Land Use and Water Quality Relationships in the Lower Little Bow River Watershed, Alberta, Canada

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Water quality in the Lower Little Bow River was monitored to determine if irrigation return flow streams had a significant impact on river water quality and to examine relationships between land use and water quality in this diverse agricultural watershed. Water samples were collected weekly or biweekly during the irrigation season and monthly in winter for three years. A comprehensive land use assessment was also completed. Significant differences in flows, and in nutrient and bacteria loads, were found along the mainstem of the river following the inflows of irrigation return water; however, differences in concentrations were only significant in a drought year when mainstem flows were reduced. Pearson correlations among land use, soil types, and water quality variables identified significant positive relationships between the proportion of cereals, irrigated land, and confined feeding operation (CFO) density and maximum concentrations of total nitrogen (TN), nitrate-nitrogen, and total phosphorus (TP) that were observed during runoff events. Most nutrient variables were inversely related to the proportion of native prairie. The variation in maximum TP and median dissolved P concentrations was largely explained by the proportion of cereals in the sub-basin, while the variation in maximum and median TN concentrations was explained by the proportions of irrigated land and native prairie, respectively. Microbiological variables were not related to any of the measured variables, suggesting that factors influencing bacteria populations operate at different scales.

Key words: land use, phosphorus, nitrogen, coliform bacteria, irrigation

Introduction

During the past two decades, there has been increasing recognition of the impacts of agricultural activities on water quality (Carpenter et al. 1998; Canada-Alberta Environmentally Sustainable Agriculture Agreement 1998). Coupled with this awareness, there has been a rise in agricultural intensity, with expanded areas of irrigated land and input-demanding row crops, along with the concentration of livestock into large confined
feeding operations (CFOs), especially in Alberta (Wentz et al. 2001; Statistics Canada 2002).

Irrigation practices, which alter runoff and drainage patterns, can influence water quality. High-quality irrigation source waters can augment flows and dilute contaminants. However, irrigated areas are often associated with increased agricultural inputs, such as fertilizers and pesticides (Bevans et al. 1998; Cuffney et al. 2000). Irrigation return flow streams, composed of excess supply water, surface and subsurface runoff, tile drainage, and groundwater, can adversely affect water quality in receiving bodies due to excess nutrients, sediment, salts, pesticides, and pathogenic bacteria (Bevans et al. 1998; Cessna et al. 2001).

In Alberta, approximately 4% or 600,000 ha of the agricultural land is irrigated, making it the largest irrigated area in Canada. Irrigation increases the diversity of crops that can be grown in southern Alberta’s semi-arid climate. Additionally, irrigation sources provide water for a large number of CFOs, mainly beef cattle feedlots. These operations produce large quantities of manure that are applied to the land as fertilizer, and can be significant sources of nutrients and bacteria to surface waters when applied in amounts exceeding crop requirements (Carpenter et al. 1998).

Several attempts have been made to relate land use and water quality in agricultural regions, with varying degrees of success. Many of these studies have been conducted in mixed land use watersheds, and have focused primarily on the proportion of agricultural versus non-agricultural land (Osborne and Wiley 1986; Castillo et al. 2000). In general, stronger relationships have been observed between land use and nitrogen, especially nitrate-nitrogen (NO₃-N), than between phosphorus variables and land use (Johnson et al. 1997; Bouraoui et al. 1999). In the Chesapeake Bay region, stream NO₃-N was related to cropland and baseflow; however, phosphorus concentrations were related only to suspended solids and not land use (Jordan et al. 1997). In Texas, stormwater nitrogen concentrations were related to several individual and composite variables characteristic of intensive agriculture; however, phosphorus concentrations in stormwater flows were related only to the proportion of land used for dairy waste applications (McFarland and Hauck 1999).

Few studies have attempted to relate bacterial indicators (fecal coliforms, *E. coli*) directly to land use; more often, the presence of pathogenic bacteria is indirectly related to land use through epidemiological studies. For instance, Michel et al. (1999) found a relationship between the incidence of verocytotoxigenic *E. coli* (VTEC) infections and the density of cattle in Ontario. However, in one of the few studies examining bacterial prevalence and land use practices, Johnson et al. (2003) found that there was no direct relationship between bacterial prevalence (*E. coli* O157:H7, *Salmonella* spp.) and manure output.

Our study had two major objectives. The first was to determine the impact of irrigation return flow streams on water quality in the Lower Little Bow River (LLB). Although previous studies have shown only minor effects of nutrients from irrigation return flows on receiving water
bodies (Joseph and Ongley 1986; Cessna et al. 2001), this study examined irrigation return flows from a more intensive agricultural region and also examined microbiological variables. Our second objective was to determine the relationships among land use, soil type, and water quality in a diverse agricultural watershed.

Materials and Methods

Site Description

The Lower Little Bow River watershed encompasses 55,664 ha north of Lethbridge in southern Alberta (Fig. 1). The Travers Reservoir at the northern end of the watershed is the principle source of water into the system. Flows are regulated to maintain steady-state flows of approximately 0.57 m$^3$ s$^{-1}$ in the winter and 0.85 m$^3$ s$^{-1}$ in the summer (Fig. 2). Release rates are more variable in the summer, depending on irrigation demand, downstream inputs from irrigation, and quantity of supply waters. Smaller inputs are added to the basin from Lethbridge Northern Irrigation District (LNID) supply reservoirs.

Soils in the watershed are predominantly Orthic Dark Brown Chernozemic, with minor areas of Orthic Brown Chernozemic and Solonetizic soils in the upper reaches. Terrain in the region is hummocky, with poorly to well-defined knobs and kettles. The wide river valley has inclined, steep slopes of high relief throughout the length of the river. Within the river valley, soils are coarse and undefined. In the northern portions, the stream channel varies from a meandering stream to a braided channel with wide floodplains, but changes to a more defined channel with a confined floodplain near the confluence with the Oldman River.

Seven sub-basins were delineated within the watershed using air photos, LNID maps, and ground-truthing. The sub-basins range in size from 3367 to 10,792 ha. While most of these watersheds contribute directly to the Lower Little Bow River, two predominantly irrigated watersheds, S1 and LB4-2, initially enter irrigation return flow streams before flowing into the mainstem of the river (Fig. 1). The irrigation return flow streams have V-shaped channels with no floodplains. Several other smaller return flow streams enter directly into the river; however, these have relatively small catchments that were not partitioned out for land use analysis.

Flow and Water Quality Monitoring

Six mainstem sampling sites (LB1-LB6), and two irrigation return flow sites (S1, LB4-2) were monitored at the outlet of each sub-basin. Each site was equipped with a stilling well, a staff gauge and a datalogger (ChartPac CP-XA, Lakewood Systems Ltd.) programmed to record gauge heights every 20 min. Flow metering was conducted throughout the open water season at each site using a Swoffer 3000 current meter (Swoffer Ltd.) to calibrate gauge heights and flow.
Fig. 1. The Lower Little Bow River basin with sub-basin boundaries and sampling sites.
Water samples were collected either biweekly (1999) or weekly (2000, 2001) during the irrigation season (mid-April to mid-October). Additional samples were taken during high flows following major precipitation events, which occurred on June 3 to 4, 1999; August 11 to 12, 1999; September 2 to 4, 2000; and June 4 to 6, 2001. In winter (November to March), samples were collected monthly, provided water was flowing. Intensive sampling was planned for spring runoff periods; however, spring runoff was not observed in any year.

Samples were collected from the centre of the stream channel at a depth of 30 cm below the surface in 2-L polyethylene bottles that had been thrice rinsed with sample water. Samples were then transported in coolers to the AAFRD Irrigation Branch lab in Lethbridge, Alta. Water temperature, staff gauge heights, and other physical parameters (turbidity,
weather, presence of livestock) were recorded at the time of sampling. In the lab, samples were filtered through 45-µm filters, and then analyzed for pH, electrical conductance (EC), nitrate (NO\textsubscript{3}-N), nitrite (NO\textsubscript{2}-N), ammonium (NH\textsubscript{4}-N), orthophosphate (PO\textsubscript{4}-P), and dissolved phosphorus (DP). Total Kjeldahl nitrogen (TKN) and total phosphorus (TP) analysis were conducted on unfiltered samples.

Bacterial samples were collected in autoclaved bottles that contained sodium thiosulphate. Analyses for fecal coliforms and \textit{E. coli} were conducted at the Provincial Laboratory of Public Health in Calgary, Alta., using the membrane filtration method (American Public Health Association 1995).

**Land Use Data**

Crop information was derived from Landsat Thematic Mapper images collected on July 8, August 18, and September 11, 1998. The temporal sequencing of images provided information on specific crop signatures; for example, the canola signature was extracted from the early summer image whereas root crops, such as sugar beets and potatoes, were identified in the late summer image. The information was assembled and classified using unsupervised classification and masking techniques. Ground-truthing was performed using a combination of reference information available from the County of Lethbridge, Alberta Agriculture, Food and Rural Development (AAFRD) data, and strategically chosen ground sites; the final product retained an accuracy of approximately 90%. Areas of native prairie were identified through the Native Prairie Vegetation Inventory, compiled by Alberta Sustainable Resource Development (ASRD). The information was derived from 1:30,000 aerial photographs, including extensive field verification by ASRD agrologists. The accuracy of the native prairie vegetation information was approximately 92%.

Data on irrigation were compiled from the Irrigation Infrastructure Management System maintained by AAFRD and the LNID. Confined feeding operation (CFO) data were based on information collected by the County of Lethbridge and compiled by the Prairie Farm Rehabilitation Administration (PFRA). The same sources were used to calculate animal manure units (AMUs), which are designed for cross-species comparisons. One AMU is the amount of manure required to fertilize 0.4 ha of corn based on a rate of 73 kg TN per year. County population data were collected by Statistics Canada in the 1996 Federal Census. Soils and landform data were assembled from the Agricultural Regions of Alberta Soils Inventory Database (AGRASID) 3.0 database (Alberta Soil Information Centre 2001). All data were compiled for use in Environmental Systems Research Institute (ESRI) Arc/Info and ArcView software.

**Data Analysis**

The program FLUX v. 5.1 (U.S. Army Corps of Engineers 1999) was used to determine loads and flow-weighted mean concentrations at each
site. The International Joint Commission (IJC) equation (Bodo and Unny 1983, 1984) was used to define the relationship between concentration and flow. Flows were stratified by dates to further refine this relationship.

Due to spatial and temporal autocorrelation of the data, mixed model analysis (Proc Mixed; SAS Institute Inc. 1999) was employed. This specialized form of general linearized models (GLM) can be used to account for both fixed and random effects in repeated measures data. The variable of interest and its covariable structure are modelled, and then the covariance is removed from the data (Littell et al. 1996). This technique is robust to missing data and can account for heterogeneous variances and correlation among observations, although data must satisfy other conditions of normality.

Mixed models were used to analyze log-transformed concentration and mass load data from the irrigation season. The models were run with site, year, and site x year as fixed effects. Mass loads for each sampling date were calculated by multiplying instantaneous sample flows and concentrations. Compound symmetry models (constant correlations among sites) were used to model the data. Significant differences among sites were determined from the probability of differences between least squared means (LSMeans) using the Tukey adjustment for multiple comparisons.

Land use, soil type and water quality variables were tested for normality using the Shapiro-Wilk test (p < 0.05) in Proc Univariate (SAS Institute Inc. 1999) and then transformed to normality where possible. Preliminary relationships were examined using Pearson correlations among normally distributed variables. Step-wise multiple regression analyses were then performed on variables showing significant relationships with land use.

Results and Discussion

Land Use

Agricultural intensity increases with distance downstream from the Travers Reservoir. The northern sub-basins (LB2 and LB3) are dominated by native prairie and large cow-calf operations (Table 1). Further downstream at LB3, dryland agriculture begins on the plains above the river valley. Irrigated land supplied by the LNID, along with CFOs, begin in sub-basin LB4-2 and occur mostly west of the river. Most of the water is applied using centre pivot sprinklers, but some wheel-move sprinklers and flood irrigation are also present. To the east of the Lower Little Bow River, rangeland and dryland farming are dominant with a few privately irrigated areas.

The proportion of irrigated land in the sub-basins ranges from 0% in LB2 and LB3 to 64% in S1 (Table 1). Consistent with this trend, the proportion of land under native prairie decreases from 74% in LB2 and 58% in LB3 to 4% in S1. Sub-basin S1 contained the largest number of CFOs, with 22 (mostly beef cattle feedlots) in 1998. CFOs were also present in LB4-2, LB5 and LB6 (Table 1).
<table>
<thead>
<tr>
<th>Variable</th>
<th>LB2</th>
<th>LB3</th>
<th>LB4-2</th>
<th>S1</th>
<th>LB4</th>
<th>LB5</th>
<th>LB6</th>
</tr>
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<tbody>
<tr>
<td>Area (km²)</td>
<td>33.7</td>
<td>93.8</td>
<td>102.9</td>
<td>85.7</td>
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<td>Population density (per km²)</td>
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<td>1.5</td>
<td>3.0</td>
<td>3.2</td>
<td>1.8</td>
<td>1.6</td>
<td>3.4</td>
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<td>Confined feeding operation (CFO) density (per km²)</td>
<td>0</td>
<td>0</td>
<td>0.02</td>
<td>0.26</td>
<td>0</td>
<td>0.13</td>
<td>0.05</td>
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<td>Animal manure unit (AMU) density (per km²)</td>
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<td>0</td>
<td>20.1</td>
<td>333.5</td>
<td>0</td>
<td>268.5</td>
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<td>Unclassified %</td>
<td>8.2</td>
<td>8.6</td>
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<td>7.0</td>
<td>7.2</td>
<td>5.6</td>
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<td>Cereal %</td>
<td>9.7</td>
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<td>2.0</td>
<td>1.0</td>
<td>1.6</td>
<td>2.4</td>
<td>3.9</td>
<td>0.5</td>
<td>1.6</td>
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<td>Forage %</td>
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<td>1.3</td>
<td>3.7</td>
<td>2.1</td>
<td>1.7</td>
<td>0.2</td>
<td>1.5</td>
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<td>Beet %</td>
<td>0.1</td>
<td>0.4</td>
<td>1.0</td>
<td>4.3</td>
<td>2.5</td>
<td>7.9</td>
<td>4.2</td>
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<td>Potato %</td>
<td>1.0</td>
<td>0.1</td>
<td>0.8</td>
<td>2.9</td>
<td>1.5</td>
<td>1.5</td>
<td>1.6</td>
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<td>Fallow %</td>
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<td>Native %</td>
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<td>58.3</td>
<td>12.7</td>
<td>3.6</td>
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<td>Irrigated %</td>
<td>0</td>
<td>0</td>
<td>18.6</td>
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<td>23.0</td>
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<td>Orthic D. Br. Chernozem Med.-Coarse, Glacio-Fluvial (OD_MC_FL)</td>
<td>3.8</td>
<td>11.0</td>
<td>0</td>
<td>0</td>
<td>6.7</td>
<td>17.7</td>
<td>4.7</td>
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<td>Orthic D. Br. Chernozem Med. Glacio-lacustrine (OD_ME_LC)</td>
<td>0</td>
<td>5.5</td>
<td>24.6</td>
<td>46.8</td>
<td>22.0</td>
<td>39.2</td>
<td>38.2</td>
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<td>Orthic D. Br. Chernozem Medium-Fine Till (OD_MF_TL)</td>
<td>23.9</td>
<td>17.5</td>
<td>46.7</td>
<td>44.1</td>
<td>28.6</td>
<td>0</td>
<td>0</td>
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<tr>
<td>Inclined steep single slope, high relief (I3h)</td>
<td>38.4</td>
<td>17.7</td>
<td>5.4</td>
<td>1.3</td>
<td>18.6</td>
<td>3.8</td>
<td>0</td>
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</tbody>
</table>
Hydrology

During the irrigation season, flows at LB2 and LB3 mirror those at the outlet of the Travers Reservoir (LB1), but are slightly lower (Fig. 2). Irrigation return flow streams (LB4-2, S1) from the LNID enter the system below LB3 (Fig. 1). The LNID also spills extra water into the river to supply irrigators on the east side of the river. Consequently, flows increase and become much more variable from LB4 to LB6 (Fig. 2). Releases from the Travers Reservoir were reduced throughout 2001 (Fig. 2C) due to minimal spring runoff and drought conditions in 2000 and 2001. Contributions from the LNID supply waters and return flow streams offset low releases from the Travers Reservoir in the lower reaches of the river. Mixed model analysis confirmed that flows below the return flow entrances were significantly higher than those upstream of the irrigated sub-basins in all three years (data not shown).

Flows within irrigation return flow streams were small relative to the river, and were much more variable both within and among years. Site S1 represented the largest average return flow on the Lower Little Bow River in 1999, with an average irrigation season flow of 0.24 m³ s⁻¹ (Fig. 2A). A reservoir outlet just upstream of S1 was accidentally left open for much of the summer of 1999. Therefore, flows in 2000 and 2001 were much lower, averaging 0.10 m³ s⁻¹ (Fig. 2B and C). Site LB4-2 was the largest return flow in 2000 and 2001, averaging 0.36 m³ s⁻¹ and 0.38 m³ s⁻¹, respectively.

Water Chemistry

Water released from the Travers Reservoir (LB1) had consistently low values of all water quality variables that never exceeded water quality guidelines. Increased water residence time and sedimentation improved water quality at the outlet of the reservoir compared to inflowing waters (Sosiak 2000). Due to its location downstream of the reservoir, site LB1 was not associated with a watershed and runoff from significant precipitation events had negligible effects on water quality.

Phosphorus

Total phosphorus (TP) levels increased downstream within the mainstem; however, the highest mean concentrations of TP were found in the irrigation return flow streams (Fig. 3A). For most sites, the June 1999 precipitation event produced the maximum concentrations observed during the three years. However, in the upper reaches (LB1–LB3), maximum concentrations were observed under low flow conditions in spring 2001. The mixed model analysis indicated a significant site x year interaction; therefore, each year will be discussed separately.

In 1999, the highest TP concentrations were observed at S1 and were significantly greater than all mainstem sites (Fig. 3A). This site was followed by LB4-2, which was significantly greater than upstream sites (LB1-LB3), but not downstream sites (LB4-LB6). There were also significant differences between downstream sites and most upstream sites (LB1-LB2).
In 2000, site LB4-2 had the highest TP concentrations and was significantly greater than all mainstem sites. Again, downstream sites were significantly greater than upstream sites, except for the difference between LB3 and LB4, which was not significant (Fig. 3A). Overall, TP concentrations were slightly lower in 2000 than the previous year due to dry weather conditions, except at LB4-2.

Drought conditions persisted in 2001; however, reduced flows in the mainstem resulted in higher mean concentrations at the upstream sites (LB1 to LB3), while concentrations in the lower reaches were similar to 1999 values. The pattern of TP concentrations in 2001 was similar to the pattern in 2000; however, all downstream sites, including the one immediately below the inflows of the two return flow streams were significantly different from upstream sites, indicating that the return flow streams may have adversely affected phosphorus concentrations in the mainstem in the low flow year.

Fig. 3. Bar charts of (a) mean TP concentrations and (b) loads in the Lower Little Bow sub-basin. Bars with different letters above them were significantly different (p < 0.10) from each other in the mixed model analysis.
In addition to climatic differences, the variability in return flow concentrations and flows contributed to the interaction between site and year. TP concentrations at S1 were significantly greater in 1999 than in 2000 and 2001. The flushing of a small reservoir upstream of S1 for much of the summer of 1999 contributed to this difference. Additionally, pipeline construction resulted in more water being diverted down the return flow at LB4-2 in 2000 and 2001 than in 1999 and the higher concentrations of TP being observed at this site.

Dissolved phosphorus concentrations exhibited a slightly different pattern than total phosphorus with few significant differences among mainstem sites. However, the irrigation return flow streams still had the highest concentrations in all three years (Fig. 4A). Patterns were slightly different among years and sites as confirmed by the significant site x year interaction in the mixed model analysis.

Dissolved phosphorus was significantly different between years, with lower mean concentrations observed in 2000. DP was elevated fol-

![Fig. 4. Bar graphs of (a) DP concentrations and (b) loads with significant differences denoted by the letters above the bars.](image-url)
lowing both the major precipitation event of June 3, 1999, and the smaller rainfall event on August 11 to 12, 1999, while in 2001 reduced flows contributed to the higher mean DP concentrations in the upper reaches of the basin. Maximum concentrations were observed at all mainstem sites in March 2001, but only irrigation season values (mid-April to mid-October) were included in the analysis.

Among the sites, LB4-2 and LB3 tended to have lower proportions of DP compared with other sites. This suggests that soil and/or channel erosion may be a factor at these sites. This observation is consistent with the higher flows observed at LB4-2 in 2000 and 2001 compared with 1999 and with field observations at this site. The stilling well had to be relocated in 2000 due to bank erosion.

Total phosphorus in the return flows was generally within the range reported by Joseph and Ongley (1986) for other return flows in Alberta (0.02–0.5 mg L⁻¹), but were greater than mean concentrations reported for return flow streams downstream of flood-irrigated fields in Alberta (Oosterveld and McMullin 1979) and Saskatchewan (Cessna et al. 2001). Total dissolved phosphorus has seldom been measured. DP concentrations were also within the range reported by Joseph and Ongley (1986) (0.007–0.130 mg L⁻¹), but were less than the mean value (0.044 mg L⁻¹) reported in the same study.

Due to higher flows in the lower reaches of the basins, mass loads of total and dissolved phosphorus were significantly greater in LB4, LB5, and LB6 than at upstream sites and most irrigation return flow streams (Fig. 3B, 4B). Among the irrigation return flow streams, S1 had the greatest loads of TP and DP in 1999, but the lowest loads in 2000 and 2001. The other return flow site, LB4-2, showed increasing loads of TP during the study period and had much higher loads of TP than DP, compared with mainstem sites. Estimates from the FLUX model suggest that LB4-2 accounted for 34% of the TP in 1999, 46% in 2000 and 75% in 2001, while S1 contributed approximately 29%, 14% and 18% in 1999, 2000 and 2001, respectively. Flow data from all three years were used to calibrate the model, which may account for the underestimation of the loads at S1 in 1999, compared with the instantaneous loads used in the mixed model analysis. Total loads at LB6 decreased during the three years, with TP loads in 2001 only 54% of that estimated for 1999. During the same time period, TP loads at LB4-2 increased slightly.

Nitrogen

Total nitrogen (TN) was less variable among sites than TP. Peak concentrations were observed at most sites following the precipitation event in June 1999; however, a few sites had higher values in March and April (LB1, LB2, and S1), especially in 2001. Samples were usually below guidelines for TN, except for occasional samples in October, March, and April.

The return flow streams, S1 and LB4-2, had the highest TN concentrations in all three years. Site S1 had the highest TN values in 1999 and 2001, which were significantly greater than concentrations at all other
sites, except for LB4-2 in 2001 (Fig. 5A). In 2000, S1 ranked second out of the 8 sites, while LB4-2 had the highest TN values. The site immediately downstream of the irrigation return flows was only significantly different in 2001, suggesting that the irrigation return flows only affected mainstem TN concentrations in low flow years. There was a significant effect of site and a site x year interaction, but no significant year effect. TN concentrations at S1 were greater in 1999 than in 2000.

Nitrate-N concentrations were not a concern in the Lower Little Bow River watershed. This fraction rarely made up a significant proportion of the TN, except at the beginning of the irrigation season in smaller irrigation return flow streams (AAFRD, unpublished data). The irrigation return flow streams had the highest concentrations, with S1 having the highest concentrations in 1999 and LB4-2 having the highest concentrations in 2000 and 2001. Both return flow streams had significantly greater concentrations of NO₃-N than most of the mainstem sites. There were no significant differences among mainstem sites as median values were below detection limits (0.04 mg L⁻¹) in all years.

Fig. 5. Bar graphs of (a) TN concentrations and (b) loads with significant differences denoted by the letters above the bars.
Several studies of irrigation systems have reported low or undetectable nitrate values unless there is a substantial groundwater contribution to the return flow stream. Denitrification has been cited as a possible explanation for the low values, which are often lower than those found in supply waters (Oosterveld and McMullin 1979; Greenlee et al. 2000; Cessna et al. 2001). Total nitrogen has rarely been examined.

Total nitrogen loads followed a very similar pattern to TP loads, with the greatest loads occurring in the lower reaches following the entrance of the irrigation return flow streams (Fig 5B). Again, S1 had the highest nitrogen load among the return flow streams in 1999, while LB4-2 had the highest loads in 2000 and 2001. However, loads from both sites were comparable to the lowest loads on the mainstem at site LB1. Flux-estimated TN loads at LB4-2 were 19% of the load at LB6 in 1999, 25% in 2000, and 42% in 2001, while loads at S1 were 21% in 1999, 11% in 2000, and 14% in 2001.

**Bacteria**

Fecal coliform and *E. coli* abundances were very similar to each other, but reflected a slightly different pattern than those observed for nutrients. Similar to nutrient variables, the greatest concentrations of bacteria followed major precipitation events and were significantly greater downstream following the inflow of the irrigation return flow streams. However, elevated concentrations were also observed in the upper reaches of the basin in 1999 (Fig. 6), reflecting the high intensity of grazing in these reaches.

The greatest concentrations were observed at the irrigation return flow site, LB4-2 in all three years. These concentrations were not significantly different from downstream sites, except for LB6 in 2000 and 2001, but were significantly different from sites upstream. Conversely, S1, which had high nutrient concentrations, had relatively low concentra-

![Fig. 6. Bar graph of *E. coli* concentrations with significant differences denoted by the letters above the bars.](image-url)
tions of bacteria, comparable to those found in the upper reaches of the basin (LB2 and LB3).

Fecal coliform abundances in LB4-2 were similar to those found by Greenlee et al. (2000) in the nearby Battersea Drain return flow stream. The other return flow stream, S1, had much lower fecal coliform abundances that were similar to the other three irrigation return flow streams in the study (Greenlee et al. 2000).

Coefficients of variation in the FLUX model were quite large; therefore, loading estimates were deemed to be unreliable. The relationship between bacteria concentration and flow is complicated by both bacterial life cycles and irrigation flows, which may not be related to upland contributions.

Land Use—Water Quality Linkages

In spite of our small sample size, several significant relationships between water quality and land use emerged from our analysis. Maximum NO$_3$-N (NO$_3$-N$_{\text{max}}$) and TN (TN$_{\text{max}}$) were positively correlated with several indicators of intensive agriculture, including CFO and AMU densities, the proportion of irrigated land, cereals, and potatoes (Table 2). Both variables were inversely correlated with the proportion of native grass and positively related to medium-textured, Orthic Dark Brown Chernozemic soils of glacio-lacustrine origin (OD_ME_LC) (Table 2). NO$_3$-N$_{\text{max}}$ was also related to the proportion of canola. Median TN (TN$_{\text{med}}$) values, which were observed under typical summer flow conditions, were correlated with population density and inversely correlated with the proportion of inclined slopes and native prairie. Most of the variation in TN$_{\text{med}}$ concentrations could be explained by the proportion of native prairie, whereas most of the variation in TN$_{\text{max}}$ concentrations was explained by the proportion of irrigated land (Table 3).

Unlike other studies in the sub-humid regions of North America (Johnson et al. 1997; Bouraoui et al. 1999), we did not find significant relationships between land use and NO$_3$-N$_{\text{med}}$ concentrations. Stronger correlations were found between land use and maximum concentrations of nutrients and other contaminants. These maxima were observed following precipitation events, which provided a transport mechanism between the land and the river. Groundwater can provide another pathway for NO$_3$-N to enter the river, but it was only a factor during extremely low flows observed during winter. Tile drains can also be a source of NO$_3$-N (Culley and Bolton 1983); however, in semi-arid Alberta, tile drainage is rare. The narrow range of NO$_3$-N$_{\text{med}}$ concentrations, likely due to denitrification, also contributed to the lack of relationships with land use variables.

Median concentrations of dissolved inorganic phosphorus (PO$_4$-P$_{\text{med}}$) and DP (DP$_{\text{med}}$), and maximum concentrations of TP (TP$_{\text{max}}$) were all positively correlated with the proportion of cereals and irrigated land (Table 2), while median TP (TP$_{\text{med}}$), DP$_{\text{med}}$, and TP$_{\text{max}}$ were inversely correlated with the proportion of native prairie. TP$_{\text{max}}$ also showed significant correlations with other crop types (potatoes), land uses (CFO densi-
Table 2. Selected Pearson correlation coefficients among water quality, soil, and land use variables

<table>
<thead>
<tr>
<th>Water quality variable</th>
<th>Pop. densa</th>
<th>CFOs</th>
<th>Cereal</th>
<th>Canola</th>
<th>Forage</th>
<th>Potato</th>
<th>Irrigated</th>
<th>Native</th>
<th>OD_ MC_FL</th>
<th>OD_ ME_LC</th>
<th>OD_ MF_TL</th>
<th>I3h</th>
</tr>
</thead>
<tbody>
<tr>
<td>NO₃-Nmed</td>
<td>0.332</td>
<td>-0.164</td>
<td>0.142</td>
<td>0.142</td>
<td>0.801</td>
<td>-0.261</td>
<td>-0.079</td>
<td>-0.744</td>
<td>-0.437</td>
<td>-0.015</td>
<td>0.613</td>
<td>-0.216</td>
</tr>
<tr>
<td>Log NO₃-Nmax</td>
<td>0.642</td>
<td><strong>0.931</strong></td>
<td><strong>0.937</strong></td>
<td><strong>0.937</strong></td>
<td>0.321</td>
<td><strong>0.856</strong></td>
<td><strong>0.925</strong></td>
<td>-0.780</td>
<td>-0.432</td>
<td>0.778</td>
<td>0.406</td>
<td>-0.612</td>
</tr>
<tr>
<td>TNmed</td>
<td><strong>0.822b</strong></td>
<td>0.540</td>
<td><strong>0.826</strong></td>
<td>0.086</td>
<td><strong>0.811</strong></td>
<td>0.484</td>
<td>0.694</td>
<td>-0.920</td>
<td>-0.477</td>
<td>0.695</td>
<td>0.553</td>
<td><strong>-0.756</strong></td>
</tr>
<tr>
<td>Log TNmax</td>
<td>0.734</td>
<td><strong>0.899</strong></td>
<td><strong>0.969</strong></td>
<td>0.108</td>
<td>0.374</td>
<td><strong>0.844</strong></td>
<td><strong>0.960</strong></td>
<td>-0.846</td>
<td>-0.443</td>
<td>0.848</td>
<td>0.365</td>
<td>-0.691</td>
</tr>
<tr>
<td>PO₄-Pmed</td>
<td>0.558</td>
<td>0.558</td>
<td><strong>0.892</strong></td>
<td>0.359</td>
<td>0.574</td>
<td>0.700</td>
<td><strong>0.827</strong></td>
<td>-0.745</td>
<td>-0.627</td>
<td>0.518</td>
<td><strong>0.775</strong></td>
<td>-0.400</td>
</tr>
<tr>
<td>Log PO₄-Pmax</td>
<td>-0.129</td>
<td>0.388</td>
<td>0.279</td>
<td>0.128</td>
<td>-0.045</td>
<td>0.398</td>
<td>0.275</td>
<td>-0.087</td>
<td>-0.600</td>
<td>-0.093</td>
<td>0.501</td>
<td>0.332</td>
</tr>
<tr>
<td>TPmed</td>
<td>0.667</td>
<td>0.314</td>
<td>0.622</td>
<td>0.052</td>
<td><strong>0.865</strong></td>
<td>0.250</td>
<td>0.455</td>
<td>-0.784</td>
<td>-0.486</td>
<td>0.477</td>
<td>0.618</td>
<td>-0.588</td>
</tr>
<tr>
<td>TPmax</td>
<td>0.741</td>
<td><strong>0.865</strong></td>
<td><strong>0.970</strong></td>
<td>0.108</td>
<td>0.478</td>
<td><strong>0.826</strong></td>
<td><strong>0.928</strong></td>
<td>-0.874</td>
<td>-0.510</td>
<td><strong>0.798</strong></td>
<td>0.475</td>
<td><strong>-0.673</strong></td>
</tr>
<tr>
<td>DPmed</td>
<td>0.689</td>
<td>0.737</td>
<td><strong>0.921</strong></td>
<td>0.217</td>
<td>0.655</td>
<td>0.700</td>
<td><strong>0.842</strong></td>
<td>-0.858</td>
<td>-0.553</td>
<td>0.687</td>
<td>0.675</td>
<td>-0.602</td>
</tr>
<tr>
<td>DPmax</td>
<td>0.024</td>
<td>-0.145</td>
<td>0.111</td>
<td>0.480</td>
<td>0.337</td>
<td>0.197</td>
<td>0.063</td>
<td>-0.187</td>
<td>-0.754</td>
<td>-0.115</td>
<td>0.517</td>
<td>0.297</td>
</tr>
<tr>
<td>F. Colimed</td>
<td>0.516</td>
<td>0.054</td>
<td>0.356</td>
<td>0.131</td>
<td>0.706</td>
<td>0.065</td>
<td>0.251</td>
<td>-0.541</td>
<td>-0.103</td>
<td>0.424</td>
<td>0.329</td>
<td>-0.554</td>
</tr>
<tr>
<td>Log F. Colimax</td>
<td>0.202</td>
<td>0.493</td>
<td>0.154</td>
<td>0.075</td>
<td>0.637</td>
<td>0.451</td>
<td>-0.245</td>
<td>0.216</td>
<td>-0.336</td>
<td>0.180</td>
<td>0.270</td>
<td>-0.127</td>
</tr>
<tr>
<td>E. coli₅med</td>
<td>0.351</td>
<td>-0.154</td>
<td>0.166</td>
<td>0.171</td>
<td>0.687</td>
<td>-0.155</td>
<td>0.506</td>
<td>-0.384</td>
<td>-0.125</td>
<td>0.298</td>
<td>0.356</td>
<td>-0.067</td>
</tr>
<tr>
<td>Log E. coli₅max</td>
<td>0.124</td>
<td>-0.344</td>
<td>-0.061</td>
<td>0.118</td>
<td>0.536</td>
<td>-0.192</td>
<td>-0.181</td>
<td>-0.204</td>
<td>-0.365</td>
<td>-0.059</td>
<td>0.336</td>
<td>-0.004</td>
</tr>
</tbody>
</table>

a All variables are relative measures.

b Bold correlation coefficients denote significance (p < 0.05).
ty) and soil types (OD_ME_LC). In contrast with previous studies, significant relationships were observed between phosphorus variables and land use. The relationship between $\text{TP}_{\text{max}}$ and the proportion of cereals was consistent with studies from the U.S. that link high export values of particulate phosphorus with croplands (Sharpley et al. 1999); however, in this study, $\text{DP}_{\text{med}}$ was also associated with cereals and irrigated land. This may be due to increased soil-water contact time on irrigated soils, which can increase runoff concentrations of DP (Sharpley et al. 1999). Additionally, other Alberta studies have suggested that DP concentrations in runoff may be higher than in other regions (Wright et al. 2002).

$\text{DP}_{\text{max}}$ was inversely correlated with the proportion of medium- to coarse-textured Orthic Dark Brown Chernozemic soils of glacio-fluvial origin, which were not found in the two return flow sub-basins; however, it showed no relationship with other land use variables. The lack of relationship between DP and land use may be due, in part, to the variable responses observed following the precipitation events. For example, there was very little response at LB3 following the June 3, 1999, event (17 µg L$^{-1}$ at LB3 versus 44 µg L$^{-1}$ at LB2), and the opposite response following the August 1999 event, despite similarities in land use in these two sub-basins. In the latter event, DP accounted for most of the TP in most mainstem sites (LB3-LB6), possibly due to greater crop cover that reduced losses of particulate phosphorus (Wendt and Corey 1980) or due to differences in intensity (Kunishi et al. 1972) and duration of the event (Pierson et al. 2001) that promoted desorption of phosphorus from the sediments.

Of the phosphorus variables, only $\text{TP}_{\text{max}}$ was correlated with livestock variables (CFOs and AMUs), suggesting that these land uses contributed a relatively larger proportion of organic and/or particulate P compared with dissolved and inorganic forms during storm events. Higher particulate phosphorus concentrations are not surprising due to the larger volumes of sediment transported during storm events. Organic phosphorus forms comprise a large fraction of the phosphorus in soil solution of manure-amended soils (Chardon et al. 1997); however, no studies have examined their contribution to surface runoff. Due to design

### Table 3. Regression models, coefficients and probabilities for selected water quality variables

<table>
<thead>
<tr>
<th>Variable</th>
<th>Model</th>
<th>$R^2_{\text{adj}}$</th>
<th>Significance level</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\text{TN}_{\text{med}}$</td>
<td>$-3.86 \times (\text{Native}) + 518.83$</td>
<td>0.847</td>
<td>0.003</td>
</tr>
<tr>
<td>$\text{TN}_{\text{max}}$</td>
<td>$189.45 \times (\text{Irrigated}) + -921.39$</td>
<td>0.829</td>
<td>0.004</td>
</tr>
<tr>
<td>$\text{DP}_{\text{med}}$</td>
<td>$0.212 \times (\text{Cereal}) + 7.65$</td>
<td>0.848</td>
<td>0.003</td>
</tr>
<tr>
<td>$\text{TP}_{\text{med}}$</td>
<td>$25.63 \times (\text{Forage}) + 16.0$</td>
<td>0.748</td>
<td>0.012</td>
</tr>
<tr>
<td>$\text{TP}_{\text{max}}$</td>
<td>$16.73 \times (\text{Cereal}) + -75.30$</td>
<td>0.930</td>
<td>&lt;0.001</td>
</tr>
</tbody>
</table>
criteria and siting restrictions, most phosphorus would likely be contributed via land application of manure and not from the operations themselves, as none of the observed events were extreme. McFarland and Hauck (1999) reported significant relationships between phosphorus concentrations during storm events and fields with applied dairy waste.

Most of the variation in DPmed and TPmax concentrations was explained by the proportion of cereals in the sub-basin (Table 3). This land use was dominant in many of the sub-basins, but particularly in the irrigation return flow basins where there were V-shaped channels without floodplains. The proportion of forages in the sub-basin best explained TPmed concentration (Table 3), although this land use was relatively minor in the basin (Table 1).

Fecal coliforms and \textit{E. coli} exhibited no significant correlations with land use or soil classes. Previous studies have shown that distance to water and grazing intensity were important factors in explaining \textit{E. coli} concentrations in water (Tian et al. 2002); however, neither of these factors was measured in our study. Although there have been many reports of coliform bacteria survival of over a year in fecal matter (Marsh and Campling 1970; Buckhouse and Gifford 1976), most bacteria have high mortality rates outside of the host due to exposure to unfavorable water temperature (Thelin and Gifford 1983) or poor soil moisture (Guan and Holley 2003) and UV radiation (Davies-Colley et al. 1999; Thelin and Gifford 1983) on land. Survival can be facilitated in beds and banks of rivers that act as reservoirs for bacteria during low flows and sources during high flow events (Kunkle 1970 in Thelin and Gifford 1983). The complexity of these temporal and spatial factors makes it very difficult to relate indicator bacteria to land use.

**Conclusions**

Although flows relative to the mainstem were small, the two largest return flows had significant impacts on the Lower Little Bow River during low flow years, increasing concentrations of TN, TP, and DP within the mainstem. The proportion of irrigated area in each of the sub-basins was also correlated with many of the water quality variables and explained most of the variation in TNmax. Strong positive relationships were also observed between the proportion of cereals and phosphorus concentrations (DPmed, TPmax) and inverse relationships were observed between native prairie and TNmed values.

The impact of CFOs is a concern in the Lower Little Bow basin. Maximum TP, TN, and NO3-N were all associated with CFOs and manure production, likely through runoff from land application of manure. However, median values and microbiological parameters were not correlated with this land use. In general, maximum water chemistry values had stronger correlations with soil type and land use classes than median values. Nearly all of the maximum values were recorded following a single major precipitation event. This observation illustrates the need for a transport mechanism, namely surface and subsurface runoff, to link...
upland management and in-stream quality. During the study period (1999–2001), there was one year of near-normal precipitation and two years of below average precipitation. As long as flows are not substantially reduced, water quality tends to be slightly better in low-runoff years due to smaller non-point source contributions.

Total nitrogen and NO₃-N did not exceed water quality guidelines and had a smaller range of values; therefore, they generally had weaker relationships with land use variables. Phosphorus variables showed more correlations with land use, which is in contrast with previous studies, which have found that soil interactions can play a large role in phosphorus dynamics and may hinder land use-water quality associations. Although there were some relationships between soil classes and phosphorus, these were likely due to the location of intensive agricultural activities rather than the soil type itself.

Bacteria had only localized effects and were not related to irrigation or other general land use activities. There were some high incidences of bacteria in catchments on native range. Although it may have strengthened the relationships, numbers of rangeland cattle or stocking densities could not be reliably estimated due to the variability in grazing practices (i.e., rotational grazing, variable cattle numbers, wintering sites).

The limited relationships observed between land use and bacteria may be due both to the relatively broad scale at which land use activities were measured and the natural life cycles of bacteria. Fecal coliforms and E. coli show more dynamic patterns and greater variation than inorganic water chemistry variables because they have limited reproduction and high mortality outside of their host organisms. Therefore, bacteria would most likely be contributed from riparian areas or point sources discharging to the rivers, whereas a greater proportion of phosphorus and nitrogen could be contributed from upland sources.

Acknowledgements

We wish to thank T. Entz for statistical advice, D. Mikalson for processing FLUX data, and the staff of AAFRD Irrigation Branch for collecting water samples. This study was conducted as part of the Oldman River Basin Water Quality Initiative.

References


U. S. Army Corps of Engineers. 1999. FLUX v. 5.1: stream load calculations. Environmental Laboratory, USAE Waterways Experimental Station, Vicksburg, Mississippi.

Received: April 7, 2003; accepted: August 28, 2003.