli* retention. One potential explanation for the larger *E. coli* retention is that sediment exiting the wetland may not have been the sediment that entered the wetland, so that both sediment deposition and channel erosion were occurring along the wetland reach. However, resuspension of sediment can also remobilize *E. coli* that was stored in the sediment, as fecal coliforms can persist in stream bed sediments up to at least 6 wk (Muirhead et al., 2004; Bai and Lung, 2005; Jamieson et al., 2005).

Retention of NVSS tended to be greater than retention of VSS. Retention of both NVSS and VSS were positively correlated with the magnitude of the event load. The proportion of the nonvolatile component (NVSS) of total suspended solids decreased by 17 ± 6% (mean ± SD) as water passed through the reference wetland (from 77 to 60% NVSS), indicating that NVSS was preferentially retained through sedimentation. In contrast, in the channelized wetland, the proportion of NVSS was nearly equal (3 ± 5%) in effluent (59%) and influent (62%) loads.

A portion of the *E. coli* suspended within the water column was likely attached to sediment particles (Ferguson et al., 2003; Tyrrel and Quinton, 2003; Collins et al., 2005). Therefore, retention of *E. coli* can be attributed to some extent to sedimentation. While *E. coli* attachment was not quantified for this system, previous studies assumed that adsorption of fecal coliforms to sediment particles in nonpoint source municipal and agricultural land runoff ranged between 50 and 90% in various transport and deposition models (Steets and Holden, 2003; Bai and Lung, 2005). Auer and Niehaus (1993) found that the majority of fecal coliforms were attached to small and medium sized particles ranging from either 0.45 to 1 μm or 6 to 10 μm in diameter (Gannon et al., 1983). As retention rates of *E. coli* exceeded retention of sediment in both wetlands,

Table 2. Result of general linear analysis to determine differences in filtration efficacy between a channelized and reference wetland, as well as the impacts of irrigation and grazing management on loads of *E. coli*, nitrate, total N, SRP, total P, total suspended solids (TSS), volatile (VSS) and nonvolatile suspended solids (NVSS) in runoff from flood irrigated foothill pastures grazed by beef cattle. Analysis is based on 44 event loads (11 irrigation events by 2 wetlands by 2 sites) collected at the UC Sierra Foothill Research and Extension Center in Yuba County, CA. Final linear model for each pollutant analyzed (e.g., *E. coli*, nitrate) is reported. *P*-value (*P*) reflects the significance that each independent variable has on event load for each pollutant.

	$\log_{10}$ (*E. coli*)		Nitrate		TN		SRP		TP		Log_{10} (TSS)		Log_{10} (VSS)		Log_{10} (NVSS)	
Independent variable	Coefficient‡	*P*	Coefficient	*P*	Coefficient	*P*	Coefficient	*P*	Coefficient	*P*	Coefficient	*P*	Coefficient	*P*	Coefficient	*P*
Wetland																
Reference†	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–
Channelized	0.4 (0.1)	0.003	20 (22)	0.415	185 (126)	0.148	14 (5)	0.009	10 (11)	0.366	−0.4 (0.1)	0.001	−0.3 (0.1)	0.011	−0.5 (0.1)	0.001
Sample site																
Upstream†	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–
Downstream	−0.6 (0.1)	<0.001	−63 (22)	0.007	−286 (122)	0.024	−10 (5)	0.038	−34 (11)	0.004	−0.7 (0.1)	<0.001	−0.5 (0.1)	< 0.001	−0.8 (0.1)	<0.001
Average inflow (L sec^{-1})	0.05 (0.00)	<0.001	3 (1)	0.016	23 (5)	<0.001	1.4 (0.3)	<0.001	3.0 (0.5)	<0.001	0.04 (0.01)	<0.001	0.04 (0.00)	< 0.001	0.04 (0.00)	<0.001
Cattle grazing during event																
Absent†	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–
Present	0.5 (0.1)	<0.001	30 (17)	0.093	351 (97)	<0.001	2 (5)	0.745	7 (6)	0.375	0.23 (0.02)	0.019	0.25 (0.08)	0.005	0.2 (0.1)	0.065
Wetland x sample site																
Channelized x upstream†	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–	0.00	–
Channelized x downstream	0.4 (0.2)	0.015	83 (31)	0.016	276 (172)	0.075	14 (9)	0.130	32 (14)	0.043	0.7 (0.2)	<0.001	0.6 (0.2)	0.000	0.8 (0.2)	0.002
Intercept	9.4 (0.2)	<0.001	70 (24)	0.008	47 (159)	0.768	4 (7)	0.588	16 (12)	0.204	0.6 (0.2)	0.001	0.02 (0.13)	0.875	0.4 (0.2)	0.032

† For categorical variables (wetland, sample site, cattle grazing during event, and wetland by sample site interaction) one level of the categorical variable was set as the referent condition (wetland = reference, sample site = upstream, cattle grazing during event = absent) to which the other level is compared (wetland = channelized, sample site = downstream, cattle grazing during event = present).

‡ The regression coefficient (1 standard error) for each independent variable introduced in the analysis. For categorical variables, the coefficient for the referent level is 0.00, and the coefficient for the other level indicates the mean difference of that level from the referent level. For continuous variables (average inflow rate) the coefficient value reflects the change in unit pollutant load associated with each incremental unit change in the independent variable.

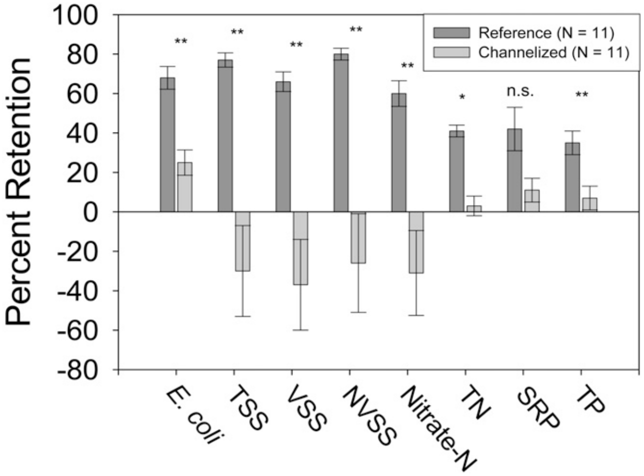

Fig. 6. Comparison of percent retention for all pollutants by the reference (n = 7) and channelized wetland (n = 7). (= $p < 0.05$; * = $p < 0.1$; n.s. = $p > 0.1$. Error bars indicate 1 SE.)**

other processes likely influenced the fate and transport of *E. coli* to result in the greater than 90% load retentions observed during low inflow events to the reference wetland. In addition to sedimentation, *E. coli* retention may be attributed to infiltration, predation, inactivation by UV radiation, and adsorption to particles and vegetation (Gerba et al., 1999).

Particulate phosphorus can be retained through sedimentation of adsorbed P which can occur readily in wetlands and buffer strips (Uusi-Kamppa et al., 1997). Sorption to bottom sediments can be an important retention mechanism for retention of SRP (Macrae et al., 2003). The total mass of TP retained was greater than the mass of SRP retained during events in each wetland (Fig. 4 and 5). Partially due to this difference in mass, the mean percent retention efficiencies for TP and SRP were similar within each wetland (Fig. 6).

Retention of nitrate may occur through denitrification in the anaerobic, hyporheic zone in flooded wetland ecosystems (Gersberg et al., 1983). In a quantification of nitrogen transformation processes, Cooke (1994) found that nitrate removal was primarily due to denitrification, but a combination of transformation to ammonium and assimilation accounted for about a third of the nitrate removal that was observed. Nitrate retention observed from load calculations was confirmed for the reference wetland through the higher recovery rates of bromide than nitrate in the Br-NO_3 injection (Fig. 7).

Bromide mass was highly conserved within all Br-NO_3 injection events from both wetlands with a calculated recovery of 100 ± 5% downstream of the wetland relative to the upstream sampling location. In the reference wetland, nitrate recovery standardized to the recovery rate of bromide at the downstream location ranged from 59 to 77% (Fig. 7a) with the higher recovery rate occurring at the greater inflow rate. On the other hand, in the channelized wetland, nitrate recovery was similar to bromide recovery (Fig. 7b). The standardized nitrate recovery ranged from 80 to 102%, meaning that less nitrate was retained in the channelized wetland during the injections than in the reference wetland. In some irrigation events on the channelized wetland, a net export of nitrate was observed. Transformation of organic forms to mineral forms within the wetland environment can result in export of nitrate from a wetland. In systems that undergo repeated cycles of oxidation and reduction due to periodic inundation (as is the case with these wetlands), mineralization from organic matter can release both nitrate and SRP.

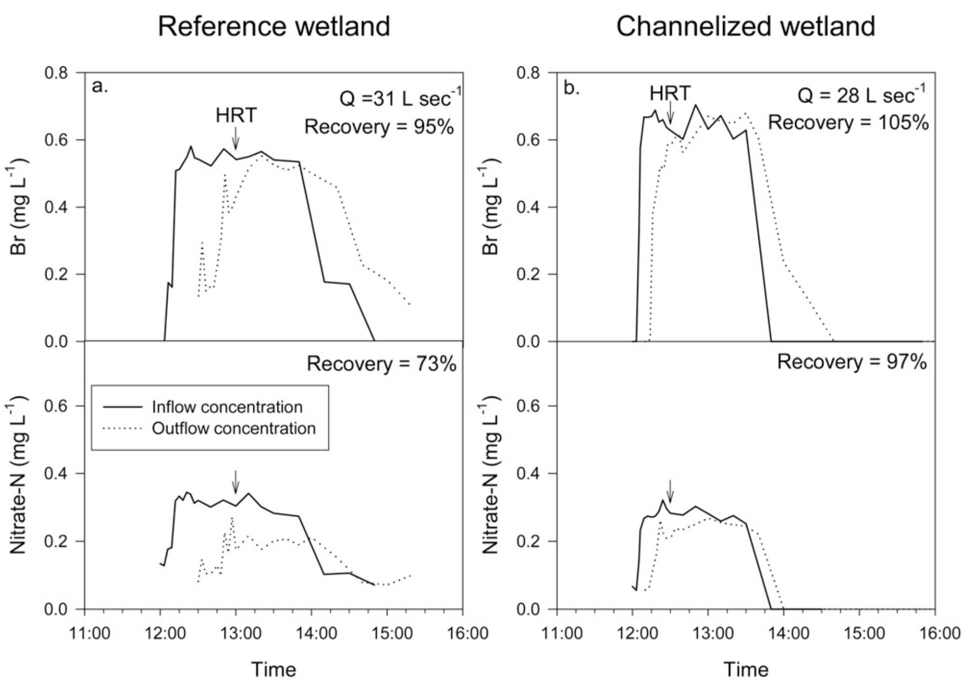

Fig. 7. Bromide and nitrate concentrations in inflow and effluent measured during Br/NO_3 injection studies in the reference (a) and channelized (b) wetland during events with similar water inflow rates (28 and 31 L sec^{-1}, respectively).

Impacts of Grazing and Irrigation Management on Event Loads

The presence of cattle grazing in the pasture during irrigation inflow events resulted in significantly ($p < 0.1$) greater *E. coli*, TN, TSS, VSS, NVSS, and NO_3 event loads compared to irrigation events where cattle were absent from the pasture during irrigation (Table 2). For example, examination of the coefficient for cattle present in the pasture for the nitrate model reported in Table 2 indicates that on average 30 g more nitrate load was associated with irrigation events which occurred in the presence of cattle compared to events without cattle ($p = 0.093$). Cattle grazing during irrigation inflow events was not a significant predictor in the model developed for total P ($p = 0.375$) or SRP ($p = 0.745$) (Table 2). Increases in the average inflow rate (L sec^{-1}) during an irrigation event were significantly associated with increased event loads for all pollutants ($p < 0.10$) (Table 2). For example, the coefficient for average inflow rate in the final nitrate model reported in Table 2 indicates that with each incremental (1 L sec^{-1}) increase in average water inflow rate the model predicts an average increase in nitrate load of 3 g at both sample locations of each wetland ($p = 0.016$).

Effect of Water Inflow Rate and Pollutant Inflow Load on Wetland Efficiency

Hydraulic residence time (HRT) has often been identified as a major factor that can influence the efficiency of a wetland to retain pollutants and contaminants (Greenway and Woolley, 1999; Blahnik and Day, 2000; Jordan et al., 2003; Toet et al., 2005). Longer residence times, as well as lower hydraulic loading rates for a given wetland, generally result in greater retention of pollutants (Blahnik and Day, 2000; Knight et al., 2000). Large runoff events may essentially overwhelm the assimilative capacity of a wetland (Woltemade, 2000). However, Hey et al. (1994) suggested that a relationship between inflow rate and retention efficiency may not be apparent if the system is operating well below its loading rate capacity.

Shorter residence times and an entrenched channel likely contributed to the lower pollutant retention efficiency in the channelized wetland. We observed consistently longer HRTs in the reference wetland than in the channelized wetland. Based on bromide recovery curves during events with similar inflow rates (28 and 31 L sec^{-1}), the HRT in the reference wetland was 54 min while the HRT in the channelized wetland was 25 min (Fig. 7). In both wetlands, lower water inflow rates resulted in longer hydraulic residence times. The HRT in the reference wetland ranged from 38 min at the highest inflow rate (43 L sec^{-1}) to over 2 h at the lowest flow rate (13 L sec^{-1}). In the channelized wetland, the HRT ranged from 18 to 47 min at flow rates of 34 and 8 L sec^{-1}, respectively.

Longer residence times allow greater time for processes such as microbial uptake and nutrient transformations to occur than in the channelized wetland. In addition, decreased flow rates that occurred with longer residence times allowed for greater settling of particles and associated contaminants (i.e., TP, *E. coli*). In the channelized wetland, low flows had poor connectivity with the majority of the wetland as water remained almost entirely within the entrenched channel. Wetland channel complexity, which increases transient storage capacity, was qualitatively examined by the shape of the Br curves during the Br-NO_3 injection. A wetland with a small transient storage zone should exhibit a "square" curve for Br over time, whereas the rising and falling limb of the Br curve should be more spread out as channel complexity and transient storage increase (Grimm et al., 2005). The rising and falling limbs of the bromide recovery curves from the reference wetland showed a greater spread (Fig. 7a) than the channelized wetland which exhibited a relatively "square" curve (Fig. 7b).

Average inflow rates to the wetlands varied by a factor of 10 (from 0.006 to 0.06 $m^3 s^{-1}$) resulting in a more than twofold change in hydraulic residence times. Retention efficiencies were typically greater when residence times were longer, but this correlation was not observed for all pollutants. In the reference wetland, lower water inflow rates typically resulted in greater retention efficiencies for nitrate, *E. coli*, and sediments. Linear regression analysis revealed significant, negative relationships between average event inflow rate and percent retention of *E. coli* and NO_3–N load ($p < 0.05$). Percent retention of both VSS and NVSS was significantly decreased as inflow rate increased, and increased as inflow event load increased at the reference wetland ($p < 0.05$). No significant linear relationships were found between percent retention, water inflow rate, or inflow pollutant load for SRP, TN, or TP at the reference wetland. In the channelized wetland, trends toward increasing efficiency with decreasing inflow rate were apparent, but the relationships were not significant for any pollutant except TN ($p = 0.041$).

Management Implications

Irrigation and grazing management decisions were shown to affect pollutant loading to the wetlands. Higher irrigation inflow rates typically resulted in higher concentrations and larger pollutant loads in tailwaters leaving the pastures during an irrigation event. Grazing the pasture during an irrigation inflow event increased *E. coli*, sediment, nitrate, and total N loads entering the wetland. By employing a combination of pasture and grazing management techniques and allowing for flow through a healthy wetland, large improvements in water quality can be realized. Best management plans may benefit from considering these pasture, grazing and wetland management strategies as parts of a regime designed to improve water quality of rangeland runoff.

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