Decadal and seasonal water quality trends downstream of urban and rural areas in southern Alberta rivers
B. Turnbull and M. C. Ryan

ABSTRACT

Seasonal and long-term water quality changes downstream of urban and agricultural land uses were compared using chloride, fecal coliform (FC), Escherichia coli and discharge from five long-term river monitoring sites in the Bow and Oldman watersheds in southern Alberta. Water quality data from up- and downstream locations of two major urban centers (Calgary and Lethbridge) and single sampling locations downstream of three agricultural sites were evaluated. Significant monotonic, decadal increases in chloride mass fluxes observed downstream of both urban areas were consistent with increasing chloride fluxes in wastewater effluent from increasing populations. Significant step function decreases in FC concentrations downstream of the two urban centers (89% at Calgary, 70% at Lethbridge) observed after UV disinfection were introduced at upstream wastewater treatment plants, suggesting wastewater disinfection improved river water quality. Significant monotonic decreases in pathogen indicators were found at only one of the three agricultural sampling locations. Seasonal variations in indicator bacteria were consistent with a constant source at the urban downstream sites, while variable seasonal loading patterns at the agricultural sites were attributed to seasonally changing land use. This suggests that the urban centers are more significantly mitigating pathogens in rivers than rural areas despite their significant growth.

Key words | agriculture, pathogen indicator, river, wastewater, water quality

INTRODUCTION

Ensuring clean water for downstream users is a major pressure placed on urban centers and agricultural operations. This is particularly true in water-scarce areas where imbalances between availability and demand, water quality degradation, competing water uses, inter-regional, and international conflicts often occur (Pereira et al. 2002). Sustainability of scarce water resources has been improved by the significant enhancements of urban wastewater treatment practices over the past decades through the implementation of new technologies to remove nutrients, the incorporation of UV disinfection techniques (which typically decreases pathogen concentrations in effluent by more than four orders of magnitude; Metcalf & Eddy Inc. 2003), and increasing regulation of urban wastewater effluent quality (Melosi 2000). Non-point or diffuse impacts from agricultural activity are not as transparently regulated or managed. Although significant efforts have been made towards ‘best’ or ‘beneficial’ management practices (BMPs) in agricultural areas, it is often unclear how extensive changes in practices have been, and whether there has been a consequent improvement in river water quality (Logan 1993; Gitau et al. 2005; Sharpley et al. 2009).

Pathogens are of primary concern with respect to drinking water quality since waterborne disease can result from the consumption of inadequately treated water, from irrigation, and/or from recreational contact (Arnone & Walling 2007; Hrudey & Hrudey 2007). Parameters of interest with respect to water quality in this study include fecal coliforms (FC) and Escherichia coli, which are used as indicators of fecal contamination in water since they can be positively correlated with the presence of pathogenic enteric bacteria (Hyland et al. 2003). The Canadian Recreational Water

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Quality guideline for *E. coli* is 200 cfu 100 mL\(^{-1}\) (Health and Welfare Canada 1992), although pathogens can be associated with lower indicator bacteria concentrations (Edge et al. 2012). Although FC was the more commonly used pathogen indicator historically (and is still acceptable when it forms a high fraction of the *E. coli* group; Health Canada 2010), most agencies now use *E. coli*. The Guidelines for Canadian Drinking Water Quality objective for *E. coli* and FC is none detectable per 100 mL.

Chloride is a conservative solute, little affected by biologic river processes or chemical reactions, that serves as a useful tracer of treated wastewater effluent (Iwanyshyn et al. 2008) and manure (Rodvang et al. 2004). Increased chloride concentrations in rivers, especially near urban areas, are often attributed to treated wastewater effluent (Vanderberg et al. 2005) and road salting in temperate areas (Interlandi & Crockett 2005; Mullaney et al. 2009). River chloride concentrations are not usually present at or above surface water guidelines for the protection of aquatic life (120 and 640 mg L\(^{-1}\) for short- and long-term exposure, respectively; CCME 2011), however. River chloride mass fluxes from a constant source (like wastewater treatment effluent) would be expected to vary inversely with river discharge (Kim et al. 2002; Iwanyshyn et al. 2008), as do geologic sources (Freeze & Cherry 1979). Non-constant sources of river-borne pathogen indicators (i.e., due to seasonal changes in land use) would not necessarily be related to discharge.

Urban and agricultural land uses can both impact downstream water quality (Cessna et al. 2001, Brezonik & Stadelmann 2002; Hyland et al. 2005; Byrne et al. 2006; Mul laney et al. 2009). Agricultural influences on bacteriological water quality are typically related to manure application or feedlot runoff, which when combined with overland flow from heavy precipitation events or return flow from irrigated areas, can adversely affect river water quality (Cessna et al. 2001; Little et al. 2003; Edge et al. 2012). Seasonal and event-based variability in river water quality can make it particularly difficult to discern agricultural management-induced water quality changes, however, even in multi-year research studies (Miller et al. 2010, 2011).

Urban wastewater effluent is also an important cause of surface water quality degradation (Neal et al. 2000; Sosiak 2002). Although wastewater treatment has been critical in the reduction of the impact of sewage effluent on rivers, improved treatment is countered by continually increasing urban populations and effluent volumes (Carey & Miglia cio 2009).

Southern Alberta’s semi-arid climate makes its population and industry highly reliant on rivers that originate in the eastern slopes of the Rocky Mountains, and water scarcity has resulted in a moratorium on new surface water licenses (AENV 2006). The highest concentrations of *E. coli* and FC and nutrients in southern Alberta are typically observed in high flow periods in the summer months, particularly following major precipitation events (Hyland et al. 2003; Little et al. 2005) suggesting they may be related to land use activities.

This paper evaluates historic water quality data for rivers up- and downstream of two urban centers, and downstream of three intensive agricultural areas. Seasonal concentration patterns and relationships with river discharge and decadal scale changes in mass fluxes were evaluated to understand the nature of the relative sources, and their relative changes with time. Mass fluxes (the product of concentration and discharge) are used to consider the total load of chloride and/or pathogen indicators in order to remove any influence of changing river discharge on the decadal scale (Hirsch et al. 1991; Valeo et al. 2007). Chloride mass fluxes were evaluated for monotonic trends to evaluate whether increased wastewater effluent chloride from increasing urban populations was evident in the data. Since the implementation of wastewater disinfection at wastewater treatment plants (WWTPs) was expected to result in significant decreases in *E. coli* and FC concentrations in WWTP effluent, the data were evaluated for a step function change ‘before’ and ‘after’ the implementation of WWTP disinfection. Water quality data at agricultural sites were evaluated in a similar manner.

**STUDY SITES, WATER QUALITY, AND DISCHARGE DATA**

The Oldman and Bow Rivers are major tributaries of the Saskatchewan–Nelson River system. They originate in the Rocky Mountains of southwestern Alberta, Canada (Sosiak 2002) and join to form the South Saskatchewan River upstream of
Medicine Hat (Figure 1). Both rivers are significant in size and flow (Table 1), and have similar annual hydrographs, with peak flow occurring in mid-June. The land use in both watersheds changes from alpine to foothills (with mixed and non-intensive land use) in the upper reaches, with irrigation and increasing (but variable) agricultural intensity in the lower, prairie reaches. Each watershed has a major urban center with significant municipal wastewater effluent discharge. The region has relatively low annual precipitation rates (with a net evapotranspiration deficit), and a burgeoning animal agriculture industry (Hyland et al. 2005).

Water quality and discharge data were collated for sites where an Environment Canada water gauging station and Alberta Environment (AENV) long-term water quality monitoring occurred in reasonably close proximity (Figure 1; Table 2). River discharge was obtained from Environment Canada, and chloride, E. coli, and FC data were obtained from Alberta Environment and the City of Calgary (for one water quality sampling site only) from approximately 1970 to 2008 for five sites. One urban site on each river (1U and 3U) had paired sampling locations: one was located upstream of each city, and a second was located downstream of the WWTP outfalls (the downstream sampling location was selected far enough away from the effluent outfall to ensure full effluent mixing across the river; Vanderberg et al. 2005). Three agricultural sites consist of single sampling locations downstream of predominantly agricultural areas. These included agricultural sites on each of the two rivers further downstream from the urban centers (2A and 4A). A fifth site (5A), located on a minor tributary of the Bow River, was chosen since it had extensive water quality data, is a third agricultural site.

Water quality data at the downstream location for one urban site (Site 1U; Figure 1) were collated from AENV and City of Calgary sampling programs since samples were taken within 1 km of one another (Table 2). Site 4A did not have a nearby gauging station so flow was interpolated between Site 3U and a gauging station near the Oldman River mouth (marked as a ‘discharge only’ location on Figure 1). A consistent relationship was observed between

![Figure 1](image-url)  
**Figure 1** | Geographic locations of paired urban sites (grey filled symbols, ‘U’ = upstream and ‘D’ = downstream) and agricultural (black filled symbols, ‘A’) sites along the Bow and Oldman Rivers and Crowfoot Creek. More details on long-term water quality and river discharge monitoring locations in southern Alberta are found in Table 2.

<table>
<thead>
<tr>
<th>Table 1</th>
<th>Watershed area, flow length, average annual discharge, and the population of major urban centers in the three rivers evaluated</th>
</tr>
</thead>
<tbody>
<tr>
<td>River</td>
<td>Watershed area (km²)</td>
</tr>
<tr>
<td>Bow R.</td>
<td>26,200</td>
</tr>
<tr>
<td>Oldman R.</td>
<td>26,700</td>
</tr>
<tr>
<td>Crowfoot Cr.</td>
<td>1,600</td>
</tr>
</tbody>
</table>

Population data are from Alberta Municipal Affairs.
the two gauging stations, and flow was approximated by relative distance to each gauging station. Crowfoot Creek data were obtained from a single site on the tributary near Cluny where discharge data were not available.

Wastewater treatment plant upgrades to include UV disinfection (Metcalf & Eddy Inc. 2003) in Calgary occurred in 1994 and 1997 (for their larger and smaller WWTPs, respectively; Sosiak 2002) and in 1999 in Lethbridge (Hyland et al. 2003; Table 2). There is no significant municipal effluent discharge to Crowfoot Creek.

Contract laboratories conducted all AENV chloride analyses. Prior to 1980, chloride analyses for the AENV sampling program were conducted using nitrate reduction by mercuric thiocyanate and ferric ammonium sulfate and autoanalyser colorimetry (with a detection limit of 0.5 mg/L). Subsequent to 1980, chloride analyses were conducted by ion chromatography (APHA 2005; detection limit 0.01 mg/L). Fecal coliform and E. coli analyses in the AENV sampling programs were conducted at Alberta’s Provincial Laboratory using membrane filtration (APHA 2005). Chloride analyses at the City of Calgary were conducted by ion chromatography (APHA 2005; detection limit of 0.01 mg/L), and E. coli by Colilert® Quantitray method (APHA 2005). Chloride concentrations were very seldom below the detection limit in any of the data, and the detection limit for E. coli and FC (<1 cfu 100 mL⁻¹) was constant over the period of time considered.

### DATA ANALYSIS

Changes in the estimated WWTP effluent-chloride mass fluxes were compared to water quality data to confirm that increasing population was reflected in river water quality. Food is a significant source of salt (and chloride) in wastewater, with typical effluent concentrations between 80 and 120 mg/L (Metcalf & Eddy Inc. 2003). Calgary’s treated wastewater effluent, which has chloride concentrations in this range, adds approximately 40 tons Cl day⁻¹ of chloride to the Bow River (Iwanyshyn et al. 2008). These data were used to estimate a per capita chloride mass flux in WWTP effluent of about 44 mg person⁻¹ day⁻¹. The predicted chloride mass flux from treated wastewater effluent for a given urban center was then estimated as the product of the average annual population near the sampling site (i.e., Calgary for the Bow River and Lethbridge for the Oldman River) by the per capita chloride mass flux (which was assumed to remain constant with time).

Although the two coliform groups, E. coli and FC, were reasonably well correlated (data not shown), the parameters were statistically analyzed independently since they represent different subsets of the coliform group. Chloride and pathogen indicator concentrations at each site were cross-plotted with daily average flow and Julian day (to look for seasonal patterns) for each site except for Crowfoot Creek (which had insufficient chloride concentration data).

### Table 2

<table>
<thead>
<tr>
<th>Site</th>
<th>Location</th>
<th>Major land use</th>
<th>Sampling location names</th>
<th>Gauging locations</th>
<th>Dates WWTP disinfection implemented</th>
</tr>
</thead>
<tbody>
<tr>
<td>1U</td>
<td>Bow R</td>
<td>Urban</td>
<td>Upstream (Cochrane); Downstream (Stier’s Rancha/Policeman’s Flats) of Calgary</td>
<td>Calgary</td>
<td>1994, 1997</td>
</tr>
<tr>
<td>2A</td>
<td>Bow R</td>
<td>Agricultural</td>
<td>Near Ronalanea</td>
<td>Near Ronalane</td>
<td>n.a.c</td>
</tr>
<tr>
<td>3U</td>
<td>Oldman R</td>
<td>Urban</td>
<td>Discharge upstream of Lethbridgea</td>
<td>Lethbridge</td>
<td>1999</td>
</tr>
<tr>
<td>4A</td>
<td>Oldman R</td>
<td>Agricultural</td>
<td>Near Tabera</td>
<td>Interpolated between 3U and station near mouth</td>
<td>n.a.</td>
</tr>
<tr>
<td>5A</td>
<td>Crowfoot Cr</td>
<td>Agricultural</td>
<td>Near Clunya</td>
<td>Not available</td>
<td>n.a.</td>
</tr>
</tbody>
</table>

*aAlberta Environment historic water quality sampling location.
*bCity of Calgary historic water quality sampling location (2004–2007 data only).
*c n.a. = not applicable.
Mass fluxes for each parameter were estimated by multiplying parameter concentration by average daily flow on the sampling date.

The water quality data were analyzed for seasonal patterns by plotting all data as a function of Julian day, and for differences in mass fluxes over time. Parametric assumptions of normality and equal variance were tested for each of these parameters using the Anderson–Darling test and Levene’s test, respectively. Non-normal data were log transformed to correct violations. A log-10 transformation corrected for deviations from normality and homoscedasticity in a few cases. These data were then subjected to a one-way analysis of variance (ANOVA) and Tukey pairwise comparison between months. For the rest of the data sets where parametric assumptions could not be met, a non-parametric Kruskal–Wallis test was performed followed by a post-hoc Kruskal–Wallis pair-wise comparison.

Two types of statistical analyses were used to evaluate whether changes in mass fluxes with time were significant. Chloride, FC, and E. coli mass fluxes were analyzed for statistically significant monotonic changes using a Kendall-tau test with 95% significance. Sen slopes were calculated to provide an estimate of the approximate magnitude of significant monotonic trends (Sosiak 2002). Changes in mass fluxes for each parameter were estimated by multiplying parameter concentration by average daily flow on the sampling date.

### Table 3

<table>
<thead>
<tr>
<th>Site</th>
<th>Mean monthly chloride concentrations (mg L(^{-1})) standard error</th>
<th>(mg L(^{-1})) standard error</th>
</tr>
</thead>
<tbody>
<tr>
<td>1U (upstream)</td>
<td>1.92 (0.11)</td>
<td>1.52 (0.16)</td>
</tr>
<tr>
<td>1U (downstream)</td>
<td>7.48 (0.51)</td>
<td>4.73 (0.39)</td>
</tr>
<tr>
<td>2A</td>
<td>10.74 (0.61)</td>
<td>4.11 (0.38)</td>
</tr>
<tr>
<td>3U (upstream)</td>
<td>2.01 (0.15)</td>
<td>1.41 (0.14)</td>
</tr>
<tr>
<td>3U (downstream)</td>
<td>5.85 (0.35)</td>
<td>4.62 (0.35)</td>
</tr>
<tr>
<td>4A</td>
<td>6.76 (0.82)</td>
<td>3.88 (0.85)</td>
</tr>
</tbody>
</table>

Site 5A is not included since there were insufficient chloride data.

Concentrations with different superscript letter groupings denote significantly different monthly concentrations (\(P < 0.05\)) at each sampling site (i.e., within each row).

### Table 4

<table>
<thead>
<tr>
<th>Site</th>
<th>1U (downstream location)</th>
<th>2A</th>
<th>3U (downstream location)</th>
<th>4A</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chloride</td>
<td>0.051 (0.0089)(^a)</td>
<td>0.331 (0.0024)(^a)</td>
<td>0.156 (0.00024)(^a)</td>
<td>Not sign.(^a)</td>
</tr>
<tr>
<td>E. coli</td>
<td>Not sign.(^a)</td>
<td>Not sign.(^a)(^b)</td>
<td>Not sign(^a)</td>
<td>−0.11 (−9640)(^a)</td>
</tr>
<tr>
<td></td>
<td>−382(^b)</td>
<td>−1,706(^b)</td>
<td>−7,466(^b)</td>
<td></td>
</tr>
<tr>
<td>Fecal coliforms</td>
<td>Not sign.(^a)</td>
<td>Not sign.(^a)(^b)</td>
<td>Not sign(^a)</td>
<td>−0.131 (−12,062)(^a)</td>
</tr>
<tr>
<td></td>
<td>−10,873(^b)</td>
<td>−2,461(^b)</td>
<td>−2,3478(^b)</td>
<td></td>
</tr>
</tbody>
</table>

A negative value indicates a decrease in mass flux, whereas a positive value indicates an increase in mass flux. Note that trends in water quality at the upstream locations for Sites 1U and 3U were not tested since no visual trends were evident. Site 5A is not included due to a relatively short data record, and the absence of significant WWTP effluent discharge.

\(^a\)Kendall-tau test (Sen’s slope) to evaluate monotonic change.

\(^b\)Mann–Whitney test to evaluate step-function change.
fluxes of chloride, E. coli, and FC were also evaluated for step changes associated with WWTP disinfection implementation at the urban sites (1U and 3U) and agricultural sites with significant WWTP discharge upstream (2A and 4A; Table 2). The pre- and post-disinfection values were compared using a Mann–Whitney test and analyzed for 95% significance. In each case, a similar number of years to that available after the upgrades were used before the upgrade(s) in the comparison (Hirsch et al. 1999). Data before 1994 were compared to data after 1997 for the Bow River sites, and data before and after 1999 for the Oldman River sites (Table 2). The same periods were used to evaluate chloride mass fluxes, which (given chloride's conservative nature) should be unaffected by disinfection or other WWTP upgrades.

To maintain consistency, changes over similar time periods were evaluated for Crowfoot Creek even though no WWTP upgrades occurred upstream. Outliers were removed only if the data could be explained by flooding or major precipitation events.

RESULTS AND DISCUSSION

Chloride

Chloride concentrations in the rivers upstream of large urban areas were low (typically <2 mg L⁻¹; Table 3) and mass flux showed little seasonal variation (Table 4). All of the downstream sites exhibited a similar (and expected)

![Figure 2](https://iwaponline.com/wqrj/article-pdf/47/3-4/406/163549/406.pdf)

**Figure 2** | Chloride concentration (mg L⁻¹) vs. daily average discharge (m³ s⁻¹). Site 5A is not included due to insufficient chloride data.
inverse relationship between chloride concentration and daily average discharge (Hem 1985; Figure 2). In general, the highest chloride concentrations were observed during periods of low flow while the lowest concentrations were observed at high flow, consistent with earlier work (Grasby et al. 1999). This relationship was best exemplified on the Bow River site downstream of Calgary where the highest chloride concentrations (>20 mg L\(^{-1}\)) were only present at low flows (<70 m\(^3\) s\(^{-1}\)) and low chloride concentrations (<5 mg L\(^{-1}\)) at the highest flows (>250 m\(^3\) s\(^{-1}\)). Lower chloride concentrations under high flow conditions suggest point sources contribute steady and significant chloride mass fluxes that are diluted by high flow runoff during storm events (Neal et al. 2000).

The lowest chloride concentrations were observed at all sites during the seasonally high flow periods (Figure 2) in June and July (Table 3), with at least some seasonality in chloride concentrations observed at all sites (Figure 3). Infrequently high chloride concentrations (>20 mg L\(^{-1}\)) were observed in the late winter–early spring period on the Bow River at Calgary. These peaks may be attributed to significant melt or precipitation events that cause rapid transport of road salt to the river via stormwater. Road density, potential evapotranspiration, and the percentage of annual runoff from saturated overland flow can all contribute to sporadic, high chloride concentrations (Mullaney et al. 2009).

Measured chloride mass fluxes, and the estimated mass flux due to municipal sewage, showed a statistically significant increasing trend with time (Figure 4; Table 4), which is commonly observed in urban basins for which long-term data are available (Mullaney et al. 2009). The predicted chloride mass flux from wastewater (indicated with a solid line in Figure 4) tends to track the lowest values observed at both Sites 1U (Bow River) and 3U (Oldman River), with
some variability above these low values (Figure 4). This suggests that although wastewater effluent is a major source of chloride, other significant sources exist (e.g., road salts; Interlandi & Crockett 2003; Thunqvist 2004; Mulaney et al. 2009; Novotny et al. 2009). The predicted wastewater chloride mass flux in the Oldman at Site 4A similarly tracks the lowest values observed. In contrast, significantly lower chloride mass fluxes were often observed relative to those predicted in the Bow River at Site 2A (Figure 4). Given the lack of any major tributaries and the conservative nature of chloride, these data are puzzling. They may be associated with sporadically low-chloride concentration irrigation return flow (diverted from upstream of Calgary’s WWTPs) and/or groundwater–surface water interaction. Regardless, chloride concentrations at the Bow River at Site 2A station show a monotonic increase consistent with increasing wastewater chloride loading. The relatively low chloride mass flux in the Oldman River relative to the Bow River is a reflection of the lower upstream population sizes (Table 1).

E. coli and fecal coliform

Since both E. coli and FC concentrations were consistently low (<2 cfu 100 L⁻¹; data not shown) at the upstream locations of the urban sites (1U and 3U) they are not considered further here. The bacterial concentrations at the other locations were variable, ranging from <1 to more than 1,000 cfu 100 mL⁻¹ at all sites except 2A (where a maximum concentration >650 cfu 100 mL⁻¹ was observed (Figure 5). Some variation of E. coli and FC concentration with flow is observed at the downstream locations of the urban sites (1U and 3U) and agricultural sites (Figure 5), although the trends are not as clear as those observed for chloride. These relationships were analyzed for significance using simple linear regression. The rural sites on both the
Figure 5 | E. coli (○) and fecal coliform (▲) counts (cfu 100 mL⁻¹) vs. daily average discharge (m³ s⁻¹) at five sites.
Figure 6  |  E.coli (o) and fecal coliform (x) mass flux (cfu s⁻¹) vs. Julian day at all five sites.
### Table 5 | Mean monthly *E. coli* (cfu 100 mL\(^{-1}\) ± standard error) concentrations at the five sampling sites (located in Figure 1)

<table>
<thead>
<tr>
<th>Site</th>
<th>Jan</th>
<th>Feb</th>
<th>Mar</th>
<th>Apr</th>
<th>May</th>
<th>Jun</th>
<th>Jul</th>
<th>Aug</th>
<th>Sep</th>
<th>Oct</th>
<th>Nov</th>
<th>Dec</th>
</tr>
</thead>
<tbody>
<tr>
<td>1U (downstr)</td>
<td>101.6(^{A\B}) (25.7)</td>
<td>258.0(^{A\C}) (122)</td>
<td>92.6(^{A\B}) (35.0)</td>
<td>44.7(^{A}) (15.7)</td>
<td>141.3(^{A\B\C}) (41.8)</td>
<td>313.0(^{A\B\C}) (130)</td>
<td>93.8(^{A\B}) (18.5)</td>
<td>522.8(^{A\B\C}) (72.6)</td>
<td>402.7(^{A\B\C}) (88.0)</td>
<td>81.6(^{A}) (28.9)</td>
<td>50.3(^{A}) (20.5)</td>
<td>87.5(^{A\B\C}) (17.5)</td>
</tr>
<tr>
<td>2A</td>
<td>4.2(^{A}) (1.42)</td>
<td>4.6(^{A}) (1.88)</td>
<td>3.4(^{A}) (1.02)</td>
<td>8.46(^{A\B}) (3.69)</td>
<td>76.8(^{A\C\D\E\F\G\H\I\J\K\L\M\N\O\P\Q\R\S\T\U\V\W\X\Y\Z}) (25.3)</td>
<td>32.1(^{A\B\C\D\E\F\G\H\I\J\K\L\M\N\O\P\Q\R\S\T\U\V\W\X\Y\Z}) (9.81)</td>
<td>35.1(^{A\B\C\D\E\F\G\H\I\J\K\L\M\N\O\P\Q\R\S\T\U\V\W\X\Y\Z}) (10.1)</td>
<td>20.3(^{A\B\C\D\E\F\G\H\I\J\K\L\M\N\O\P\Q\R\S\T\U\V\W\X\Y\Z}) (5.24)</td>
<td>14.4(^{A\B\C\D\E\F\G\H\I\J\K\L\M\N\O\P\Q\R\S\T\U\V\W\X\Y\Z}) (5.97)</td>
<td>5.4(^{A}) (1.68)</td>
<td>(0.45)</td>
<td></td>
</tr>
<tr>
<td>3U (downstr)</td>
<td>73.2(^{A\B\C}) (65.4)</td>
<td>75.5(^{A\B\C}) (25.7)</td>
<td>17.0(^{A\B\C}) (11.8)</td>
<td>1.00(^{A}) (0.00)</td>
<td>145(^{A\B\C}) (134)</td>
<td>450(^{A\B\C}) (250)</td>
<td>111(^{A\B\C}) (82.3)</td>
<td>65.3(^{A\B\C}) (23.7)</td>
<td>16.0(^{A\B\C}) (6.24)</td>
<td>20.3(^{A\B\C}) (2.35)</td>
<td>14.4(^{A\B\C}) (14.4)</td>
<td></td>
</tr>
<tr>
<td>4A</td>
<td>13.5(^{A}) (5.32)</td>
<td>10.4(^{A}) (4.86)</td>
<td>33.8(^{A}) (29.8)</td>
<td>7.29(^{A}) (3.67)</td>
<td>332.0(^{A\B\C}) (182)</td>
<td>127.8(^{A\B\C}) (40.4)</td>
<td>127.8(^{A\B\C}) (15.6)</td>
<td>60.4(^{A\B\C}) (13.7)</td>
<td>36.8(^{A\B\C}) (7.33)</td>
<td>24.2(^{A\B\C}) (2.75)</td>
<td>9.38(^{A\B\C}) (54.7)</td>
<td></td>
</tr>
<tr>
<td>5A</td>
<td>56.6(^{A\B\C}) (20.9)</td>
<td>n/a</td>
<td>169.0(^{A\B\C}) (100)</td>
<td>192.6(^{A\B\C}) (39.9)</td>
<td>316.0(^{A\B\C}) (89.5)</td>
<td>134.3(^{A\B\C}) (18.3)</td>
<td>129.1(^{A\B\C}) (52.3)</td>
<td>38.7(^{A\B\C}) (33.1)</td>
<td>n/a</td>
<td>n/a</td>
<td>n/a</td>
<td></td>
</tr>
</tbody>
</table>

Concentrations with different superscript letter groupings denote significantly different monthly concentrations (\(P < 0.05\)) at each sampling site (i.e., within each row).

Data were not available (n/a) for some months for Site 5A.

### Table 6 | Mean monthly fecal coliform (cfu 100 mL\(^{-1}\) ± standard error) concentrations at the five sampling sites (located in Figure 1)

<table>
<thead>
<tr>
<th>Site</th>
<th>Jan</th>
<th>Feb</th>
<th>Mar</th>
<th>Apr</th>
<th>May</th>
<th>Jun</th>
<th>Jul</th>
<th>Aug</th>
<th>Sep</th>
<th>Oct</th>
<th>Nov</th>
<th>Dec</th>
</tr>
</thead>
<tbody>
<tr>
<td>1U downstr</td>
<td>983.0(^{B\C}) (±429)</td>
<td>765.0(^{A\B\C}) (±221)</td>
<td>270.2(^{A\B\C}) (±79.2)</td>
<td>87.6(^{A\B\C}) (±400)</td>
<td>549.0(^{A\B\C}) (±105)</td>
<td>582.0(^{A\B\C}) (±259)</td>
<td>1,093(^{C}) (±113)</td>
<td>629.0(^{B\C}) (±241)</td>
<td>1026(^{B\C}) (±42.3)</td>
<td>112(^{A}) (±262)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2A</td>
<td>17.77(^{A\B}) (±8.24)</td>
<td>16.16(^{A\B}) (±5.39)</td>
<td>11.14(^{A\B}) (±4.96)</td>
<td>8.15(^{A\B}) (±2.84)</td>
<td>20.27(^{A\B}) (±5.13)</td>
<td>126.7(^{A\B}) (±27.7)</td>
<td>11.1(^{A\B}) (±11.7)</td>
<td>76.0(^{A\B}) (±18.6)</td>
<td>25.7(^{A\B}) (±5.83)</td>
<td>15.9(^{A\B}) (±4.81)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>3U downstr</td>
<td>286.0(^{A\B\C}) (±152)</td>
<td>276.0(^{A\B\C}) (±129)</td>
<td>50.4(^{A\B\C}) (±24.7)</td>
<td>21.8(^{A\B\C}) (±12.8)</td>
<td>179.0(^{A\B\C}) (±164)</td>
<td>411.0(^{A\B\C}) (±253)</td>
<td>88.1(^{A\B\C}) (±90.3)</td>
<td>23.3(^{A\B\C}) (±19.1)</td>
<td>34.1(^{A\B\C}) (±15.9)</td>
<td>614.0(^{A\B\C}) (±238)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>4A</td>
<td>38.1(^{A\B\C}) (±9.33)</td>
<td>21.7(^{A\B\C}) (±5.93)</td>
<td>16.36(^{A\B\C}) (±11.8)</td>
<td>18.6(^{A\B\C}) (±5.32)</td>
<td>309.0(^{A\B\C}) (±164)</td>
<td>185.2(^{A\B\C}) (±253)</td>
<td>69.9(^{A\B\C}) (±90.3)</td>
<td>29.4(^{A\B\C}) (±19.1)</td>
<td>35.9(^{A\B\C}) (±15.9)</td>
<td>26.6(^{A\B\C}) (±439)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>5A</td>
<td>n/a</td>
<td>n/a</td>
<td>127.8(^{A\B\C}) (±57.2)</td>
<td>204.0(^{A\B\C}) (±74.5)</td>
<td>248.7(^{A\B\C}) (±87.7)</td>
<td>351.9(^{A\B\C}) (±45.8)</td>
<td>247.4(^{A\B\C}) (±93.3)</td>
<td>147.0(^{A\B\C}) (±23.5)</td>
<td>n/a</td>
<td>n/a</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Concentrations with different superscript letter groupings denote significantly different monthly concentrations (\(P < 0.05\)) at each sampling site (i.e., within each row).

The superscript ‘NS’ indicates no significant monthly differences were observed.

Data were not available (n/a) for some months for Site 5A.
Figure 7 | E.coli (o) and fecal coliform (x) mass flux (cfu s⁻¹) vs. time at all five sites. The dates that UV disinfection was implemented at WWTPs upstream of sampling locations are plotted as vertical lines (Site 1U – 1994 (Bonnybrook), 1997 (Fish Creek); Site 3U – 1999).
Bow and Oldman Rivers (2A and 4A) showed the most significant relationship with increasing FC and E. coli correlating with increasing discharge (Figure 5), perhaps due to overland flow during periods of high melt and/or precipitation. The high bacteria concentrations at high flows are thought to be due to overland flow during high precipitation and/or significant melt events, especially in rural areas (Acharya et al. 2008).

Neither of the urban sites showed a clear relationship between E. coli and FC and river discharge. This is consistent with a relatively constant source of fecal contamination (e.g., urban wastewater effluent), which is relatively uninfluenced by major precipitation events and/or seasonal changes in land use.

The E. coli and FC mass fluxes were also plotted against Julian day (Figure 6) to evaluate seasonal trends. In general, all three rural sites showed highest coliform mass fluxes from May to July, while sites downstream of urban centers showed a less distinct seasonal trend. The E. coli and FC mass flux were also significantly lower in the fall/winter months at all rural sites (Tables 5 and 6). These trends are consistent with other studies that found the highest FC and E. coli counts near agricultural land occurred during the summer months, when water temperatures were greater than 15°C, and following rainfall events (LeChavallier et al. 1991; Hyland et al. 2005), possibly due to increased livestock distribution near rivers or manure application to crops during non-winter months.

The E. coli mass fluxes were plotted against time to observe decadal trends and potential trends associated with WWTP effluent disinfection (Figure 7). The bacteria concentrations downstream of urban areas (Sites 1U and 3U) did not exhibit any significant monotonic trends (Table 4). The decrease in the mass flux of E. coli and FC at 1U and 3U following the addition of disinfection to WWTPs is significant (Table 4). This is consistent with a positive downstream water quality outcome as a consequence of increased wastewater disinfection.

There was also a significant step function and monotonic decrease in E. coli and FC at Site 4A (Table 4). It was not possible to determine if this was due to WWTP disinfection (at the 3U site), or improved agricultural management. Site 2A downstream of Calgary did not show this same significant decrease in coliforms (Table 4). Both E. coli and FC were consistently lower at Site 2A when compared to 1U over the entire sampling period (i.e., before and after disinfection was implemented). Decreasing FC with flow distance below Calgary was found by Vanderberg et al. (2005) who determined that wastewater effluent plume was a significant source of FC on the Bow River, with concentrations declining steadily to below background levels by 13 km from the WWTP. No significant changes in E. coli and FC fluxes with time were observed at Site 5A.

### CONCLUSIONS

Chloride mass fluxes were used as a conservative indicator of human and agricultural contributions to bacterial water quality degradation in southern Alberta rivers. Significant decadal scale chloride mass flux increases downstream of major urban centers are consistent with increased wastewater contributions from increasing urban populations. A typical, inverse relationship between chloride concentration and discharge was observed at all sites. Chloride concentrations were significantly lower during high discharge periods (late spring to early summer).

The relationship between pathogen indicators and discharge were less typical. Pathogen indicators’ concentrations were significantly higher in summer months at all agricultural sites, suggesting they are related to land use practices. Seasonal and river discharge-related variations in pathogen indicators were less clear downstream of urban centers.

Significant, step-function decreases in pathogen indicator fluxes were observed at both urban sites in association with implementation of UV disinfection upgrades to upstream WWTPs. It was not possible to ascertain whether the step-function decrease in pathogen indicator fluxes at one agricultural site (4A) was due to changes in agricultural intensity or practices, or to increased disinfection of WWTP effluent at Site 3U. No significant decreases were observed in the other two of the three agricultural sites.

Improved wastewater treatment in urban centers has apparently had a positive effect on the downstream water quality (despite substantial increases in population), while any changes in agricultural practices and/or intensity have
not resulted in consistent improvement in bacteriological water quality in agricultural areas.

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REFERENCES


Health Canada 2010 Guidelines for Canadian Drinking Water Quality. Federal-Provincial-Territorial Committee on Drinking Water of the Federal-Provincial-Territorial Committee on Health and the Environment, Ottawa, ON.


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