Agricultural Water Management 97 (2010) 247–258

Contents lists available at ScienceDirect

Agricultural Water Management

journal homepage: www.elsevier.com/locate/agwat

Influence of streambank fencing with a cattle crossing on riparian health and water quality of the Lower Little Bow River in Southern Alberta, Canada

J. Miller [a,*], D. Chanasyk [b], T. Curtis [a], T. Entz [a], W. Willms [a]

[a] Agriculture and Agri-Food Canada, 5403-1st Ave. South, Lethbridge, Alberta, Canada T1J 4B1

[b] Room 851, General Services Bldg., Department of Renewable Resources, University of Alberta, Edmonton, Alberta, Canada T6G 2H1

ARTICLE INFO

Article history:
Received 12 June 2009
Accepted 14 September 2009
Available online 22 October 2009

Keywords:
Cattle exclusion
Riparian health
River water quality
Sediment
Nutrients
Bacteria

ABSTRACT

The goal of beneficial management practices (BMPs) such as streambank fencing is to prevent or reduce water pollution of surface water bodies. We conducted a four year (2004–2007) study on a fenced 800-m reach of the Lower Little Bow (LLB) River in southern Alberta, Canada. Our hypothesis was that riparian health would be improved by streambank fencing, and that cattle exclusion would prevent water pollution within the fenced reach. Physical, chemical, and microbiological variables in the river were determined throughout the four years, and water quality variables at the upstream (control) and downstream (BMP-impact) sites during the post-BMP phase were evaluated using a paired *t*-test. The overall health of the riparian area, based on a visual assessment of vegetative, soils, and hydrologic features, was improved from a score of 65% (healthy but with problems) for pre-BMP phase in 2001 to 81% (healthy) for post-BMP phase in 2005. The majority of water quality variables were not significantly ($P > 0.10$) different at the downstream and upstream sites during streambank fencing. The evidence from our study indicated that streambank fencing improved the riparian health, and that the BMP prevented the majority of water quality variables from increasing downstream.

1. Introduction

Beneficial or best management practices (BMPs) are methods and practices or combination of practices for preventing or reducing nonpoint source pollution to a level compatible with water quality goals (Novotny and Olem, 1994). Livestock access to stream and riparian areas has been found to negatively affect water quality and seasonal quantity, stream channel morphology, hydrology, riparian zone soils, instream and bank vegetation, and aquatic and riparian wildlife (Kauffman and Krueger, 1984; Trimble and Mendel, 1995; Belsky et al., 1999). The main processes that contribute to livestock-induced water quality pollution of streams and rivers are direct fecal defecation into the stream (Miner et al., 1992; Larsen, 1995), runoff of fecal material from the adjacent land (Larsen et al., 1994), increased erosion of streambanks by cattle shearing, and re-suspension of river sediments by cattle trampling (Kauffman and Krueger, 1984; Trimble and Mendel, 1995; Belsky et al., 1999).

Streambank fencing may be a BMP to exclude cattle from streams and riparian areas to improve riparian health and water quality (Davis et al., 1991; Mostaghimi et al., 2001). Although the effectiveness of grazing strategies for riparian areas has been demonstrated, there are many circumstances where streambank fencing may be the only effective option, and streambank fencing will provide the most rapid recovery of riparian vegetation (Fitch and Adams, 1998). In Alberta, streambank fencing and total cattle exclusion from riparian areas is generally only recommended for severely degraded reaches (N. Ambrose, Alberta Riparian Habitat Management Society, personal communication, 2008).

Studies on the effect of exclusion fencing on various riparian health parameters have been reviewed and summarized by Sarr (2002) and Agouridis et al. (2005). Livestock exclusion may increase growth and vigour of riparian vegetation, reduce bare ground exposure, shift plant communities from forb- or nonnative grass-dominated communities towards native grass- and sedge-communities, and favour increases in willow shrubs or cottonwood trees (Sarr, 2002; Agouridis et al., 2005; Muenz et al., 2006). Streambank fencing may also change the physical characteristics of streams. For example, streambank erosion, stability and wetted channels may decrease; whereas overhanging banks, river depth, and channel bed particle size and riffle substrate may increase (Sarr, 2002; Agouridis et al., 2005; Ranganath et al., 2009). Improved riparian health of fenced reaches may also increase the filtration or buffer capacity of the cattle excluded reach to filter contaminants (Fitch and Ambrose, 2003).

Published scientific studies have been conducted on the impacts of streambank fencing on water quality of streams or rivers in eastern North America (Owens et al., 1996; Galeone, 1999;

* Corresponding author. Tel.: +1 403 317 2219; fax: +1 403 317 2187.
E-mail address: millerjj@agr.gc.ca (J. Miller).

0378-3774/$ – see front matter. Crown Copyright © 2009 Published by Elsevier B.V. All rights reserved.
doi:10.1016/j.agwat.2009.09.016

Line et al., 2000; Meals, 2001; Line, 2003; Muenz et al., 2006; Ranganath et al., 2009) and Australia (McKergow et al., 2003), but we are not aware of any studies in the Northern Great Plains region of North America. Many studies have found significant improvements in certain water quality variables following livestock exclusion (Owens et al., 1996; Line et al., 2000; Meals, 2001; Line, 2003; Muenz et al., 2006); however, some studies have reported little or no improvement in water quality after streambank fencing (McKergow et al., 2003; Ranganath et al., 2009).

Studies on the effect of cattle exclusion or cattle access on riparian or water quality variables have utilized paired reaches (Ranganath et al., 2009; Muenz et al., 2006), before-after (Owens et al., 1996; McKergow et al., 2003), upstream-downstream-before-after designs (Line et al., 2000; Line, 2003), paired watersheds (Galeone, 1999; Meals, 2001), and upstream-downstream designs (Vidon et al., 2008). The upstream-downstream assumes that any changes in the response variables are due to the BMP implementation between the upstream and downstream sites, and this design is most suitable when surface runoff is moving through the BMP into the adjacent water body (Mesner et al., 2008). The advantages and disadvantages of upstream-downstream design in comparison to other BMP designs have been reported by Mesner et al. (2008) and Jackson and Morris (2002).

The objective of this study was to determine the effect of streambank fencing with a cattle crossing on riparian health and river water quality. Our hypothesis was that riparian health would be improved by streambank fencing, and that cattle exclusion would prevent water quality pollution at the downstream site.

2. Materials and methods

2.1. Site description

The location of this BMP within the WEBs (Watershed Evaluation of BMPs project) watershed is shown in Fig. 1. This BMP site is located within the mixedgrass natural subregion (Adams et al., 2005). Dominant vegetation within this subregion is wheat grass (*Agropyron* spp.) and needle-and-thread grass (*Stipa comata*). Dominant soils are Dark Brown Chernozems (Typic Haploboroll). The dominant geology of the study site is coarse gravel and sand with minor silt beds of the Pleistocene or Holocene periods (Shetson, 1987).

2.2. Lower Little Bow River

The LLB River is a nonincised, regulated small river of stream order ≤ 3 (Strahler, 1952). The river is intensively managed for irrigation, and flows are controlled by releases from a dam at the Travers Reservoir. However, flows in the river are most variable during the summer, as they are affected by inputs of rainfall and irrigation return flows and withdrawal of irrigation water (Little, 2001). Mean daily flow rates in the LLB River during this study (2004–2007) ranged from <1 to $12.7\ m^3\ s^{-1}$. Irrigation return flows (runoff and drainage water) enter the LLB River between early May and early October. The river bottom is coarse sediment consisting of mainly sand. The width of the river ranges from approximately 8 to 9 m and the depth from about 0.5 to 1.0 m. The LLB River flows through mainly native rangeland, with some reaches flowing through cultivated land.

2.3. BMP history and implementation

The streambank fencing with cattle crossing BMP was established by Alberta Agriculture and Rural Development and the County of Lethbridge as part of the Oldman River Basin Water Quality Initiative. The county owns the land and historically leased it for cow-calf grazing. The stocking rate and history of grazing in the native pastures on the north and south side of the LLB River and the fenced riparian pasture are shown in Table 1. The total potential grazing area is about 184 ha and the area of the fenced riparian pasture is approximately 10 ha. From 2001 to 2003, the south and north pastures were rotationally grazed at a stocking

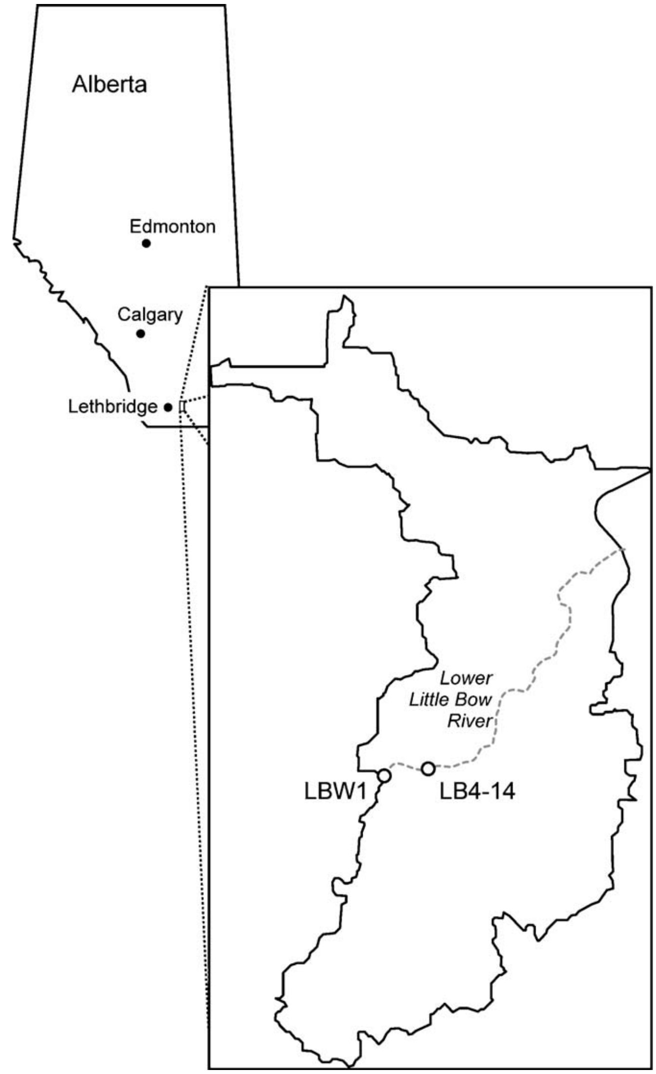

Fig. 1. Map of WEBs watershed in province of Alberta showing location of streambank fencing BMP between upstream control site (LBW1) and downstream BMP site (LB4-14). North is top of page.

Table 1
Stocking rate (cow-calf pairs) and grazing history in the native pastures on north and south side of LLB River and fenced riparian pasture from 2001 to 2007.

Years	Cattle stocking rate (AUM ha^{-1})	Pasture	
		Native pastures on north and south side of LLB River	Fenced riparian pasture[z]
2001–2003	0.50	Grazed	Excluded
2004–2007	0.40	Grazed	Excluded

The total area of all pastures is about 184 ha, the area of native pastures is about 174 ha, and the area of the fenced riparian pasture is approximately 10 ha. Exclusion fencing was installed on June 29, 2001. The grazing period was generally from June to August. Cattle may have inadvertently entered the fenced riparian pasture from the cattle crossing from 2001 to 2007.

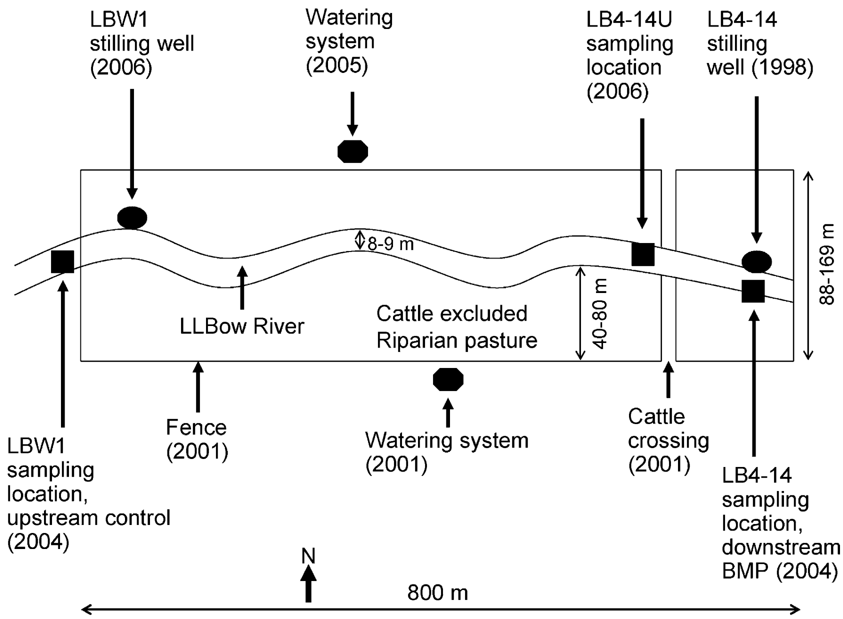

Fig. 2. Schematic diagram of streambank fencing with cattle crossing BMP showing locations of fence, cattle crossing, cattle-excluded riparian pasture, stilling wells, off-stream watering systems and water sampling locations. The year in parenthesis is when was sampling was initiated or fencing, watering system, or flow monitoring system were installed.

rate of 0.50 AUM ha^{-1}. From 2004 to 2007, and because of weed problems and overgrazing, the stocking rate was reduced to 0.40 AUM ha^{-1}. The recommended stocking rates for this area, assuming 250–350 mm of precipitation, are 0.50, 0.75, and 1.0 AUM ha^{-1} for pastures in poor, fair, and good condition (Wroe et al., 1988). Cattle were excluded from the riparian pasture from 2001 to 2007.

The experimental design is shown schematically in Fig. 2. Barbed wire fencing with a cattle crossing was installed on both sides of an 800-m reach of the Lower Little Bow (LLB) River on June 29, 2001. The distance from the edge of the river to the fence ranges between approximately 40 and 80 m. The cattle crossing is located just upstream of the BMP-impact station (LB4-14), and allows cattle to be moved between the pastures on either side of the river. Initially, a single strand of wire was strung across the river along either side of the cattle crossing to prevent cattle entering the river and cattle-excluded pasture in 2004. The wire barrier on the river had to be removed in 2004 because of new government regulations allowing access on navigable waterways. We subsequently installed small boulders and then panel fencing parallel to the river to prevent cattle entering the river and excluded pasture.

These cattle barriers were not always effective as cattle were sometimes observed to enter the stream and cattle excluded riparian pasture. Cattle entered the river and riparian pasture most frequently (5–10 times out of 60 times) in the summer of 2004 and 2005, less frequently (3–5 times out of 60 times) in 2006, and no cattle were observed to enter the area in 2007. Cattle were immediately chased out of the river or excluded pasture when they were observed in this area. Overall, we believe the low percentage of times cattle entered the riparian pasture should have had little impact on our study objectives. If cattle were grazing on one side of the river, the gate on the cattle crossing on the opposite side of the river was closed and the gate on the side that cattle were on was left open, so cattle could access the river for water as a back-up in case the off-stream watering system malfunctioned.

An off-stream watering system consisting of a submersible pump, solar panel, battery, and water trough were installed on the south side of the river immediately after the fencing was installed in 2001. A similar off-stream watering system was installed on the north side of the river in the spring of 2005. Both watering systems were located adjacent to the fence.

The dominant plant community on the south native pasture is blue grama grass (*Bouteloua gracilis*) and needle-and-thread grass. The dominant plant community in the fenced riparian pasture is salt grass (*Distichlis stricta*), western wheat grass (*Agropyron smithii*), and sedge (*Carex* sp.). Foxtail barley (*Hordeum jubatum*) also occurs in the riparian pasture, indicating saline soils in this area. The dominant plant community on the north native pasture is blue grama and needle-and-thread grass, as well as needle-and-thread grass, low sedge (*Carex* sp.), and pasture sage (*Artemisia frigida*). Blue grama grass is the co-dominant grass species in the Dry Mixedgrass Natural Subregion, but can become dominant in the Mixedgrass Natural Subregion because of overgrazing (W. Willms, 2005, personal communication).

2.4. Flow monitoring and precipitation

The monitoring station (LB4-14) immediately downstream of the fenced reach that was used to measure river flow was installed in 1998 by Alberta Agriculture and Rural Development (Little, 2001). The monitoring station (LBW1) immediately upstream of the fenced reach that was used to measure river flow was installed in the spring of 2006 by Alberta Agriculture and Rural Development. The upstream station LBW1 is hereafter referred to as the control (CON) or upstream site and the downstream station LB4-14 as the downstream or BMP site. Monitoring stations were equipped with a stilling well, staff gauge, and datalogger (ChartPac CP-XA, Lakewood Systems Ltd.) programmed to record gauge heights every 20 min.

Flow metering was conducted throughout the season to determine the relationship between gauge heights and open channel flow (Little, 2001) near the upstream and downstream sites. Flows were measured with a Swoffer 3000 current meter (Swoffer Ltd.). The relationship between stream flow and staff gauge height was determined using a power function.

Precipitation data from the Alberta Agriculture and Rural Development IMCIN (Irrigation Management Climate Information Network) weather station at Iron Springs were used for this study.

Iron Springs is within 5 km of the WEBs watershed. Weather data from the long-term weather station at the Lethbridge Research Center, which is approximately 28 km from the WEBs watershed, were used for missing data at Iron Springs.

2.5. Water sampling and field analysis

Grab samples of river water were collected weekly (for chemical analyses) and every two weeks (for bacterial analyses) at stations LBW1 (upstream control) and LB4-14 (downstream BMP-impact) during the irrigation season from April until October, and then monthly during the winter. Water samples were also collected immediately upstream of the cattle crossing (LB4-14U) in 2006 and 2007. Samples for chemical analyses were collected from the top of the water column and center of the channel in large 1-L polyethylene bottles that had been triple-rinsed with sample water. Water samples for bacterial analyses were collected in autoclaved bottles that contained sodium thiosulphate. Water samples were kept in a cooler with ice packs until samples were returned to the lab within a couple of hours.

Temperature, pH, electrical conductivity (EC), and dissolved oxygen (DO) were measured in the field using a portable water quality meter and associated probes (MultiLine P4, Wissenschaftlich-Technische, Werkstätten, Germany). Turbidity was measured in the field using the nephelometric method (APHA, 1998) with a portable turbidity meter (Hach Model 2100p, Loveland, CO). Chlorophyll-a was measured in the field during 2006 and 2007 using the fluorometric method (APHA, 1998) with a portable fluorometer (Aquafluor, Turner Designs, Sunnyvale, CA), and values are reported as relative fluorescent units (RFU).

2.6. Laboratory chemical and bacterial analyses

All N and P fractions were analyzed on an autoanalyzer (TRAACS 800, Bran and Luebbe Inc., Buffalo Grove, Illinois). Water samples transported to the laboratory were immediately filtered through a 0.45 μm filter and then analyzed for NH_3-N, NO_2-N, NO_3-N, and dissolved reactive P (DRP) or ortho-P. Ammonium-N (NH_4-N) was determined by the automated phenate method using Technicon Method 780-86T (Technicon Industrial Systems, 1987a). Nitrite-N was determined by the automated diazotization method using Technicon Industrial Method 784-86T (Technicon Industrial Systems, 1986a). Nitrate-N was determined by the automated hydrazine reduction method using Technicon Industrial Method 782-86T (Technicon Industrial Systems, 1987b). Detection limits for NH_4-N, NO_2-N, and NO_3-N were 0.1 mg L^{-1} for all years (2004–2007) of this study. The DRP was determined by using the ascorbic acid reduction method using Technicon Method 781-86T (Technicon Industrial Systems, 1987c). Detection limits for DRP were 0.002 mg L^{-1} in 2004 and 2005, and 0.01 mg L^{-1} in 2006 and 2007.

Two methods were used for analyses of total N (TN) fractions in water during this study. In 2004 and 2005, total Kjeldahl N (TKN) of unfiltered and filtered water samples were determined using a semi-micro-Kjeldahl method with potassium sulphate (K_2SO_4) digestion, and then NH_3 was analyzed using the automated sodium salicylate method using Technicon Industrial Method 786-86T (Technicon Industrial Systems, 1986b). Total N was calculated as the sum of TKN (unfiltered sample), NO_2-N and NO_3-N, with values below detection limits not included. Total dissolved N (TDN) was calculated as the sum of TKN (filtered sample), NO_2-N and NO_3-N, with values below detection limits not included. The total particulate N (TPN) fraction was calculated as the difference between TN and TDN. Detection limits for TKN in 2004 and 2005 were 0.04 mg L^{-1}.

In 2006 and 2007, total N on unfiltered samples and TDN on filtered samples (0.45 μm) was determined using the standard test method (ASTM D5176) for total chemically bound nitrogen in water by pyrolysis and chemiluminescence detection (ASTM, 2006) using a Shimadzu TOC-V instrument with TNM-1 unit (Shimadzu, Kyoto, Japan). The TPN fraction was determined as the difference between TN and TDN. The detection limit for TN and TDN in 2006 and 2007 was 0.2 mg L^{-1}.

Two methods were used for analyses of total P (TP) fractions in water during this study. In 2004 and 2005, total P (unfiltered sample) and total dissolved P (TDP) (0.45 μm filtered sample) were determined using a semi-micro-Kjeldahl method with potassium sulphate (K_2SO_4) digestion by using ENVIRODAT Method Code 582 of VMV (Valid Method Variable) Code 15421 (Environment Canada, 1996), and then ortho-P was analyzed using the automated ascorbic acid method using Technicon Industrial Method 781-86T (Technicon Industrial Systems, 1987c). The TPP fraction was calculated as the difference between TP and TDP. The detection limits for TP and TDP in 2004 and 2005 were 0.002 mg L^{-1}.

Unfiltered water samples in 2006 and 2007 were analyzed for TP by persulfate and autoclave digestion using ENVIRODAT Method 2331 or VMV Code 15423 (Environment Canada, 2007a), followed by analyses of ortho-P using ascorbic acid reduction using Technicon Method 787-86T (Technicon Industrial Systems, 1986c). Filtered (0.45 μm) water samples for TDP analyses in 2006 and 2007 were analyzed by persulfate and autoclave digestion using ENVIRODAT Method 2332 or VMV Code 15465 (Environment Canada, 2007b), followed by analyses of ortho-P using ascorbic acid reduction using Technicon Method 787-86T (Technicon Industrial Systems, 1986c). The TPP fraction was calculated as the difference between TP and TDP. Detection limits for TP and TDP in 2006 and 2007 were 0.01 mg L^{-1}. Instantaneous mass loads in kg of sediment or nutrient per year were calculated from the concentration and mean daily flow rate on the day of sampling.

Water samples for bacterial analyses were shipped by courier in cold containers to the Provincial Laboratory of Public Health in Calgary, Alberta on the day of sample collection. Analyses were completed the next morning using the membrane filtration method (American Public Health Association, 1998) as described by Little (2001). Results for fecal coliforms and *E. coli* are reported in counts or CFU (colony forming units) per 100 mL of water. Instantaneous mass loads for fecal coliforms or bacteria in CFU per year were calculated from the concentration of bacteria and the mean daily flow rate.

2.7. Riparian health assessment of the Lower Little Bow River

A pre-BMP riparian health assessment was conducted in 2001 (just prior to fencing being installed) along a 800-m reach of the LLB Bow River between stations LB4-14 (downstream BMP impact station) and LBW1 (upstream control station). The area of the riparian assessment was approximately 4 ha. A post-BMP riparian health assessment was then conducted of this same reach in 2005, four years after the fencing had been installed.

Riparian health assessments were conducted by the Alberta Riparian Habitat Management Society (Cows and Fish) using the field workbook for riparian health assessment for streams and small rivers (Fitch et al., 2001). Six vegetation factors and five soil and hydrology factors were visually assessed by riparian health specialists. The vegetation factors were vegetative cover of floodplain and streambank, invasive plant species, disturbance-increasers or undesirable herbaceous species, preferred tree and shrub establishment and regeneration, utilization of trees and shrubs, and standing decadent and deady woody material. The soil and hydrology factors were streambank root mass protection, human-caused bare ground, streambank structurally altered by human activity, pugging and/or hummocky, and stream channel incisement. Riparian health evaluation integrates physical (soils and hydrology) and vegetation features, because no one factor provides a complete picture of site health or trend in health (Fitch and Ambrose, 2003). The evaluations rely heavily on vegetation because many vegetation features integrate the effects of soil and hydrologic factors which form and operate in riparian areas. Plants are also more visible than soil or hydrologic features. Many of the measurements deal with the element of "coverage" or how much of the riparian area measured is covered, influenced or affected by vegetation or structural changes. The categories are usually expressed in percentages of the reach area.

Riparian health is classified into three categories: healthy (total score between 80 and 100%), healthy but with problems (total score between 60 and 79%), and unhealthy (total score < 60%) (Fitch and Ambrose, 2003). Healthy riparian areas have little or no impairment to riparian functions, they are resilient, stable and provide a long list of benefits and values. Healthy riparian areas but with problems have some impairment to riparian functions due to human or natural causes, many riparian functions are still being performed, but some signs of stress are apparent. Unhealthy riparian areas have impairment to many riparian functions due to human or natural causes, and most riparian functions are severely impaired or have been lost.

2.8. Experimental design, interpretation and statistical analyses

We used upstream versus downstream analyses similar to Vidon et al. (2008) for the post-BMP period to evaluate the effectiveness of this BMP and address the hypothesis of whether the BMP was preventing water pollution. As noted by Mesner et al. (2008), if river flow is similar at the upstream and downstream sites, concentrations as well as mass loads can be evaluated. The logic of our interpretation is shown in Fig. 3. Scenario 1 occurs when there is no streambank fencing and no cattle present along the river. For this scenario, the mass load or M (or concentration if flows are similar) entering the upstream site and mass load leaving the downstream site should be equal if no pollutants (P) are added to the river reach. For scenario 2, there is no streambank fencing but cattle now have access to the river reach and contribute pollution to the stream (P > 0). The mass load entering the upstream site is still M, but the mass load leaving the downstream site (M + P) is the sum of the mass input into the reach (M) and the additional pollution (P) added from cattle.

We assumed that cattle were degrading water quality within the river if no fencing was present and cattle had access to the river (Scenario 2 in Fig. 3). Rainfall simulations using the Guelph rainfall simulator (Tossell et al., 1987) were conducted on the grazed (with cattle present) and cattle-excluded riparian pasture adjacent to the fenced reach from 2005 to 2007 (Miller et al., 2008a). Runoff depth and mass loads of TN, TDN, TPN, and DRP were significantly greater for the grazed than ungrazed pasture, indicating a greater potential for cattle to contribute more runoff and nutrients to the adjacent river when cattle were present. Water quality monitoring downstream where cattle had access to the river during 2004 and 2005 also showed that many water quality variables were increasing downstream, indicating that cattle were contributing to water quality degradation (Miller et al., 2008b). In addition, Little (2001) reported that unrestricted access of livestock to the LLB River was contributing to water quality degradation. She found that bacteria concentrations in the river were high following major precipitation events, even in the upper portion of the basin, which is exclusively rangeland. Studies elsewhere have clearly shown that unrestricted cattle access to streams has a negative impact on water quality (Line et al., 2000; Line, 2003; Vidon et al., 2008). The riparian health of the reach was also dramatically improved after streambank fencing, indicating that cattle were degrading the reach in our study (Miller et al., 2008b).

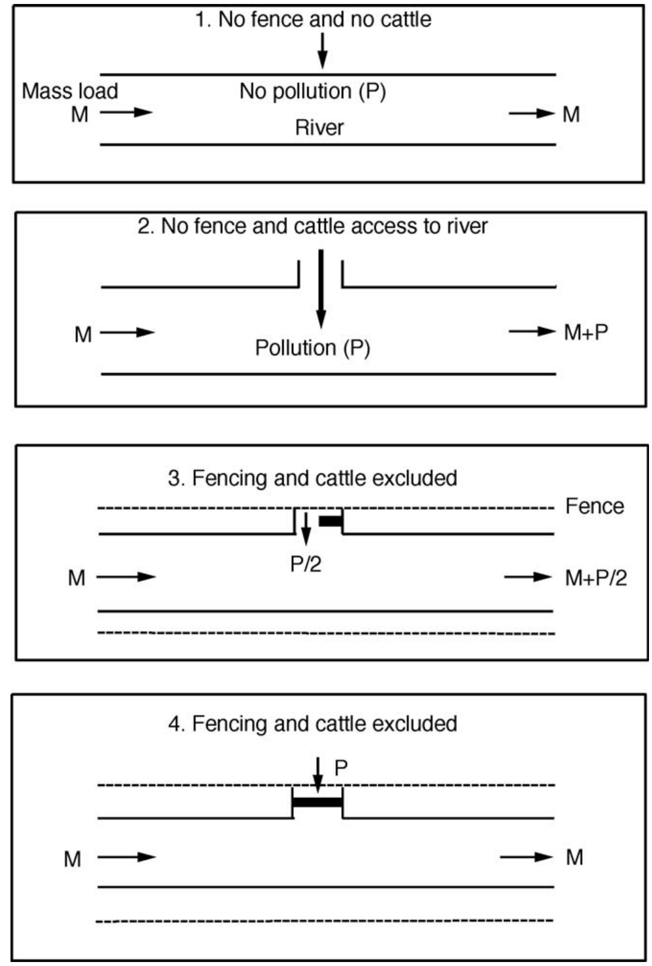

Fig. 3. Interpretation of streambank fencing BMP based on experimental design where M is the mass load of the measured variables, and P is the mass load of the off-stream pollutant.

Scenario 3 and 4 represent the effect of the streambank fencing BMP on water pollution (Fig. 3). Scenario 3 represents the case where the BMP was not effective at preventing water quality pollution (P > 0) since half of the pollution (P/2) is still delivered to the reach, resulting in M + P/2 leaving the downstream site. The value P/2 is an arbitrary number to represent <100% effectiveness of the BMP, and could be P/3, P/4, etc. The BMP is 100% effective at preventing water pollution (P = 0) for Scenario 4 since none of the pollution (P) enters the reach, and the mass load entering and leaving the reach are equal (Fig. 3). An off-stream BMP that prevents cattle from defecating into the river or prevents runoff of fecal material into the river should only prevent pollution from entering the river, and it should not decrease the mass load already in the river. Therefore, there is no reason to expect the downstream measurement to be lower than the upstream one, no matter how effective the BMP. The downstream measurement may be lower than the upstream one if the upstream water is heavily polluted and some of that pollution settles out as the water moves through the reach, or if the fenced reach acts as a filter for water in the river. However, that process should be unrelated to the off-stream BMP that prevents direct fecal deposition into the river and prevents wash-off of fecal material adjacent to the river.

Table 2
Monthly precipitation (Ppt.) at Iron Springs from 2004 to 2007 in relation to long-term (1971–2000) mean precipitation.

Month	2004		2005		2006		2007		Long-term mean
	Ppt.	% of Normal	Ppt.	% of Normal	Ppt.	% of Normal	Ppt.	% of Normal	
	mm	%	mm	%	mm	%	mm	%	mm
Jan	22.4	123	8.0	44	7.7	42	6.0	33	18.2
Feb	2.7	23	6.1	53	12.3	106	25.1	216	11.6
Mar	13.8	52	15.2	58	24.3	92	13.4	51	26.3
Apr	15.2	51	20.0	67	40.7	136	52.1	174	30.0
May	73.1	140	8.4	16	20.5	39	80.0	153	52.3
Jun	57.8	91	283.8	446	81.0	127	23.8	37	63.7
Jul	36.8	86	10.0	23	1.4	3.3	1.4	3.3	42.8
Aug	41.6	87	72.1	151	23.4	49	5.7	11.9	47.9
Sep	20.0	57	145.8	415	21.7	62	29.6	84	35.1
Oct	34.0	184	12.9	70	21.9	118	10.2	55	18.5
Nov	2.4	15	12.6	80	14.0	89	4.8	31	15.7
Dec	18.0	108	3.5	21	13.4	80	11.4	68	16.7
Total	337.6	89	598.4	158	282.3	75	263.6	70	378.8

Long-term mean precipitation from Lethbridge CDA weather station (1971–2000).

Therefore, the null hypothesis (H_0) was that the BMP of streambank fencing prevented water quality pollution, resulting in $P_{\text{fenced}} = 0$ (Scenario 4 in Fig. 3):

$$H_0 : \mathrm{WQV}_{\text{down}} - \mathrm{WQV}_{\text{up}} = 0 \tag{1}$$

where $\mathrm{WQV}_{\text{down}}$ and WQV_{up} are the water quality variables for the downstream and upstream sites, respectively. The alternate hypothesis was that the BMP of streambank fencing did not prevent water pollution, resulting in $P_{\text{fenced}} > 0$ (Scenario 3 in Fig. 3):

$$H_A : \mathrm{WQV}_{\text{down}} - \mathrm{WQV}_{\text{up}} > 0 \tag{2}$$

We assumed that Scenario 2 in Fig. 3 occurred when cattle had access to the river. Our study design can determine if the BMP is preventing pollution but it cannot quantify the reduction in P because M and P for Scenario 2 were not measured.

The differences in water quality values between the downstream and upstream sites were tested for normality and obvious outliers were removed using the Univariate procedure in SAS (SAS Institute Inc, 2005). Of the 62 datasets analyzed for outliers and normality, 63% had no outliers, 29% had one outlier, 7% had two outliers, and 3% had three outliers. Probability values ≤ 0.10 were considered significantly different for one-way paired *t*-tests between the upstream and downstream sites. The *t*-test was conducted on the untransformed data for all variables. The exception was for mass loads for bacteria, where log-transformed values were used.

A Chi-Square test (one-way) with Fisher's exact test was used to test the difference in frequency of water quality violations at the downstream (LB4-14) and upstream (LBW1) sites, and the significance level was $P \leq 0.10$. Correlations were also calculated between water quality variables ($P \leq 0.05$) using SAS.

3. Results

3.1. Precipitation

Total precipitation in 2004 (338 mm) was 89% of the long-term average value (379 mm) (Table 2), indicating a relatively normal year. In 2005, total precipitation (598 mm) was 158% of the long-term average, indicating a wetter than normal year. In 2006, total precipitation (282 mm) was 75% of the long-term average, indicating a drier than normal year. In 2007, total precipitation (264 mm) was 70% of normal, indicating a drier year.

3.2. Riparian health

For the six vegetative criteria (Table 3), clear improvements in vegetative health (Fitch et al., 2001) were observed for preferred tree and shrub establishment and regeneration (criterion #4) and utilization of preferred trees and shrubs (criterion #5), and a slight improvement was observed for disturbance-caused undesirable herbaceous species (criterion #3). There was no change in vegetative cover of floodplain and streambanks (criterion #1) and decadent and dead woody material (criterion #6). There was a decline in vegetative health for invasive plant species and cover (criterion #2). When all vegetative criteria were considered, vegetation health improved 19% (absolute) from a score of 48% (unhealthy) in 2001 to a score of 67% (healthy but with problems) in 2005.

For the soil and hydrology criteria (Table 3), improvements in health were only found for human-caused bare ground (criterion #8). There was no change in the other five criteria. When all soil and hydrology criteria were considered, soil and hydrology health was unchanged from 2001 to 2005, with an overall rating of healthy. However, the rating score was improved 13% (absolute) from 80% in 2001 to 93% in 2005.

When all vegetative and soil/hydrology criteria were considered, the overall riparian health was improved 16% (absolute) from 65% (healthy but with problems) in 2001 to 81% (healthy) in 2005.

3.3. Water quality variables

Turbidity was significantly lower at the downstream than upstream site in 2004 and 2005, the reverse trend occurred in 2006, and mean values were similar in 2007 (Table 4). Dissolved oxygen was significantly greater downstream during this study; the exception being in 2005, where DO values were similar. Water temperature and chlorophyll-a were similar at the upstream and downstream sites during the study. The significant difference in water temperature in 2005 was likely too small to have any actual environmental influence on water quality.

Concentrations of TSS, TN, and fecal coliforms were generally similar at the upstream and downstream sites during the four years (Table 4). The exception was TSS in 2005, where values were significantly lower at the downstream site. Chloride concentrations were similar at both sites in 2004 and 2005, and significantly lower downstream in 2006 and 2007. Total P concentration was similar at both sites in 2004 and 2007, but it was significantly greater downstream in 2005 and 2006. Concentrations of *E. coli* were significantly greater downstream in 2004 and 2005, and were similar at both sites in 2006 and 2007.

Table 3
Riparian health inventory of 800-m reach of Lower Little Bow river between upstream control and downstream BMP sites for the streambank fencing with cattle crossing BMP prior to fencing being installed in 2001 and then after four years of livestock exclusion in 2005.

Criteria	Pre-BMP 2001 actual score/health rating	Post-BMP 2005 actual score/health rating	Possible score	Change in score	Health trend
Vegetation					
1. Vegetative cover of floodplain and streambanks	4 (HP)	4 (HP)	6	0	No change
2. Invasive plant species (cover)	4 (HP)	3 (HP)	6	−1	Declined
3. Disturbance-caused undesirable herbaceous species	0 (UH)	1 (UH)	3	+1	Improved
4. Preferred tree and shrub establishment and regeneration	0 (UH)	4 (HP)	6	+4	Improved
5. Utilization of preferred trees and shrubs	0 (UH)	3 (H)	3	+3	Improved
6. Decadent and dead woody material	3 (H)	3 (H)	3	0	No change
Vegetation subtotal	13 (48%)	18 (67%)	27 (100%)	+5 (+19%)	Improved
Vegetation descriptive rating	Unhealthy	Healthy but with problems			
Soil/hydrology					
7. Streambank root mass protection	4 (HP)	4 (HP)	6	0	No change
8. Human-caused bare ground	2 (UH)	6 (H)	6	+4	Improved
9. Streambank structurally altered	6 (H)	6 (H)	6	0	No change
10. Human physical alteration to polygon	3 (H)	3 (H)	3	0	No change
11. Stream channel incisement	9 (H)	9 (H)	9	0	No change
Soil/hydrology subtotal	24 (80%)	28 (93%)	30 (100%)	+4 (+13%)	No change
Soil/hydrology descriptive rating	Healthy	Healthy			
Overall total/rating	37 (65%)	46 (81%)	57 (100%)	+9 (+16%)	Improved
Overall descriptive rating	Healthy but with problems	Healthy			

Riparian health assessment based on visual assessment of vegetative, soil, and hydrologic criteria by riparian health specialist using field workbook of Fitch et al. (2001). Rating: Healthy, H (80–100%), Healthy but with problems, HP (60–79%), Unhealthy, UH (<60%).

Mass loads of TSS, TN, TP, and fecal coliforms were generally similar at the upstream and downstream sites in all four years (Table 4). The exception was TP load in 2005, where it increased downstream. Mass load of Cl was similar at both sites in 2004 and 2007, and was significantly lower downstream in 2005 and 2006. Mass loads of *E. coli* were significantly greater downstream in 2004, and were similar at both sites in the other three years.

The TN loads were dominantly dissolved or TDN rather than particulate or TPN in all four years (data not shown). Concentrations and loads of TDN and TPN were similar at the downstream and upstream sites during the study. Mean NO_3-N and NH_4-N concentrations were 0.1 mg L^{-1} at the upstream and downstream sites in all four years (data not shown). Mean NO_3-N and NH_4-N loads were similar at the upstream and downstream sites in all four years. There were three exceptions: NO_3-N loads in 2004 were significantly lower at the downstream (4.5 ± 0.5 Mg y^{-1}) than upstream site (5.1 ± 0.5 Mg y^{-1}), NH_4-N loads in 2004 were significantly greater at the downstream (4.4 ± 0.2 Mg y^{-1}) than upstream site (4.1 ± 0.3 Mg y^{-1}), and NH_4-N loads in 2005 were significantly greater at the downstream (12.1 ± 1.9 Mg y^{-1}) than upstream site (11.8 ± 2.3 Mg y^{-1}).

Table 4
Influence of streambank fencing with a cattle crossing on water quality variables at upstream (control) and downstream (BMP-impact) sites in the Lower Little Bow River from 2004 to 2007.

Variable	2004		2005		2006		2007	
	Upstream (Control)	Downstream (BMP)	Upstream (Control)	Downstream (BMP)	Upstream (Control)	Downstream (BMP)	Upstream (Control)	Downstream (BMP)
Turbidity (NTU)	94 ± 23a	91 ± 22b	91 ± 30a	70 ± 19b	39.3 ± 9.6b	83.8 ± 34.5a	35.5 ± 3.5[a]	34.6 ± 3.5
DO (mg L^{-1})	6.8 ± 0.3a	7.0 ± 0.3b	10.6 ± 3.1	9.7 ± 2.3	8.9 ± 1.9b	9.0 ± 2.0a	8.7 ± 0.4b	9.0 ± 0.4a
Temp. (EC)	13.4 ± 1.1	13.2 ± 1.2	11.8 ± 0.9b	11.9 ± 0.9a	13.0 ± 1.0	13.1 ± 1.1	12.4 ± 1.0	12.4 ± 1.0
Chlorophyll-a (RFU)	NA	NA	NA	NA	62.9 ± 6.5	62.4 ± 7.3	69.4 ± 6.4	71.0 ± 6.1
Concentration (mg L^{-1})								
TSS	129 ± 30.6	127 ± 31.3	96.4 ± 19.5a	92.9 ± 16.0b	69.8 ± 9.8	91.8 ± 24.3	69.3 ± 6.1	67.7 ± 6.5
Cl	5.6 ± 0.4	5.6 ± 0.4	7.0 ± 0.4	7.0 ± 0.3	14.1 ± 0.6a	12.6 ± 0.4b	13.5 ± 0.5a	13.1 ± 0.5b
TN	0.7 ± 0.04	0.7 ± 0.04	0.8 ± 0.1	0.8 ± 0.1	0.4 ± 0.02	0.4 ± 0.02	0.4 ± 0.01	0.4 ± 0.01
TP	0.101 ± 0.018	0.099 ± 0.018	0.136 ± 0.040b	0.144 ± 0.036a	0.06 ± 0.01b	0.08 ± 0.02a	0.07 ± 0.01	0.07 ± 0.01
Concentration (Counts 100 mL^{-1})								
Fecal coliforms	445 ± 103	469 ± 85	339 ± 139	310 ± 117	150 ± 38.4	172 ± 42.0	236 ± 56.8	248 ± 60.8
E. coli	334 ± 70b	397 ± 65a	691 ± 370b	705 ± 374a	153 ± 40.4	162 ± 44.1	207 ± 51.1	218 ± 58.1
Mass load (Mg y^{-1})								
TSS	5336 ± 1237	5198 ± 1267	14334 ± 4689	13499 ± 3742	5527 ± 581	5591 ± 697	5658 ± 744	5644 ± 734
Cl	243 ± 17.4	238 ± 15.2	990 ± 182a	857 ± 139b	1512 ± 318a	1264 ± 215b	984 ± 82.8	971 ± 81.2
TN	30.8 ± 2.0	30.9 ± 1.9	114 ± 25.6	111 ± 21.8	37.0 ± 4.7	39.0 ± 6.3	28.9 ± 2.3	28.7 ± 2.4
TP	4.2 ± 0.8	4.2 ± 0.7	27.0 ± 10.7b	29.4 ± 10.1a	5.0 ± 0.9	5.3 ± 0.9	5.1 ± 0.5	5.3 ± 0.6
Mass loads (log CFU y^{-1})								
Fecal coliforms	14.1 ± 0.1	14.2 ± 0.1	14.4 ± 0.2	14.4 ± 0.2	14.0 ± 0.2	14.1 ± 0.2	14.4 ± 0.1	14.4 ± 0.1
E. coli	13.9 ± 0.2b	14.1 ± 0.1a	14.3 ± 0.2	14.3 ± 0.2	13.9 ± 0.3	14.0 ± 0.2	14.3 ± 0.1	14.3 ± 0.1

Mean ± standard error. Means are significant ($P \leq 0.10$) different (one-way paired *t*-test) when followed by different lower case letters by row for each year. Means followed by no lower case letter are not significantly different.

Table 5
Percentage of values for selected water quality variables not meeting the water quality guidelines for surface waters at upstream (control) and downstream (BMP) sites for the streambank fencing with cattle crossing BMP.

Year/site	Variable					
	Turbidity (%)	TSS (%)	DO (%)	TN (%)	TP (%)	*E. coli* (%)
2004						
Upstream	96.3a	85.2a	14.8a	18.5a	77.8a	37.0a
Downstream	96.3a	88.9a	14.8a	14.8a	70.4a	44.4a
2005						
Upstream	100a	74.2a	9.7a	19.4a	54.8a	19.4a
Downstream	97.3a	81.1a	8.1a	13.5a	64.9a	21.6a
2006						
Upstream	94.3a	91.4a	31.4a	0a	54.3a	20.0a
Downstream	94.1a	94.1a	26.5a	2.9a	61.8a	20.6a
2007						
Upstream	97.2a	88.9a	2.8a	0a	75.0a	19.4a
Downstream	97.2a	83.3a	2.8a	0a	80.6a	19.4a
WQG	8	25.0	5.0	1.0	0.05	200

WQG = water quality guidelines; guidelines for total N (TN), total P (TP), dissolved oxygen (DO), total suspended solids (TSS), and turbidity are for protection of aquatic life; guideline for *E. coli* is for recreational water; guidelines for TN and TP are for Province of Alberta (Alberta Environment, 1999); DO is CCME guideline (CCME 2002); *E. coli* is Health Canada guideline (Health Canada, 1992); TSS and turbidity guidelines are for British Columbia (BCME, 2008); no federal or Alberta guidelines were available for TSS and turbidity, so B.C. guidelines were used.
Percentage values by column within each year followed by different lower case letters are significantly different ($P \leq 0.10$, one-tailed test) using chi-square test with Fisher's exact test.

The TP loads were dominantly particulate or TPP rather than dissolved or TDP (data not shown). The exception was in 2005, where dissolved P was the dominant fraction. Concentrations and loads of TPP, TDP, and DRP were similar at the downstream and upstream sites during this study. There were two exceptions: the TDP load in 2005 was significantly greater at the downstream (17.1 ± 7.3 Mg y^{-1}) than upstream (15.6 ± 7.7 Mg y^{-1}) site, and the concentration of TPP in 2006 was significantly greater at the downstream (0.07 ± 0.02 Mg y^{-1}) than upstream site (0.05 ± 0.01 Mg y^{-1}).

There was no significant difference in the frequency of water quality violations of turbidity, DO, and concentrations of TSS, TN, TP, and *E. coli* at the upstream and downstream sites during the four years of this study (Table 5). Turbidity and TSS exceeded the water quality guidelines most frequently, followed by TP, *E. coli*, and then DO and TN. Water quality violations for all variables are for protection of surface waters for aquatic life. The exception is *E. coli*, which is for protection of surface waters for recreational use.

Strong ($r \geq 0.70$) significant positive correlations were observed between ten pairs of water quality variables (Table 6). The strongest correlation was found between fecal coliforms and *E. coli* ($r = 0.97$). Certain N and P fractions were also strongly correlated with *E. coli* concentrations. There were no strong correlations between any of the water quality variables (concentrations) and river flow.

4. Discussion

The overall riparian health of the fenced reach was improved by cattle exclusion. It is possible that improved riparian health may have increased the filtering or buffer capacity of the fenced reach for surface runoff from the adjacent land or water flowing in the river. Miller et al. (2008a) reported that the cattle-excluded riparian pasture within this fenced reach was acting as a buffer by increased canopy cover and standing litter, decreased bare soil and soil bulk density, and decreased runoff depth and mass loads of TN, TDN, TPN, and DRP.

Specific riparian health parameters that may have a greater potential to trap sediment and filter or buffer contaminants than others include vegetative cover of floodplain and streambanks (criterion #1 in Table 3), preferred tree and shrub establishment (criterion #4), streambank root mass protection (criterion #7), and stream channel incisement (criterion #11) (Fitch and Ambrose, 2003). We found that vegetation cover and root mass protection parameters were not increased by streambank fencing, but that preferred tree and shrub establishment was increased, and human-caused bare ground was decreased (Table 3). Preferred trees and shrubs such as *Populus deltoides* (cottonwood), *Populus tremuloides* (aspen), *Populus* spp. (poplar), *Salix* spp. (willows), and *Cornus stoloiferea* (red-osier dogwood) play an important role in riparian areas as their root systems are excellent bank and shoreline stabilizers, and they play a key role in the uptake of nutrients that could otherwise degrade water quality (Fitch and Ambrose, 2003). Preferred woody plant communities such as willows were not well established within the riparian area after streambank fencing. Only three shrub species were observed growing here: *Symphoricarpos occidentalis* (snowberry), *Rosa woodsii* (common wild rose), and *Atriplex nuttallii* (Nuttall's atriplex). Snowberry and common wild rose are not considered preferred woody plants as their root systems are not as efficient in binding riverbanks together compared to willows. These two species are also resistant to grazing and tend to increase in abundance in heavy grazing situations. *Nuttall's atriplex*, although more desirable than snowberry and common wild rose, still does

Table 6
Correlations between water quality variables (concentrations) at CON and BMP sites in the Lower Little Bow River from 2004 to 2007.

	pH	EC	DO	Turb	Temp	TN	TP	TSS	Fecol	Ecol	Flow	Chlor-a
pH	–											
EC	0.26*	–										
DO	0.05	0.09	–									
Turb	−0.19*	−0.06	−0.05	–								
Temp	0.04	−0.24*	−0.33*	−0.07	–							
TN	−0.42*	−0.34*	−0.05	0.67*	−0.01	–						
TP	−0.30*	−0.06	−1.10	**0.76***	0.02	**0.82***	–					
TSS	−0.15*	−0.07	−0.10	**0.92***	−0.003	0.66*	**0.71***	–				
Fecol	−0.08	−0.37*	−0.21*	0.15*	0.49*	0.53*	0.57*	0.27*	–			
Ecoli	−0.38*	−0.11	−0.21*	0.57*	0.19*	**0.81***	**0.84***	0.46*	**0.97***	–		
Flow	−0.09	0.18*	−0.07	0.006	0.16*	0.13	0.25*	−0.10	0.16	0.26*	–	
Chlor-a	−0.07	0.50*	0.11	0.42*	−0.34*	0.59*	0.37*	0.50*	−0.30*	−0.34*	−0.14	–

Strong ($r \geq 0.70$) are shown in bold.
* Correlations are significant at $P \leq 0.05$.

not provide the benefits of taller, deep-rooted shrubs such as willows. *Nuttall's atriplex* was found to be naturally regenerating within the riparian area and therefore contributed to the improved health score for this factor.

Human-caused bare ground is unprotected soil that results from human activities such as livestock grazing, and bare ground represents a loss of vegetation to filter and buffer sediment, resulting in more water and wind erosion (Fitch and Ambrose, 2003). We attributed the reduction in bare ground in the fenced riparian area to the exclusion of cattle. Much of the reduction in bare ground was along the streambank, indicating that exclusion of cattle may reduce streambank erosion. Previous researchers have reported reduced streambank erosion due to cattle exclusion (Kauffman et al., 1983; Platts and Nelson, 1985; Trimble, 1994; Zaimes et al., 2005). However, other researchers have reported no significant effect of streambank fencing on streambank erosion (Kondolf, 1993; Allen-Diaz et al., 1998; George et al., 2002). Contrasting findings may be due to the large number of factors that control the rate of bank erosion: composition of bed materials, streamflow patterns and amounts, soil moisture, frost action, channel geometry, vegetation type and cover, and activity of burrowing animals (Knighton, 1984).

No significant differences in the concentrations and loads for the majority of water quality variables (Table 4) and frequency of water quality violations (Table 5) at the upstream and downstream sites was evidence that streambank fencing generally prevented water pollution downstream (Scenario 4 in Fig. 3). The dramatic improvement in riparian health after streambank fencing clearly suggests that cattle were indeed degrading the reach prior to fencing.

However, a few water quality variables were significantly decreased or increased downstream during this study. Turbidity in 2004 and 2005, and Cl concentration in 2006 and 2007, were significantly lower downstream than upstream. It is possible that suspended sediment in the river may have settled out from the upstream to downstream site, or some filtering of sediment may have occurred along the re-vegetated streambank, lowering turbidity values downstream. Lower Cl concentrations downstream may have been due to dilution effects caused by groundwater discharge or surface runoff of water through erosion channels into the river. Turbidity in 2006, DO in 2004, 2006 and 2007, TP concentration in 2005 and 2006, TP load in 2005, *E. coli* concentration in 2004 and 2005, and *E. coli* load in 2004 significantly increased downstream. This suggested that the BMP was not effective at preventing water pollution for these variables (Scenario 3 in Fig. 3). It is also possible that surface runoff in erosion channels that discharge into this reach may have increased the turbidity, TP, and *E. coli* downstream. In addition, wildlife such as birds (e.g. ducks) and muskrats depositing fecal material in the fenced reach may have also contributed to higher *E. coli* levels downstream, as wildlife may have been attracted to the improved riparian habitat of the fenced reach. We are unsure as to why DO would increase downstream, since it is mainly controlled by water temperature, and there was no similar trend in this latter variable. There was no increase in riffles or rapids downstream within the fenced reach that would cause increased DO levels. Line (2003) reported that DO, pH and temperature in relatively short reaches of flowing stream are rarely significantly affected by non-point source pollution.

Water samples collected upstream (LB4-14U) and downstream (LB4-14) of the cattle crossing in 2006 and 2007 were compared using a paired *t*-test to determine if the cattle crossing and cattle activity at this crossing had a significant ($P \leq 0.10$) influence on water quality at the downstream site. Statistical analyses revealed no significant differences in the water quality variables analyzed (Cl, TN, TP, TSS, temperature, DO, turbidity, chlorophyll-a). Water samples at LB4-14U were not analyzed for indicator bacteria. These results indicated that cattle activity and fecal deposition at the cattle crossing and between these two stations had no influence on the measured water quality variables at the downstream BMP-impact station (LB4-14). However, it is still possible that cattle wandered further upstream in the river and may have contributed fecal material to the river. Approximately 95% of deposited manure will settle to the bottom of the stream within the first 50 m (Biskie et al., 1988); and the bacteria in the sediment may remain alive for several weeks (Sherer et al., 1992). Less is known about what happens to the nutrients that enter the stream in manure in terms of their residual influence on water quality and how far downstream they affect water quality. In comparison, Henry and Thurston-Enriquez (2003) found that a cattle crossing in Nebraska increased fecal bacteria and protozoa downstream, and they attributed this to the presence of cattle in the unrestricted pasture upstream.

We found that streambank fencing prevented degradation of the majority of water quality variables downstream along the 800-m reach of the LLB River. Line et al. (2000) used a fenced reach of only 335 m and still found significant BMP effects on water quality within their reach in North Carolina. In our study, concentrations and values of seven water quality variables (Cl, TN, temperature, fecal coliforms, *E. coli*, DO, flow) at the upstream and downstream sites during 2004–2007 were very strongly ($r \geq 0.90$) correlated, two variables were moderately ($0.50 \leq r < 0.70$) correlated (turbidity, TSS), and one variable (TP) was weakly ($r < 0.50$) correlated. Strong to very strong correlations for the majority of our water quality variables (concentrations) indicated that our reach was of sufficient length to elicit a BMP response downstream for most variables. Shirmohammadi et al. (1997) suggested that significant and strong correlations in water quality variables between upstream and downstream sites may be used to determine if the distance between upstream and downstream sites is sufficient to elicit a BMP response downstream. They attributed poor correlations between upstream and downstream sites to dilution and possible instream attenuation that may mask possible BMP effects.

Our finding of no strong correlations between the water quality variables and river flow (Table 6) was in contrast to previous findings (McKergow et al., 2003). A great deal of the variance in a water quality variable is due to river discharge, and is generally due to dilution or wash-off (Hirsch et al., 1991). Negative correlations of water quality variables with flow indicate dilution and positive correlations indicate wash-off of sediment bound contaminants from overland flow or streambank erosion. We attributed the lack of correlation of our water quality variables with river flow to the regulated nature of the LLB River, as river flows are controlled by releases from the upstream irrigation reservoir, and are also affected by irrigation return flow and withdrawal of water for irrigation. Although the lack of positive correlations between water quality variables and river flow suggests little wash-off of sediment bound contaminants from overland flow or streambank erosion, we visually observed surface runoff between the upstream and downstream sites during this study after major precipitation events, and rainfall simulation runoff was also generated from the cattle excluded and grazed native pastures between these sites adjacent to the fence on the south side of the river during this study (Miller et al., 2008a).

We did not find differences in water quality responses (Table 4) at the downstream and upstream sites among years, particularly between the wetter year of 2005 and the other drier years in 2004, 2006, and 2007. The BMPs implemented on land would be expected to have a greater effect on water quality in wetter than drier years because of increased transport of contaminants from the land to water during wetter years (Little, 2001). The latter author also reported that climate had a strong effect on water quality in the LLB River, with better water quality in drier years with less runoff. It should also be noted that cattle will frequent the river and riparian area more often in drier than wetter years, so the potential for water pollution (P) may be greater in drier years (Scenario 2 in Fig. 3).

We found that streambank fencing prevented degradation for the majority of water quality variables in the semi-arid region of western Canada. Previous studies in the more humid region of the eastern USA have reported reduced water pollution from streambank fencing using other study designs (Owens et al., 1996; Line et al., 2000; Meals, 2001; Line, 2003; Muenz et al., 2006). However, some studies have reported little or no reduction in water quality after streambank fencing (McKergow et al., 2003; Ranganath et al., 2009). The major advantage of our study design is that the BMP can be evaluated in a much shorter time period, since a pre-BMP phase is not required. Other researchers have also evaluated BMPs using data for the post-BMP period with no pre-BMP data (Edwards et al., 1997). As noted by Jackson and Morris (2002), different BMP study designs all have their advantages and disadvantages, and the research community should continue to pursue a mix of study designs.

Our finding that streambank fencing improved riparian health and prevented water pollution downstream was similar to the findings of Ranganath et al. (2009). They also found improvements in riparian health of a stream in western Virginia, but found no significant differences in benthic macroinvertebrates or water quality variables between the cattle-access and cattle-excluded reaches. They noted that improvements in these instream variables may depend more on upstream watershed-scale conditions and impacts rather than localized reach-scale livestock-access issues.

Prevention of water pollution in our fenced reach may have been due to elimination of direct fecal defecation into the stream, reduced runoff of fecal material from the adjacent land, reduced erosion of streambanks by cattle, and/or a decrease in re-suspension of river sediments by cattle trampling. Miner et al. (1992) concluded that since runoff events occur less than one percent of the time in the arid western U.S.A., that direct fecal deposition by cattle in streams and rivers was dominant for over 99% of the time. However, on cattle access reaches in our study area, we observed considerable runoff from cattle-eroded bare soil adjacent to the river that had accumulations of fecal pats, and the runoff was often tea-coloured, indicating that contaminants from the fecal material were being washed off these exposed banks into the river by runoff. Larsen (1995) estimated manure loading into a stream from cattle and concluded that the contribution from direct fecal deposition was quite small. However, he cautioned that water pollution could still occur since daily inputs from fecal loading may accumulate on the stream bottom and be resuspended by any disturbance such as high flows or cattle trampling. Cattle shearing of streambanks may contribute to increased streambank erosion and water pollution of streams, and seems to be more susceptible in streams located in finer-textured soils and that are subject to high-intensity flows with more erosive potential (Trimble and Mendel, 1995). Our river was located in medium to moderately coarse-textured soils, indicating a lower potential for streambank erosion based only on soil texture.

Although flows in the LLB River are relatively stable since they are controlled by the dam upstream, irrigation return flows can cause high intensity flows in the spring to fall period. Streambank fencing can significantly reduce streambank erosion (Kauffman et al., 1983; Platts and Nelson, 1985; Trimble, 1994; Zaimes et al., 2005). Zaimes et al. (2005) found that for streambank erosion, soil loss was 34 times lower, and phosphorus loss was 82 times lower for pastures with streambank fencing compared to continuous grazed pastures with cattle access to rivers. Although we did not measure streambank erosion in our study, the riparian health assessment found that human-caused bare ground was reduced after streambank fencing, particularly along the streambank (Table 3). This suggests that reduced streambank erosion on our reach may have been a factor in preventing water quality pollution of this reach.

5. Conclusions

We accept the hypothesis that riparian health (vegetative, soil, and hydrologic features) was improved by streambank fencing. An increase in preferred tree and shrub establishment and a reduction in human-caused bare ground have the potential to increase the buffering or filtering capacity of this fenced reach for surface runoff and water flow in the river.

We accept the hypothesis that streambank fencing prevented water quality degradation for the majority of water quality variables at the downstream site. The riparian health assessment found reduced human-caused bare ground from the pre- to post-BMP phase, particularly along the streambank, suggesting that prevention of pollution may have been enhanced by reduced streambank erosion. Fencing of cattle also eliminated wash-off of fecal material accumulating on eroded streambanks. The cattle crossing in our study did not contribute to water pollution downstream. Significant positive correlations in most water quality variables between the upstream and downstream sites indicated that the 800-m reach was likely of sufficient length to assess this BMP. Water quality variables and concentrations of sediment, N, P, and bacteria were not correlated with river flow, and were attributed to the regulated nature of this river.

Further research on this BMP may involve installing datasondes and automatic water samplers to sample the river water quality more continuously and during storm events, sampling benthic aquatic insects, sampling suspended sediment in addition to water, and continuing riparian health assessments. In addition, further research is required to determine if streambank erosion is greater for this fenced reach than adjacent unfenced reaches.

Acknowledgements

The assistance of the landowners (County of Lethbridge) and producers (Charles Sorgaard) in this study are gratefully acknowledged. The streambank fencing was installed by Alberta Agriculture and Rural Development (ARD) during the Oldman River Basin Water Quality Initiative. Stilling well installation and flow monitoring in the LLB River was provided by ARD. The advice of the WEBs committee members (Dennis Lastuka, Murray Lewis, Shane Petry, Dwayne Rogness, Carlyle Ross, Kim Schmitt, and Elwin Smith) are gratefully acknowledged. Laboratory analyses of water samples were conducted by Ki Au, Gyan Mankee, Dana Coles, and Mark Kadijk of ARD. Field and laboratory assistance was provided by the following students: Raygan Boyce, Julie Bunker, Cayley Dyck, Teneal Doram, Michael Gretzinger, Kelly Holowka, Theresa Kamphuis, Jana Hamilton, and Peter Pfitchar. The contribution of the former committee members (Troy Ormann and Gary Kindrat) was much appreciated. The riparian health assessment was conducted by Cows and Fish, and the assistance of Noreen Ambrose was appreciated. Administrative assistance was provided by Brook Harker, Terrie Scott, Valerie Stuart, Lucy Clearwater, Glenda Meyer, Leanne Oschanney and Cori Hofford with AAFC-PFRA. Funding for this WEBs project was from

Greencover Canada (AAFC-PFRA) and Ducks Unlimited, and is gratefully acknowledged.

References

Adams, B.W., Poulin-Klein, L., Moisey, D., 2005. Range Plant Communities and Range Health Assessment Guidelines for the Mixedgrass Natural Subregion of Alberta. Alberta Sustainable Development, Rangeland Management Branch, Public Lands and Forests Division, Lethbridge, AB.

Agouridis, C.T., Workman, S.R., Warner, R.C., Jennings, G.D., 2005. Livestock grazing management impacts on stream water quality: a review. J. Am. Water Resour. Assoc. 41 (3), 591–606.

Alberta Environment, 1999. Surface Water Quality Guidelines for Use in Alberta. Environ. Science Division, Alberta Environment, Edmonton, AB.

Allen-Diaz, B., Jackson, R.D., Fehmi, J., 1998. Detecting channel morphology change in California's hardwood rangeland spring ecosystems. J. Range Manage. 51 (5), 514–518.

American Public Health Association (APHA), 1998. Standard Methods for the Examination of Water and Wastewater. 20th ed. APHA, Washington, DC.

American Society for Testing and Materials (ASTM), 2006. Test Method for Total Chemically Bound Nitrogen in Water by Pyrolysis and Chemiluminescence Detection. In: Annual Book of ASTM Standards 11.02. American Society for Testing and Materials, Philadelphia, PA.

Belsky, A.J., Matzke, A., Uselman, S., 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. J. Soil Water Conserv. 54 (1), 419–431.

Biskie, H.A., Sherer, B.M., Moore, J.A., Miner, J.R., Buckhouse, J.M., 1988. Fate of organisms from manure point loading into a rangeland stream. ASAE Paper 88-2081. ASAE, ST. Joseph, MI.

British Columbia Ministry of Environment, 2008. Ambient Water Quality Guidelines (criteria) for Turbidity, Suspended and Benthic Sediments. Canadian Council of Ministers of the Environment (CCME) 2002. Canadian Environmental Quality Guidelines. Update 2002. CCME, Winnipeg, 1; MB. Accessed on March 5, 2009, http://www.env.gov.bc.ca/wat/wq/BCguidelines/turbidity/turbidity.html.

Davis, L., Brittingham, M., Garber, L., Rourke, D., 1991. Streambank Fencing. Extension Circular 397. Pennsylvania State Univ. College of Agr. Sci., University Park, PA.

Edwards, D.R., Daniel, T.C., Scott, H.D., Moore Jr., P.A., Murdoch, J.F., Vendrell, P.F., 1997. Effect of BMP implementation on storm flow quality of two northwestern Arkansas streams. Trans. ASAE 40 (5), 311–1319.

Environment Canada, 1996. Phosphorus, Total. Envirodata Method Code 582 (VMV Code 15421). Oct. 1, 1996, Environment Canada, Ottawa, ON.

Environment Canada, 2007a. Total P by Acidic Persulfate Digestion, Automated Continuous Segmented Flow Analyzer (CFA), Stannous chloride-molybdate Complex, Photometric Method. ENIVRODAT Method 2331 (VMV Code 15423). Accessed on March 5, 2009, http://map.ns.ec.gc.ca/ResEau/en/vmv.aspx.

Environment Canada, 2007b. Total Dissolved P by Acidic Persulfate Digestion, Automated Continuous Segmented Flow Analyzer (CFA), Stannous chloride-molybdate Complex, Photometric Method. ENIVRODAT Method 2332 (VMV Code 15465). Accessed on March 5, 2009, http://map.ns.ec.gc.ca/ResEau/en/vmv.aspx.

Fitch, L., Adams, B.W., 1998. Can cows and fish co-exist? Can. J. Plant Sci. 78, 191–198.

Fitch, L., Adams, B.W., Hale, G., 2001. Riparian Health Assessment for Streams and Small Rivers-field Workbook. Lethbridge, AB., Cows and Fish Program. 90 pp. Accessed on September 1, 2009, http://www.cowsandfish.org/pdfs/Streams-FieldWkbk2005.pdf.

Fitch, L., Ambrose, N., 2003. Riparian Areas: A User's Guide to Health. Cows Fish Program, Lethbridge, Alberta, ISBN No. 0-7785-2305-5.

Galeone, D.G., 1999. Calibration of paired basins prior to streambank fencing of pasture land. J. Environ. Qual. 28, 1853–1863.

George, M.R., Larsen, R.E., McDougald, N.K., Tate, K.W., Gerlach Jr., J.D., Fulgham, K.O., 2002. Influence of grazing on channel morphology of intermittent streams. J. Range Manage. 55 (6), 551–557.

Health Canada, 1992. Guidelines for Canadian Recreational Water Quality. Minister of Supply and Services. Ottawa, ON. Cat. H49-70/1991E. ISBN 0-660r-r14239-2.

Henry, C., Thurston-Enriquez, J., 2003. Impact of a constructed low water stream crossing, and innovative BMP, on microbial water quality of a rangeland stream. Paper Presented at American Society of Agricultural Engineers (ASAE) Annual International Meeting, Las Vegas, NV, July 27–30, Paper No. 032313, St. Joseph, MI.

Hirsch, R.M., Alexander, R.B., Smith, R.A., 1991. Selection of methods for the detection and estimation of trends in water quality. Water Resour. Res. 27, 803–813.

Jackson, C.R., Morris, L., 2002. Methodological issues in BMP effectiveness studies. Paper Presented at Foresty Best Management Practices Research Symposium, Atlanta, Georgia, April 15–17, 2002. Accessed on March 5, 2009, http://www.trout.forprod.vt.edu/meetings/presentations/jackson.pdf.

Kauffman, J.B., Krueger, W.C., Varva, M., 1983. Impacts of cattle on streambanks in northeastern Oregon. J. Range Manage. 36, 683–685.

Kauffman, J.B., Krueger, W.C., 1984. Livestock impacts on riparian ecosystems and streamside management implications: a review. J. Range Manage. 37, 430–438.

Knighton, D., 1984. Fluvial Forms and Processes. Edward Arnold, London.

Kondolf, G.M., 1993. Lag in stream channel adjustment to livestock exclosure, White Mountains, California. Restor. Ecol. 1 (4), 226–230.

Larsen, R.E., Miner, J.R., Buckhouse, J.C., Moore, J.A., 1994. Water-quality benefits of having cattle manure deposited away from streams. Bioresource Tech. 48, 113–118.

Larsen, R.E., 1995. Fact sheet No. 25: manure loading into streams from direct fecal deposits. University of California Cooperative Extension, San Bernardino County, CA. Accessed on May 4, 2009, http://danr.ucop.edu/uccelr/h25.htm.

Line, D.E., Harman, W.A., Jennings, G.D., Thompson, E.J., Osmond, D.L., 2000. Non-point-source pollutant load reductions associated with livestock exclusion. J. Environ. Qual. 29, 1882–1890.

Line, D.E., 2003. Changes in a stream's physical and biological conditions following livestock exclusion. Trans. ASAE 46 (2), 287–293.

Little, J., 2001. Water Quality in the Battersea Drain and Lower Little Bow River (1998-2000)-Interim Report. Irrigation Branch, Alberta Agriculture, Food and Rural Development, Lethbridge, AB.

McKergow, L.A., Weaver, D.M., Prosser, I.P., Grayson, R.B., Reed, A.E.G., 2003. Before and after riparian management: sediment and nutrient exports from a small agricultural catchment, Western Australia. J. Hydrol. 270, 253–272.

Meals, D.W., 2001. Water quality response to riparian restoration in an agricultural watershed in Vermont, USA. Water Sci. Tech. 43, 175–182.

Mesner, N., Walker, A., Paige, G., 2008. Best management practices monitoring guidance document for stream systems. Working Draft Paper. April, 2008. College of Natural Resources, Utah State University, UT.

Miller, J., Chanasyk, D., Curtis, T., Lastuka, D., Lewis, M., Petry, S., Rogness, D., Ross, C., Schmitt, K., Smith, E., Willms, W., 2008a. Influence of cattle exclusion on vegetative and soil properties, rangeland health, and rainfall simulation runoff. In: Miller, J. (Ed.), Watershed Evaluation of Beneficial Management Practices in the Lower Little Bow Watershed of Southern Alberta. Final WEBs (Watershed Evaluation of Beneficial Management Practices) Report. Agriculture and Agri-Food Canada, Lethbridge, AB. pp. 2-1–2-39.

Miller, J., Chanasyk, D., Curtis, T., Lastuka, D., Lewis, M., Petry, S., Rogness, D., Ross, C., Schmitt, K., Smith, E., Willms, W., 2008b. Influence of Off-Stream Watering with No Fencing on Quantity and Quality of Rainfall Simulation Runoff Adjacent to Lower Little Bow River. In: Miller, J. (Ed.), Watershed Evaluation of Beneficial Management Practices in the Lower Little Bow Watershed of Southern Alberta. Final WEBs (Watershed Evaluation of Beneficial Management Practices) Report. Agriculture and Agri-Food Canada, Lethbridge, AB. pp. 3-1–3-64.

Miner, J.R., Buckhouse, J.C., Moore, J.A., 1992. Will a water trough reduce the amount of time hay-fed livestock spend in the stream (and therefore improve water quality)? Rangelands 14 (1), 35–38.

Mostaghimi, S., Brannan, K.M., Dillaha III, T.A., Bruggeman, A.C., 2001. Best management practices for nonpoint source pollution control: selection and assessment. In: Ritter, W.F., Shirmohammadi, A. (Eds.), Agricultural Nonpoint Source Pollution-Watershed Management and Hydrology, Lewis Publ., Boca Raton, FL, pp. 257–304.

Muenz, T.K., Golladay, S.W., Vellidis, G., Smith, L.L., 2006. Stream buffer effectiveness in an agriculturally influenced area, southwestern Georgia: responses of water quality, macroinvertebrates, and amphibians. J. Environ. Qual. 35, 1924–1938.

Novotny, V., Olem, O.H., 1994. Water Quality: Prevention, Identification, and Management of Diffuse Pollution. Van Nostrand Reinhold, NY.

Owens, L.B., Edwards, W.M., van Keuren, R.W., 1996. Sediment losses from a pastured watershed before and after stream fencing. J. Soil Water Conserv. 51 (1), 90–94.

Platts, W.S., Nelson, R.L., 1985. Stream habitat and fisheries response to livestock grazing and instream improvement structures, Big Creek, Utah. J. Soil Water Conserv. 40 (4), 374–379.

Ranganath, S.C., Hession, W.C., Wynee, T.M., 2009. Livestock exclusion influences on riparian vegetation, channel morphology, and benthic macroinvertebrates assemblages. J. Soil Water Conserv. 64 (1), 33–42.

Sarr, D.A., 2002. Riparian livestock exclosure research in the western United States: a critique and some recommendations. Environ. Manage. 30 (4), 516–526.

SAS Institute Inc., 2005. SAS OnlineDoc 9.1.3. SAS Institute Inc., Cary, NC.

Sherer, B.M., Miner, J.R., Moore, J.A., Buckhouse, J.C., 1992. Indicator bacterial survival in stream sediments. J. Environ. Quality 21, 591–595.

Shetson, I., 1987. Quaternary Geology, Southern Alberta. Alberta Research Council, Edmonton, AB.

Shirmohammadi, A., Yoon, K.S., Magette, W.L., 1997. Water quality in mixed land-use watershed-piedmont region in Maryland. Trans. ASAE 40 (6), 1563–1572.

Strahler, A.N., 1952. Dynamic basis of geomorphology. Geol. Soc. Am. Bull. 63, 923–938.

Technicon Industrial Systems, 1986a. Nitrite. TRAACS 800 Methods, Bran and Luebbe Industrial Method no. 784-86T. Tarrytown, NY.

Technicon Industrial Systems, 1986b. Nitrogen, Total Kjeldhal. TRAACS 800 methods, Bran and Luebbe Industrial Method no. 786-86T. Tarrytown, NY.

Technicon Industrial Systems, 1986c. Phosphorus, Total. TRAACS 800 methods, Bran and Luebbe Industrial Method no. 787-86T. Tarrytown, NY.

Technicon Industrial Systems, 1987a. Ammonia in water and wastewater. TRAACS 800 methods, Bran and Luebbe Industrial Method no. 780-86T. Tarrytown, NY.

Technicon Industrial Systems, 1987b. Nitrate. TRAACS 800 methods, Bran and Luebbe Industrial Method no. 782-86T. Tarrytown, NY.

Technicon Industrial Systems, 1987c. Ortho Phosphorus. TRAACS 800 methods, Bran and Luebbe Industrial Method no. 781-86T. Tarrytown, NY.

Tossell, R.W., Dickinson, W.T., Rudra, R.P., Wall, G.J., 1987. A portable rainfall simulator. Can. Agric. Eng. 29, 155–162.

Trimble, S.W., 1994. Erosional effects of cattle on streambanks in Tennessee, USA. Earth Surf. Proc. Land. 19, 451–464.
Trimble, S.W., Mendel, A.C., 1995. The cow as geomorphic agent-a critical review. Geomorphology 13, 233–253.
Vidon, P., Campbell, M.A., Gray, M., 2008. Unrestricted cattle access to streams and water quality in till landscape of the Midwest. Agr. Water Manage. 95, 322–330.
Wroe, R.A., Smoliak, S., Adam, B.W., Willms, W.D., Anderson, M.L., 1988. Guide to Range Condition and Stocking Rates for Alberta Grasslands 1988. Public Lands, Alberta Forestry, Lands and Wildlife, Edmonton, AB.
Zaimes, G.N., Schultz, R.C., Isenhart, T.M., Mickelson, S.K., Kovar, J.L., Russell, J.R., Powers, W.P., 2005. Stream bank erosion under different riparian land-use practices in northeast Iowa. Accessed on April 29, 2009, http://www.cinram.umn.edu/afta2005/pdf/zaimes.pdf.