DOI: 10.1007/s00267-001-0054-7

Consequences of Livestock Grazing on Water Quality and Benthic Algal Biomass in a Canadian Natural Grassland Plateau

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ABSTRACT / The effects of livestock grazing on selected riparian and stream attributes, water chemistry, and algal biomass were investigated over a two-year period using livestock enclosures and by completing stream surveys in the Cypress Hills grassland plateau, Alberta, Canada. Livestock enclosure experiments, partially replicated in three streams, comprised four treatments: (1) early season livestock grazing (June–August), (2) late season livestock grazing (August–September), (3) all season grazing (June–September), and (4) livestock absent controls. Livestock grazing significantly decreased streambank stability, biomass of riparian vegetation, and the extent to which aquatic vegetation covered the stream channels compared with livestock-absent controls. Water quality comparisons indicated significant differences among the four livestock grazing treatments in Battle and Graburn creeks but not in Nine Mile Creek. In Graburn Creek, the concentration of total phosphorus in the all-season livestock grazing treatment was significantly higher than that in the livestock-absent control, and the early season and late season grazing treatments. Concentrations of soluble reactive phosphorus in the all-season livestock grazing treatment also exceeded that in livestock-absent control. In contrast, differences in water quality variables in the remaining 22 comparisons (i.e., 22 of the total 24 comparisons) were minor even when differences were statistically significant. Effects of livestock grazing on algal biomass were variable, and there was no consistent pattern among creeks. At the watershed scale, spatial variation in algal biomass was related ($P < 0.05$) with concentrations of $NO_2^- + NO_3^-$ and soluble reactive phosphorus in two of the four study creeks. Nutrient diffusing substrata experiments showed that algal communities were either nitrogen-limited or not limited by nutrients, depending on stream and season.

Agricultural activities can profoundly affect water quality by increasing concentrations of nitrogen and phosphorus and, through trophic cascades, can increase the biomass of benthic algae (Sharpely and others 1994, Quinn and others 1997, Chambers and others 2001). The effects of livestock grazing, and the need for intact riparian vegetation to reduce nutrient and sediment inputs to streams, have been well known for at least a decade (McColl 1978, Platts and Nelson 1985, Armour and others 1991). However, the effects of livestock grazing on stream ecosystems can be relatively variable depending on whether grazing is associated with radical changes in land use (e.g., conversions of forests to croplands or pasture) (Lal 1997, Cooke and Prepas 1998), and differences in watershed attributes (e.g., local and regional hydrology, geology, soil type) (Quinn and others 1997), livestock grazing practices (e.g., stock densities, grazing period, access of livestock to stream channels), and the availability of off-site water supplies and salt-licks (Godwin and Miner 1997, Sheffield and others 1997). Thus, while resource managers may be aware of the largely negative effects of livestock grazing on stream ecosystems, the absence of site-specific information precludes the development of best management practices that are suited to local physiographic conditions.

Livestock grazing has been a predominant land use within the Cypress Hills grassland plateau, Alberta, Canada, for more than a century (Lewis and Johnson 1980) and is currently authorized through agreements between livestock grazing associations and the provincial government. While the overall stocking rate based on total grassland area is relatively moderate (0.47–0.59 animal unit months), livestock utilize riparian areas disproportionately during the summer and fall periods (June–September). These grazing practices create riparian areas that are largely devoid of willow and aspen, grass communities that typically comprise short stubble (e.g., <10 cm), and highly unstable streambanks. These observations, combined with recent changes in the management of Alberta's provincial parks, have

KEY WORDS: Agriculture; Livestock; Water quality; Riparian; Algae

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raised questions about the effects of livestock grazing on water quality and overall stream health (Alberta Environmental Centre 1995).

The objectives of our study were twofold. First, over a 16-month period (May 1999–October 2000) we evaluated the effects of livestock grazing on selected riparian and stream attributes, water quality, algal biomass, and nutrient limitation status using experimental enclosures and by completing longitudinal and stream surveys. Experimental enclosures provided an opportunity to compare the effects of: (1) early season livestock grazing (June–August), (2) late season livestock grazing (August–September), (3) all season grazing (June–September), and (4) livestock-absent controls on total phosphorus (TP), soluble reactive phosphorus (SRP), total nitrogen (TN), nitrate + nitrite, ($NO_2^- + NO_3^-$), dissolved organic carbon (DOC), dissolved oxygen (DO), conductivity, turbidity, and benthic algal biomass. We also quantified concentrations and daily loadings of SRP and $NO_2^- + NO_3^-$ from small tributary springs to stream reaches within enclosures to determine whether differences among livestock treatments likely resulted from the effects of livestock grazing or groundwater inputs. Longitudinal patterns in SRP, $NO_2^- + NO_3^-$ and algal biomass were quantified in four streams to evaluate whether predicted increases in SRP and $NO_2^- + NO_3^-$ resulting from livestock grazing are important in determining watershed-scale variation in SRP, $NO_2^- + NO_3^-$ and epilithic biomass. We predicted that livestock grazing would result in detectable effects on water quality by increasing N and P concentrations, water turbidity, and reducing bank stability and riparian and instream vegetation cover. Second, we use the results to evaluate the need to develop alternative livestock grazing management practices in the Cypress Hills.

Methods

Study Area

Vegetation and precipitation. The Cypress Hills (maximum elevation 1466 m above sea level [masl], area 1820 km^2), located 45 km southeast of Medicine Hat, Alberta, Canada, is a nonglaciated remnant of an extensive depositional plain that covered much of North America about 30 million years ago (Coleman 1968). The majority of the Cypress Hills consists of an extensive grassland plateau and plateau slopes located in Alberta and Saskatchewan. These slopes extend off the plateau and become part of an extensive area of grassland plains. In Alberta, approximately 30% of the Cypress Hills were designated as a provincial park (i.e., Cypress Hills Provincial Park) in 1951.

In contrast to the surrounding plains, the Cypress Hills plateau was not reduced by the erosional action of the Laurentian glacier that eroded adjacent plains by about 500 m between 25,000 and 1 million years ago. As a result, the Cypress Hills are underlain by Oligocene (Cypress Hills formation: 30 million years), Paleocene (60–70 million years) and Upper cretaceous formations (70–1000 million years) (Tokarsky 1985). The Cypress Hills Formation (>1350 masl) consists of quartzite and chert gravel and a limestone conglomerate, interbedded with sandstone, silts, and clays. The predominance of moderately fine and textured gravel and cobble formations results in moderately to well-drained soil conditions and a preponderance of groundwater springs.

The Cypress Hills is part of the Mixed grass and Montane ecoregions (Strong and Leggat 1992). Annual precipitation within the Mixed grass ecoregion (326 mm) is lower than that on the Montane ecoregion, where about 440 mm fall annually (Strong and Leggat 1992). Summer rainfalls typically result from local thunderstorms (Coleman 1968) and mean daily summer (May–August) and winter (November–February) temperatures typically range from about 12 to 15°C and −5 to −6°C, respectively. Daily minimum temperatures during the winter often range from −5 to −20°C. These conditions result in moderate potential evaporation rates and an annual water deficit of about 100 mm (Coleman 1968).

Vegetation on the plateau slopes is comprised predominantly by a native rough fescue (*Festuca campestris*)–shrubby cinquefoil (*Potentilla fruticosa*) complex with lodgepole pine (*Pinus contorta*), white spruce (*Picea glauca*), balsam poplar (*Populus balsamifera*), and trembling aspen (*Populus tremuloides*), underlain by dark brown or black chernozems and gleysols. At higher elevations of 1200–1300 masl, grasslands are underlain with dark brown chernozems and eutric brunisols with trembling aspen, balsam poplar, lodgepole pine, and white spruce on gray luvisols dominant at elevations above 1300 masl (Greenlee 1980, Looman and Best 1987). While detailed surveys of soils were not undertaken, soils within riparian areas included dark brown chernozems, eutric brunisols, and gray luvisols.

Streams. Streams in the Cypress Hills originate from springs located in moderately to well-defined coulees adjacent to the grassland plateau. In upper reaches where livestock enclosures were established. Substratum is dominated by large cobbles and gravels with stream channel widths and depths ranging from about 1–4 m and 0.2–1 m, respectively. At greater distances downstream, streams increase in size (width and depths

1–5 m and 0.5–1.5 m) and watershed vegetation is dominated by mixed conifer–deciduous forest.

Streams are circumneutral or slightly alkaline, hard, with relatively low concentrations of dissolved carbon. Water clarity is generally high with concentrations of dissolved oxygen, NO_2^- + NO_3^- and SRP in summer and fall ranging from 4 to 8 mg/liter, 100 to 400 μg/liter, and 20–40 μg/liter, respectively.

Fieldwork. Fieldwork was completed in the four adjacent watersheds of Battle (watershed area 192 km^2), Graburn (153 km^2), Nine Mile (70 km^2), and Storm (59 km^2) creeks located in the Cypress Hills Provincial Park. Effects of livestock grazing on selected stream and riparian attributes, water quality, and epilithon were evaluated by establishing livestock enclosures in Battle, Graburn, and Nine Mile creeks combined with longitudinal stream surveys. These surveys also included sampling in Storm Creek, an adjacent watershed that comprised an upstream reach heavily grazed by livestock (i.e., cow–calf pairs), and a downstream reach that was not grazed between October 1998 and September 2000. The downstream reach also historically experienced relatively low use by livestock, predominantly low densities of steers, compared to higher densities of cow–calf pairs in other study watersheds.

Livestock grazing. The Cypress Hills are grazed by livestock during the summer and fall periods (i.e., June–October) and are removed from the park to avoid the harsh winter conditions. To facilitate management, the Alberta Cypress Hills Provincial Park is divided into livestock grazing units. Stocking rates for each unit are determined by the Alberta Government Provincial Park staff and are based primarily on a stocking model that accounts for grass production and consumption. Evaluations of the ecological effects of current livestock grazing practices on stream health are not a dominant consideration in determining stocking rates. Commercial fertilizers are not applied to grasslands within the Cypress Hills Provincial Park.

During the summer–fall period, livestock graze riparian areas disproportionately because they provide high-quality forage, including clover species, and adjacent streams represent the primary water supply for livestock. While not quantified, livestock use riparian areas adjacent to the plateau grasslands, including those immediately downstream of livestock enclosures, to a greater extent compared with those located lower (10–20 km) in the watershed. However, riparian areas in lower reaches of watersheds are also grazed and do not differ markedly from those in upper reaches.

Livestock Enclosures

Design. Enclosures were established in the upper (i.e., headwater) reaches of all study streams (watershed areas above enclosures: Battle Creek 29 km^2, Graburn Creek 28 km^2, Nine Mile Creek 21 km^2) in May 1999 and, depending on treatment, contained livestock between June and September 1999 and 2000. Enclosures comprised irregular-shaped fenced areas that enclosed streams and adjacent riparian and upland areas (Table 1) and were designed to allow comparisons of the effects of four livestock grazing treatments: (1) no livestock grazing (livestock absent), (2) early season (primarily June–August) livestock grazing, (3) late season (primarily August–September) livestock grazing, and (4) all-season (predominantly June–September) livestock grazing (Table 1). Treatments were interspersed with buffer areas (i.e., additional livestock-absent areas) to minimize contamination of downstream sites by upstream livestock grazing treatments. For all streams, the livestock-absent treatment was located at the most upstream site, followed downstream by the early season grazing treatment, the late season grazing treatments and the all-season grazing treatment. While variable in width (i.e., the extent to which enclosures encompassed upland areas located perpendicular to stream flow: Battle Creek 60–550 m, Graburn Creek 60–400 m, Nine Mile Creek 60–550 m), treatments and buffer areas extended between 50–60 and 40–45 m of stream channel, respectively. All treatments were established in Battle and Graburn creeks with two treatments (i.e., the two livestock treatments of no livestock grazing and early season livestock grazing) established in Nine Mile Creek.

Depending on treatment, enclosures were stocked with six randomly selected crossbred yearling steers weighing 256–443 kg (565–975 lb). This number of animals, combined with enclosure areas and expected grass production estimates, ensured that animals would have sufficient forage to grow during the summer–fall period and produced stocking densities that resembled current grazing practices in the Cypress Hills (Korpela 2001).

Precipitation in the Battle, Graburn, and Nine Mile watersheds was estimated by monitoring rain gauges between May and September in 1999 and 2000 (Korpela 2001) combined with monthly precipitation data collected by the Alberta Environment (Government of Alberta) at the township of Elkwater located in the Cypress Hills. Stream discharge was typically measured monthly between June and October 1999 and May and October 2000 within all enclosures using a March McBirney Flowmate meter.

Table 1. Areas of livestock enclosure treatments and buffers and livestock stocking periods for Battle, Graburn, and Nine Mile creek enclosures, 1999 and 2000, Cypress Hills, Alberta

	Enclosure areas (ha)		
	Battle Creek	Graburn Creek	Nine Mile Creek
Buffer	0.59	0.51	0.87
No livestock grazing	0.53	0.43	0.68
Buffer	0.43	0.34	0.42
Early season grazing	3.40	2.99	3.31
Buffer	0.33	1.99	0.11
Late season grazing	3.61	3.02	-
Buffer	1.12	2.09	-
All-season grazing	5.19	5.57	-
Buffer	0.66	0.38	-
Year and grazing season	Livestock grazing periods		
	Battle Creek Period	Graburn Creek Period	Nine Mile Creek Period
1999			
Early season	10 June–9 Aug	8 June–13 Aug	24 June–13 Aug
Late season	9 Aug–29 Sept	13 Aug–6 Oct	
All-season	10 June–29 Sept	8 June–6 Oct	
2000			
Early season	12 June–15 Aug	12 June–11 Aug	9 June–13 Aug
Late season	16 Aug–25 Sept	12 Aug–25 Sept	
All-season	12 June–25 Sept	12 June–25 Sept	

Sampling of enclosures.

Streambanks, riparian, and instream channel characteristics. Effects of the livestock grazing treatments on streambank stability, riparian vegetation biomass, and instream vegetation cover within enclosures were evaluated by completing stream surveys in October 1998 and October 2000. Streambank stability was assessed visually in Battle, Graburn, and Nine Mile creeks in October 1998 (i.e., prior to establishment of the enclosures) and October 2000 (i.e., after the completion of the enclosure experiment). We systematically established 25 or 45 transects (lengths of stream channel were 5 m) along all study streams in October 1998 and quantified streambank stability along each 5-m channel reach by estimating the proportion of streambanks that contributed sediments to the stream channel (i.e., unstable banks) compared with those that did not (i.e., stable banks). For all streams, this resulted in ten estimates (five estimates from the left bank and five from the right bank) in all buffer and livestock treatments.

The extent to which livestock reduced the biomass of riparian vegetation located between 1 and 10 m from the stream channel was determined by clipping vegetation from five randomly selected replicate areas (0.1 m^2) within buffer and livestock enclosures in October 2000. Samples were dried to constant weight at 40°C and weighed to the nearest 0.1 mg. Biomass of riparian vegetation located between 10 and 40 m upstream and downstream of enclosures was determined using techniques described above.

Differences in instream vegetation cover among buffer and livestock treatments were quantified in October 2000 using the 5-m-long transects established to estimate streambank stability. For each transect, we estimated the proportion of stream channels (i.e., wetted areas) covered with vegetation, primarily grasses [brook grass (*Catabrosa aquatica*)], forbs [American brooklime (*Veronica americana*)], willowherb species (*Epilobium* spp.), yellow monkey-flower (*Mimulus guttatus*), common horsetail (*Equisetum arvense*), and sedges (*Carex* spp.).

Repeated measures ANOVA (RM-ANOVA) and ANOVA were used to test for differences in the proportion of banks that were stable, riparian vegetation biomass, and instream vegetation cover among buffer and livestock enclosures. These tests were completed separately for Battle, Graburn, and Nine Mile creeks. When ANOVA models were statistically significant, we compared differences among buffer and livestock grazing treatments with Bonferroni adjusted orthogonal contrasts on $\log_{10}$ (i.e., riparian biomass) or arcsin square root (i.e., percent stable banks, percent instream vegetation cover) transformed data. Statistical analyses were completed using SAS (1987).

Water chemistry. Water samples for physicochemical analyses (Table 2) were collected from buffer and live-

Table 2. Water quality variables, analytical techniques, and detection limits used to evaluate effects of livestock grazing on water quality in Cypress Hills streams[a]

Variable	Analytical method	Detection limit
Total nitrogen	Persulfate digestion (Ameel et al. 1993)	5 μg/liter
Nitrate and nitrite	Cadmium reduction method (Stainton and others 1977)	1 μg/liter
Total phosphorus	Modified (Prepas and Rigler 1982) potassium persulfate method (Menzel and Corwin 1965)	1 μg/liter
Soluble reactive phosphorus	Molybdate blue technique (Murphy and Riley 1962)	0.5 μg/liter
Dissolved organic carbon	Total organic carbon combustion infrared method (APHA 1992)	0.01 mg/liter
Turbidity	Hach turbidimeter model 2100A	0.2 NTU
Dissolved oxygen	WTW Multiline P4 portable meter fitted with a CellOx 325 probe	0.01 mg/liter
pH (pH)	Fisher Scientific Accumet pH meter 925	0.01 units
	WTW Multiline P4 portable meter fitted with a SenTix 41 probe	0.01 units
Conductivity (Cond)	Copenhagen Model CDM 83 Conductometer	< 1 μS/cm
	WTW Multiline P4 portable meter fitted with a With a TetraCon 325 probe	1 μS/cm
Chlorophyll *a*	Ethanol extraction and fluorometry (Nusch 1980) using a Turner designs model 10 fluorometer	0.1 $\mu g/cm^2$

[a]NTU = nephlometric turbidity unit. Conductivity was measured in the laboratory and in the field.

stock treatments monthly between June and October 1999 and May and October 2000. For each monthly collection, samples were taken from enclosures on the same day; however, because minimal rain fell during the study period, collection of water samples for chemical analyses typically occurred during stable periods of flow rather than that based on increasing or decreasing hydrographs following precipitation events.

Water samples for TP, SRP, TN, and $NO_2^- + NO_3^-$ were collected in 1-l polystyrene bottles and analyzed in the laboratory. Unfiltered water was used for TP and TN analyses, whereas filtrate was used for SRP (Millipore HA, 45 μm) and $NO_2^- + NO_3^-$ analyses. Samples for SRP were analyzed within 24–36 hours of collection. Water pH was measured both in the laboratory and in the field using a water quality meter, whereas turbidity was measured solely in the laboratory (Table 2).

Effects of livestock grazing on water quality were determined by quantifying differences in water chemistry in treatments compared with that in the buffer located immediately upstream of the livestock treatment. We assumed that the effects of the livestock grazing treatment on water quality would be highest (i.e., most detectable) at the most downstream area in each of the livestock enclosure treatments. Thus, water samples were collected within the last 5 m of enclosure and buffer areas.

The effects of the four livestock grazing treatments on water quality were evaluated with Wilcoxon paired-rank sample tests, where differences in water chemistry between treatments and the buffer located immediately upstream were paired through time. Consequently, this approach defines the effect of a livestock grazing treatment based on changes in water chemistry immediately upstream of the livestock enclosure compared with that leaving the enclosure. This resulted in nine (i.e., the late season grazing treatments) and 11 difference values (i.e., no grazing treatment, early season grazing treatment, all-season grazing treatment) throughout the June–October 1999 and May–October 2000 period. Variable sample replication among the four livestock grazing treatments arises because while the livestock-absent and early and all season livestock grazing treatments were initiated in June 1999, the late season livestock grazing treatment was not initiated until August 1999 (i.e., two months later).

Our experimental design raises concerns about the statistical independence of data when the four experimental treatments are located in a series, from upstream to downstream, along the same stream channel. Under this design, an effect of livestock grazing on water quality in an upstream enclosure could affect water quality at sites downstream. We tested for autocorrelations (i.e., a correlation of a series with itself shifted by particular lag of *k* observations) in all eight water quality variables (i.e., total phosphorus, soluble reactive phosphorus, total nitrogen, nitrate + nitrite, dissolved organic carbon, dissolved oxygen, turbidity and conductivity) using the Box Ljung Q statistic. Because our primary focus was to test for autocorrelations in water chemistry among adjacent enclosures (i.e., buffer and livestock enclosures), these analyses were completed after lagging data by one observation (i.e., one site). Tests for autocorrelations were completed separately for all water chemistry variables for each of the 11 sampling periods (i.e., samples collected between June and October 1999 and May and October 2000) and resulted in a total of 264 possible comparisons (3 streams × 8 water quality variables × 11 sampling periods). Analyses were completed using Statistica Software (Statistica 1999) with $\alpha = 0.05$.

Epilithon. Samples for epilithic biomass determination were collected from livestock treatments and buffers in June, July, August, and October 1999 and from livestock treatments on 21 September 2000. An individual sample was removed from each of four randomly selected individual stones collected from shallow areas at each site where water depths ranged from about 0.10 m to 0.35 m. A randomly selected 9.6-cm^2 area (Scrimgeour and Chambers 1997) was scraped from the upper surface of each stone, stored in the dark and on ice for 2–6 hours before being frozen prior to analyses. Epilithic biomass, expressed as chlorophyll *a* (Chl *a*), for individual stone scrapings was determined following the method of Nusch (1980) and the concentration determined on a Turner Designs model 10 series fluorometer. Differences in epilithic biomass among livestock treatments and buffers were evaluated using ANOVA as described previously.

We tested for spatial autocorrelations in mean algal biomass among the livestock grazing enclosures using methods described previously for water quality variables. These analyses were completed using mean algal biomass estimates obtained from Battle, Graburn, and Nine Mile creeks in June, July, August, and October 1999. Low sample sizes ($N = 2$ and 4) precluded testing for spatial autocorrelations in algal biomass among enclosures in 2000.

Sampling of springs. Observations in 1999 indicated that livestock enclosures in all three creeks received inputs from 5 to 10 small (width 0.05–0.5 m), low volume (<0.001 m^3/sec) springs. Because these springs could account for substantial inputs of $NO_2^- + NO_3^-$ and SRP that could mask effects of livestock grazing treatments, we collected water samples from all springs within and adjacent to stream enclosures for $NO_2^- + NO_3^-$ and SRP analyses approximately monthly between June 1999 and October 1999 and May and October 2000. These data allowed us to compare concentrations of $NO_2^- + NO_3^-$ and SRP in streams with those in springs. Because the majority of springs are small (width 0.10–0.5 m), water samples were collected using a 250-ml syringe or a 120-ml polypropylene scoop.

In June, August, September, and October 2000, we estimated daily loadings of $NO_2^- + NO_3^-$ and SRP from springs to streams by combining the collection of water samples for SRP and $NO_2^- + NO_3^-$ analyses with estimates of discharge from springs. Two to three estimates of discharge were taken from individual springs by determining the length of time required for spring flow to fill the polypropylene scoop.

Our preliminary observations indicated that the presence of small groundwater springs could mask effects of the four livestock grazing treatments if inputs of $NO_2^- + NO_3^-$ and SRP from groundwater springs were moderately high and positively related with instream concentrations of $NO_2^- + NO_3^-$ and SRP. We quantified relationships between concentrations of $NO_2^- + NO_3^-$ and SRP in streams with daily loadings of $NO_2^- + NO_3^-$ and SRP from springs using linear regression. These analyses were completed separately using concentrations and loading estimates of $NO_2^- + NO_3^-$ and SRP obtained from Battle, Grabun, and Nine Mile creeks in June, August, September, and October, 2000.

Longitudinal stream surveys and nutrient limitation.

Nutrients and epilithon. Longitudinal patterns in $NO_2^- + NO_3^-$ and SRP were evaluated in 1999 (24 August: Battle, Graburn, and Nine Mile creeks) and 2000 (29 June: Battle, Graburn, Nine Mile creeks; 24 August: Battle, Graburn, Nine Mile, Storm creeks; 4 October: Battle, Graburn, Nine Mile, Storm creeks) whereas spatial variation in epilithon was evaluated only in 2000 on 24 August (Battle, Nine Mile, Graburn, and Storm creeks) and 4 October (Battle and Graburn creeks). Water and algal samples were collected at 0.5- to 3-km intervals, determined largely by trail access to streams, along 4- to 12-km lengths of all study streams and analyzed as described previously. These data allowed comparisons of concentrations of nutrients and epilithic biomass in upper reaches with that in lower reaches to determine: (1) the extent to which algal biomass could be predicted from concentrations of SRP and $NO_2^- + NO_3^-$ and (2) whether predicted increases in SRP and $NO_2^- + NO_3^-$ resulting from the all-season livestock grazing treatment established in Battle and Graburn creeks affected watershed-scale variation in epilithic biomass compared to that in Nine Mile and Storm creeks, where these treatments where not established.

Nutrient limitation. Spatial patterns in nutrient limitation were investigated using nutrient-diffusing substrata (NDS) (Scrimgeour and Chambers 1997) in June–July (summer: Battle, Graburn, and Nine Mile creeks) and August (early fall: Battle, Nine Mile, Graburn, and Storm creeks). Briefly, each NDS consisted of a porous clay pot (6 cm high, 11 cm wide, volume 325 ml) filled with a hot solution containing agar (20 g/liter in autoclaved deionized, distilled water) mixed with one of the test compounds: nitrogen (0.8 M $NaNO_3$), phosphorus (0.5 M KH_2PO_4), nitrogen (0.8 M $NaNO_3$) + phosphorus (0.5 M KH_2PO_4) enriched or unamended controls, and sealed with a 4-mm polypropylene base (12 cm diam) using aquarium safe silicone sealant (Scrimgeour and Chambers 1997). NDS were deployed about 50-m downstream of the livestock enclosures in the summer at Battle, Graburn, and Nine Mile creeks (28 June–10 July) and in the fall at Battle, Graburn, Nine Mile, and Storm creeks (7–18 August). Deploying NDS at sites downstream of enclosures ensured that nutrients released from NDS did not affect nutrient levels measured in enclosures. During removal from Battle, Graburn, and Nine Mile creeks, epilithon was scraped from a randomly selected 9.6 cm^2 area from each NDS, whereas epilithon from Storm Creek was removed from each of two randomly selected 9.6-cm^2 areas. All samples were kept on ice in the field, frozen, and later analyzed for Chl *a* as described previously. At each site, ANOVA and Bonferroni-adjusted orthogonal contrasts were based on 9 or 10 replicates of each of the four treatments.

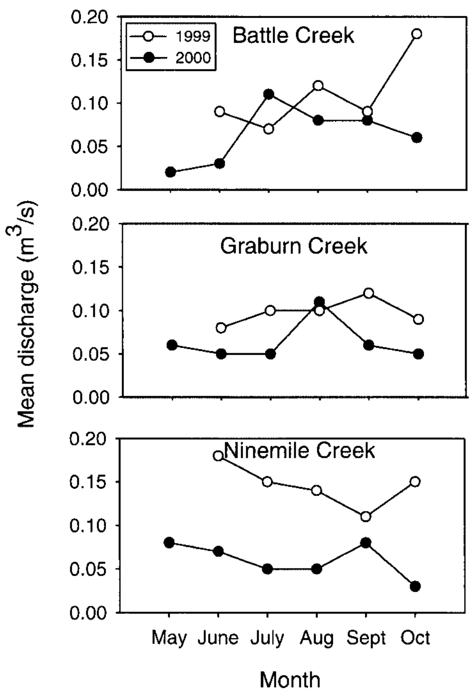

Figure 1. Mean monthly discharge in enclosures in Battle, Graburn and Nine Mile creeks, June–October 1999 and May–October 2000.

Results

Precipitation and Stream Discharge

Mean monthly precipitation in Battle, Graburn, and Nine Mile creeks during the summer–fall period was almost 50% lower in 2000 (overall mean 155 mm) than that in 1999 (100 mm) and coincided with substantially reduced flows, especially during the fall period (Figure 1). Discharge was highly variable between years and was about twofold lower in 2000 compared to that in 1999 for all streams (Figure 1). On average, discharge at the most downstream location in enclosures was 28% (Nine Mile Creek), 34% (Graburn Creek), and 45% (Battle Creek) higher than that at the most upstream enclosure.

Livestock Enclosures

Streambanks, riparian, and instream channel characteristics.

Streambanks. Exclusion of livestock for two years (i.e., October 1998–September 2000) resulted in significant improvements in streambank stability (Figure 2). The proportion of stable streambanks in Battle, Graburn, and Nine Mile creeks was significantly affected by the interaction of livestock enclosure type and year in Battle Creek (RM-ANOVA: $F_{(8,81)} = 14.0$, $P < 0.0001$), Graburn Creek ($F_{(8,81)} = 10.1$, $P < 0.0001$) and in Nine Mile Creek ($F_{(4,45)} = 6.3$, $P < 0.005$). The proportion of stable streambanks increased by about 50% from 46%–70% prior to the exclusion of livestock to 73%–94% in livestock-absent enclosures (i.e., buffers and the no-graze treatment) but not in livestock-present treatments (overall range 50%–67%; Figure 2). The propensity of stable streambanks was also significantly ($P < 0.05$) higher in the non-grazed reach of Storm Creek (mean ± 1SE = 90% ± 4%) compared with the grazed reach (48% ± 3%) (Figure 2).

Riparian vegetation. The biomass of riparian vegetation located 1–10 m from stream channels also differed significantly among livestock treatments and sites located upstream and downstream of enclosures (ANOVA: Battle Creek: $F_{(10,44)} = 10.3$, $P < 0.001$; Graburn Creek: $F_{(10,44)} = 8.8$, $P < 0.001$; Nine Mile Creek: $F_{(6,28)} = 26.0$, $P < 0.001$) (Figure 3). Riparian biomass in buffers and the livestock-absent treatments exceeded that in the early season, late season, all season livestock grazing treatments, and sites located immediately upstream and downstream of enclosures (Bonferroni adjusted orthogonal contrasts; Figure 3). Similarly, biomass of riparian vegetation in the nongrazed area of Storm Creek was 25 times (*t* test: $t = 7.8$, $df = 8$, $P < 0.001$) greater than that in the area grazed by livestock (Figure 3). The absence of significant differences in

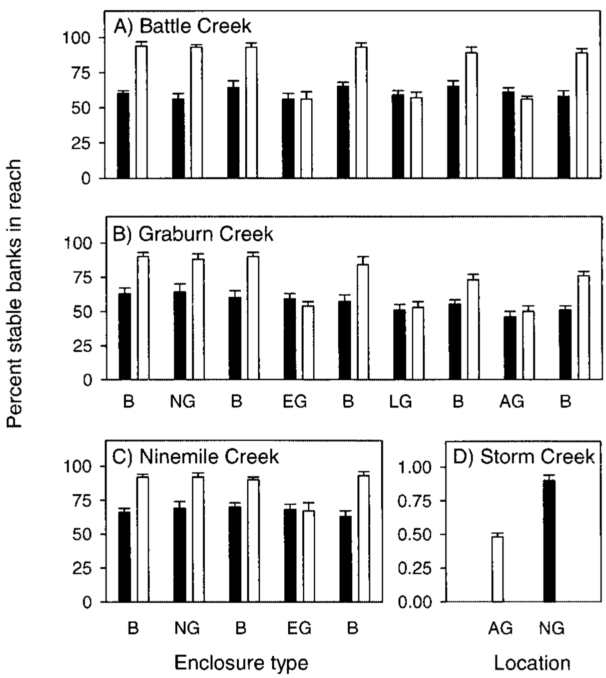

Figure 2. Mean (± 1 SE) percent stable streambanks in livestock enclosures in Battle, Graburn, and Nine Mile creeks in October 1998 (filled histograms) and October 2000 (open histograms) (A–C), and grazed (open histogram) and nongrazed (filled histogram) areas of Storm Creek, October 2000 (D). B = buffer, NG = no livestock grazing, EG = early season livestock grazing, LG = late season livestock grazing, AG = all-season livestock grazing.

biomass among livestock-absent enclosures in Battle, Graburn, and Nine Mile creeks and the nongrazed reach in Storm Creek ($F_{(3,80)} = 0.7, P > 0.05$) suggest that exclusion of livestock from Battle, Graburn, and Nine Mile creeks for about two years resulted in rapid accrual of vegetation (Figure 3).

Instream vegetation cover. The extent to which vegetation covered stream channels also differed among enclosures (Battle: $F_{(8,36)} = 9.8$, $P < 0.0001$; Graburn: $F_{(8,36)} = 7.0$, $P < 0.0001$; and Nine Mile: $F_{(4,20)} = 11.1$, $P < 0.0001$) (Figure 3). In the absence of livestock, vegetation typically covered about 65%–80% of the stream channel but was at least twofold lower (20%–40%) when livestock were present. Vegetation covered about 75% of stream channels in the nongrazed reach of Storm Creek and was about threefold higher compared with the upstream grazed reach (Figure 3). The extent to which livestock reduced instream vegetation cover was less apparent in Battle Creek compared to the others probably because three escaped cattle fed within buffers prior to being returned to their respective enclosures later in September 2000.

Water Chemistry.

Seasonal patterns. Concentrations of TP and SRP in enclosures in Battle Creek typically ranged from 60 to 90 μg/liter and 50 to 80 μg/liter, respectively and were generally higher than that in Nine Mile (typical ranges were 30–60 μg/liter, TP/liter, 20–40 μg SRP/liter) and Graburn creeks (typical ranges of 20–50 μg TP/liter, 20 to 40–50 μg SRP/liter. Total nitrogen and NO_2^- + NO_3^- concentrations in Graburn and Nine Mile creeks generally increased throughout the summer–fall period from about 200 to 1000, μg TN/liter and from 100 to 800 μg NO_2^- + NO_3^- (Graburn Creek) and from 200 to 400–500 μg TN/liter and from 30 to 300–500 μg NO_2^- + NO_3^-/liter (Nine Mile Creek). While concentrations of NO_2^- + NO_3^- in Battle Creek also increased throughout the summer–fall period (20–150 μg/liter in 1999 and from <10 to 20–100 μg/liter in 2000), seasonal increases in TN were less apparent and increased about twofold (i.e., 100–200 μg/liter) between June and October 1999 and May and October 2000.

In 1999, concentrations of DOC decreased throughout the summer–fall period from about 2 to 1.6 mg/liter in Battle and Graburn creeks and from 2.7 to 2 mg/liter in Nine Mile Creek. Concentrations of DOC were similar in all streams in 2000 but increased between May and September in Battle (1.1 to 2–3.2 mg/liter) and Graburn creeks (1.8 to 3 mg/liter) and declined thereafter. Concentrations of DO did not differ appreciably between May and October 2000. Turbidity levels in all streams were extremely low (<1 NTU) and varied relatively little among years or with season.

Effects of livestock grazing treatments. We tested for spatial autocorrelations in all eight water quality variables using the Box-Ljung Q statistic after introducing a spatial lag of one site. Results of these tests showed that 12 of the 264 (i.e., 4.5% of all tests completed) individual tests (i.e., 8 water quality variables × 3 streams × 11 sampling periods) revealed significant ($P < 0.05$) autocorrelations between sites. The majority of autocorrelations among water chemistry variables were observed in Graburn Creek (7 of the 12) with two and three significant autocorrelations observed in Battle and Nine Mile creeks, respectively. Because significant autocorrelations between sites violate the assumption of independence of observations, we completed Wilcoxon paired-rank tests after removing data sets where significant autocorrelations were observed. Consequently, of the 24 comparisons (i.e., 8 water quality variables × 3 streams) of water quality variables among the four livestock grazing treatments, 15 were based on 9 and 11 values, 7 were based on 8 and 10 values, 1 was based on 7 and 9 values (conductivity in Nine Mile

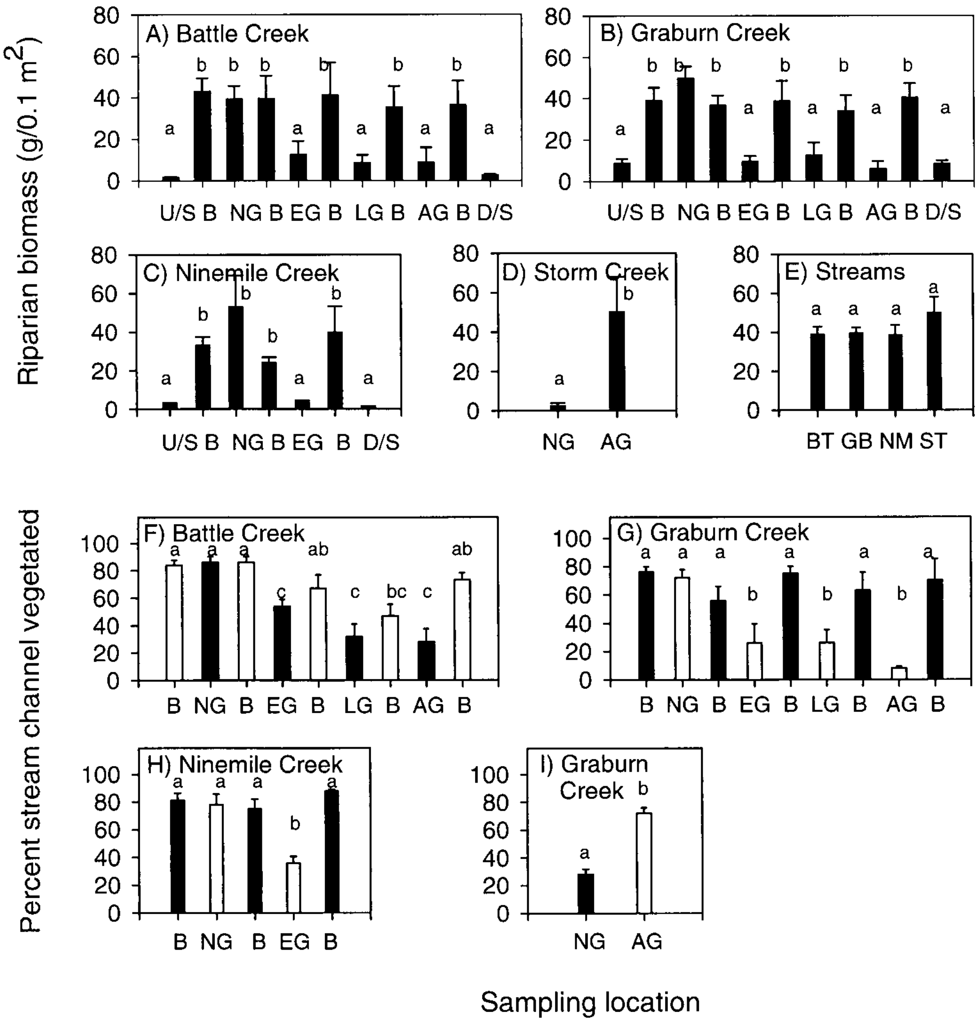

Figure 3. Mean (± 1 SE) biomass (g/0.1 m^2) of riparian vegetation in Battle, Graburn, and Nine Mile creeks (**A–C**), grazed and nongrazed areas of Storm Creek (**D**), and nongrazed treatment areas in Battle, Graburn, and Nine Mile creeks with the nongrazed area in Storm Creek, October 2000 (**E**) and mean (± 1 SE) percent vegetation cover in stream channels in Battle, Graburn, Nine Mile, and Storm creeks in October 2000 (**F–I**). Sampling locations represent sites: (1) located within (B = buffer, NG = no livestock grazing, EG = early season livestock grazing, LG = late season livestock grazing, AG = all-season livestock grazing) or adjacent (i.e., U/S = upstream of livestock enclosures, D/S = downstream of livestock enclosures) to enclosures, (2) nongrazed (NG) and all-season grazed (AG) sites in Storm Creek, and (3) stream-scale comparisons: BT = Battle Creek, GB = Graburn Creek, NM = Nine Mile Creek, ST = Storm Creek. Histograms sharing the same letter are not significantly different.

Creek) and another based on 6 and 8 values (NO_2^- + NO_3^- in Graburn Creek). Comparisons using samples sizes of 7 and 9 values likely resulted in low statistical power.

Bonferroni-adjusted, Wilcoxon paired-rank tests revealed significant differences in water chemistry among the four livestock treatments in Battle and Graburn but not Nine Mile Creek (Figure 4). In Graburn Creek, concentrations of total phosphorus in the all-season livestock grazing treatment was significantly higher than that in the livestock-absent control, and the early season and late season grazing treatments. Concentrations of soluble reactive phosphorus in the all-season livestock grazing treatment also exceeded that in livestock-absent control.

However, in the majority of cases (i.e., 22 of the 24 remaining comparisons), differences among treatments were minor even when differences were statistically significant ($P < 0.05$). For example in Battle Creek, turbidity was significantly higher in the early season grazing treatment compared to the late season grazing treatment, but differences among treatment

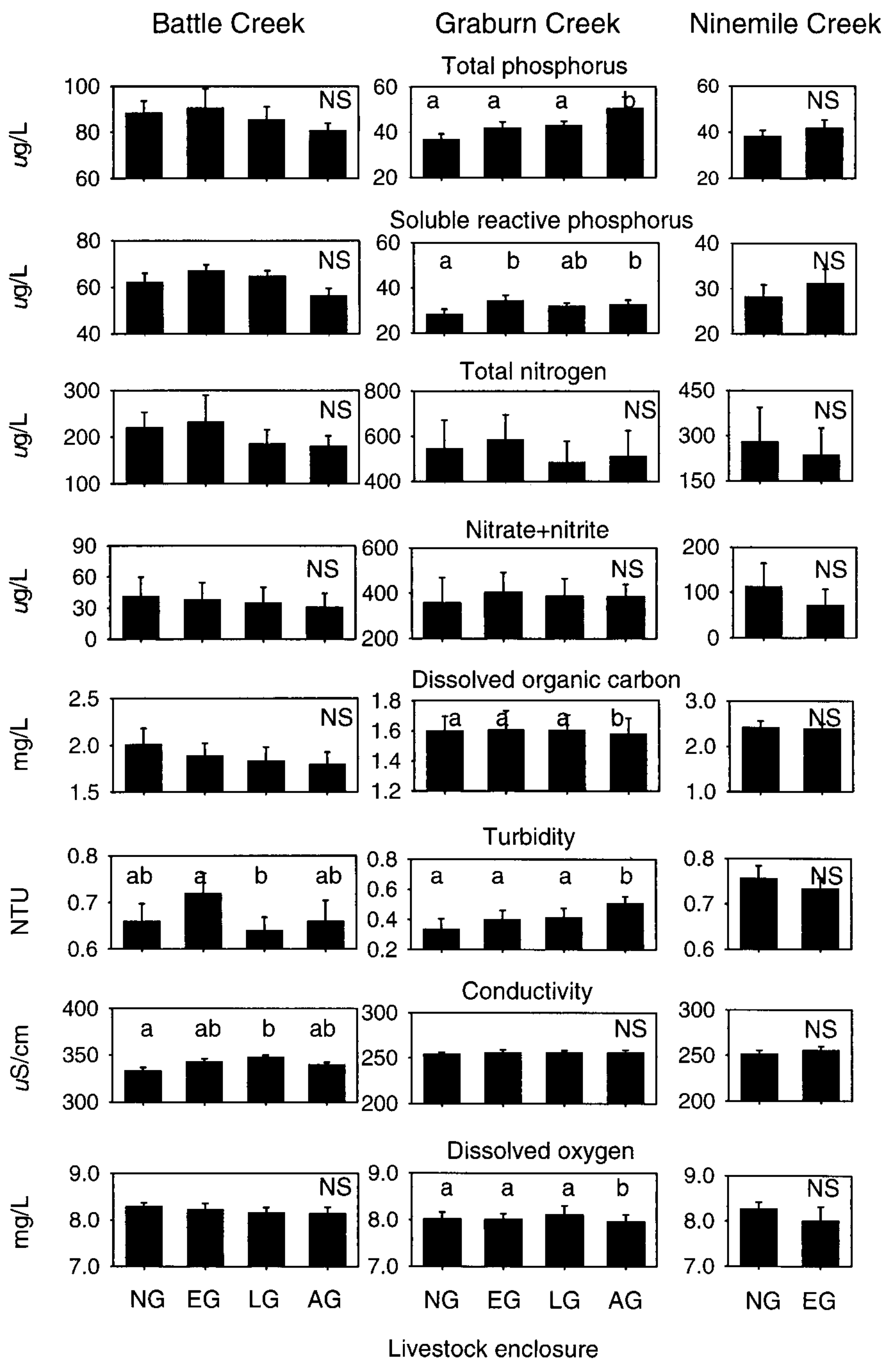

Figure 4. Comparison of mean (± 1SE) water quality parameters among livestock grazing treatments in Battle, Graburn, and Nine Mile creeks, 1999–2000. Differences among livestock grazing treatments were determined using Bonferroni-adjusted, Wilcoxon paired-rank tests. NS = not significant; histograms sharing the same letter are not significantly.

means were small (i.e., <0.1 NTU). Similarly, conductivity in the livestock-absent grazing treatment was significantly greater than that in the late season grazing treatment despite a difference of only 9 μS/cm among treatments. Differences in concentrations of total nitrogen and $NO_2^- + NO_3^-$ among livestock enclosures were not apparent in any of the three streams, although low sample sizes (i.e., 6 and 8) in Graburn Creek reduced statistical power of these tests.

Lastly, turbidity and concentrations of DOC and DO also differed ($P < 0.05$) among the four livestock treatments but differences among means were minimal (i.e., DOC = 0.03 mg/liter, DO = 0.2 mg/liter, NTU = 0.17) (Figure 4).

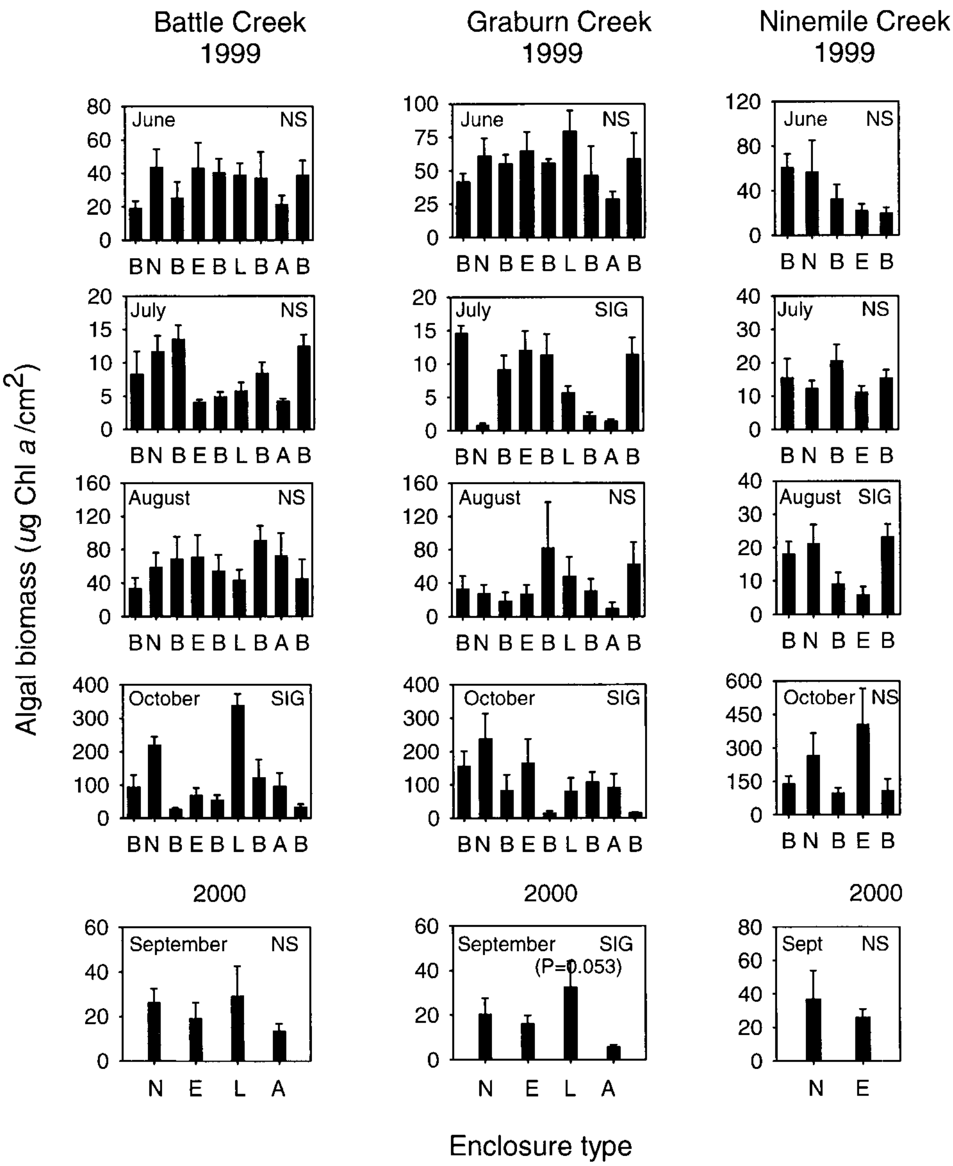

Figure 5. Mean algal biomass (μg Chl a/cm^2) on upper stone surfaces in livestock enclosures in Battle, Graburn, and Nine Mile creeks, June–October 1999 and September 2000. B = buffers, N = no livestock grazing, E = early season livestock grazing, L = late season livestock grazing, A = all-season livestock grazing. NS = not significant.

Epilithon. Results from Box-Ljung Q tests showed that mean algal biomass in enclosures were not autocorrelated ($P > 0.05$) with that in the enclosure located immediately downstream for any of the 12 comparisons (i.e., 3 streams × 4 sampling periods). Tests for autocorrelations in water quality variables also indicated that water quality at a given site was typically (252 of the 264 comparisons) not correlated with that in the enclosure located immediately downstream. These data suggest that the algal biomass in buffers and the livestock grazing treatments are independent and that comparisons of mean algal biomass within enclosures using ANOVA is valid.

Differences in algal biomass among livestock grazing treatments were highly variable, and there was no consistent pattern among creeks (Figure 5). In the majority of cases, effects of livestock grazing on algal biomass were not detectable (ANOVA on $\log_{10}$-transformed data, $P > 0.051$; Figure 5) and significant ANOVA models did not always indicate effects of livestock grazing. For example, the significant ANOVA model for Battle Creek in October 1999 ($F_{(8,27)} = 7.9$, $P < 0.001$) resulted from significantly higher algal biomass in the late-season grazing treatment compared with all the buffer areas (Bonferroni adjusted orthogonal contrasts). Algal biomass in the livestock-absent treatment did not differ significantly from that in the early season, late season, and all-season grazing treatments. In Gra-

Table 3. Number, mean concentrations, and total daily loadings of soluble reactive phosphorus (SRP) and nitrate + nitrite (NO_3^-) in groundwater springs input entering livestock enclosures in Battle, Graburn, and Nine Mile creeks, Cypress Hills, Alberta[a]

	Concentrations (μg/liter)								
	Battle Creek			Graburn Creek			Nine Mile Creek		
Enclosure	*N*	SRP	NO_3^-	*N*	SRP	NO_3^-	*N*	SRP	NO_3^-
B	0	—	—	0	—	—	1	16	266
NG	1	78	224	0	—	—	4	36	86
B	1	87	238	0	—	—	0	—	—
EG	0	—	—	5	81	1310	0	—	—
B	0	—	—	0	—	—	0	—	—
LG	3	80	292	0	—	—	—	—	—
B	1	76	180	2	65	1298	—	—	—
AG	0	—	—	2	66	1558	—	—	—
B	1	73	148	1	62	976	—	—	—

	Daily loadings (mg/day)					
	Battle Creek		Graburn Creek		Nine Mile Creek	
	SRP	NO_3	SRP	NO_3^-	SRP	NO_3^-
B	—	—	—	—	57	975
NG	58	167	—	—	225	545
B	108	296	—	—	—	—
EG	—	—	1472	23759	—	—
B	—	—	—	—	—	—
LG	24	85	—	—		
B	393	934	562	11183		
AG	—	—	438	10343		
B	507	1023	278	4351		

[a]Livestock grazing treatments: B = buffer (no livestock), NG = no livestock grazing, EG = early season livestock grazing, AG = all-season livestock grazing; N= number of springs. Concentrations are based on samples collected in 1999 and 2000 whereas loadings are based on samples collected in 2000.

burn Creek significant differences in algal biomass in July 1999 (ANOVA: $F_{(8,27)} = 21.3, P < 0.001$) resulted primarily from differences between buffers located immediately upstream and downstream of the livestock-absent treatment compared with the livestock-absent treatment and the buffer located immediately downstream of the all-season grazing treatment compared with that in the all-season grazing treatment. At the end of the experiment (i.e., September 2000), differences in algal biomass among livestock grazing treatments were apparent only in Graburn Creek (ANOVA: $F_{(3,12)} = 3.4, P = 0.053$) where algal biomass in the livestock-absent, early season, and late season treatments was significantly higher than that in the all-season grazing treatment (Bonferroni adjusted orthogonal contrasts) (Figure 5).

Chemistry of springs. The upper reaches of Battle, Graburn and Nine Mile creeks receive small, low-volume inputs (0.01–0.09 liters/sec) from spring-fed streams located adjacent to the main channel (Table 3). Concentrations of SRP and $NO_2^- + NO_3^-$ in these inputs were highly variable and ranged from 16 to 87 and 86 to 1558 μg/liter, respectively. Concentrations of SRP and $NO_2^- + NO_3^-$ in groundwater springs typically exceeded that in stream channels (Table 3). In Battle Creek, concentrations of SRP and $NO_2^- + NO_3^-$ exceeded that in stream channels by about 0.2 (SRP) and four- to sixfold ($NO_2^- + NO_3^-$) whereas in Graburn Creek, concentrations of SRP and $NO_2^- + NO_3^-$ in springs exceeded that in the stream channel by about two- and threefold, respectively. In Nine Mile Creek, concentrations of SRP in springs (16–36 μg/liter) were similar to that in the stream channel (29–34 μg/liter) whereas concentrations of $NO_2^- + NO_3^-$ were up to 2.5-fold higher in springs compared with stream channels (Table 3).

Conversion of concentrations recorded in 2000 to daily loadings showed that inputs of SRP and $NO_2^- + NO_3^-$ varied by 21-, 5-, and 4-fold (SRP) and 6-, 5-, 2-fold among enclosures in Battle, Graburn and Nine Mile creeks, respectively (Table 3). On average, loadings of SRP from springs to enclosures were highest in

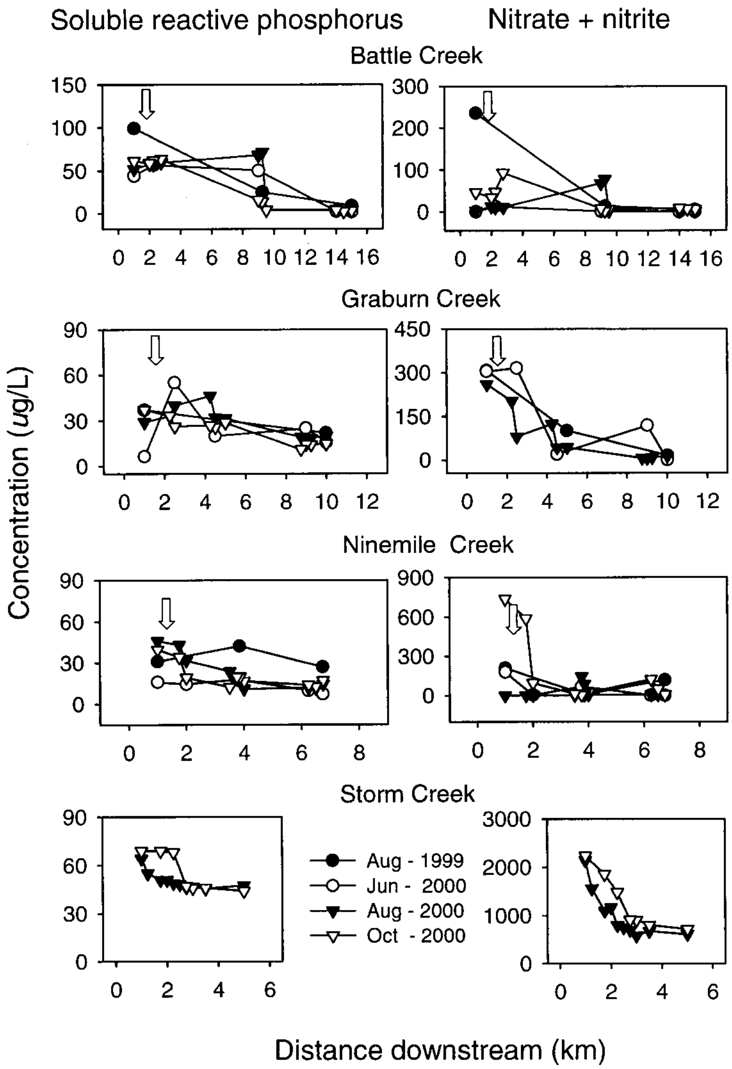

Figure 6. Longitudinal variation in concentrations of nitrate + nitrite and soluble reactive phosphorus in Battle, Graburn, Nine Mile, and Storm creeks, 1999 and 2000. Arrows indicate approximate location of livestock enclosures.

Graburn Creek (mean 689 mg/day) compared to Battle (mean 218 mg/day) and Nine Mile Creek (mean 141 mg/day). Daily loadings of NO_2^- + NO_3^- from springs to enclosures were highest in Graburn Creek (mean 12,409 mg/day) compared to Nine Mile (mean 760 mg/day) and Battle Creek (mean = 501 mg/day).

Linear regression showed that concentrations of SRP and NO_2^- + NO_3^- in enclosures Battle, Graburn, and Nine Mile creeks were unrelated ($P > 0.05$) with daily loadings of SRP and NO_2^- + NO_3^- from springs in June, August, September, and October, 2000.

Longitudinal Patterns in SRP, NO_2^- + NO_3^- and Benthic Algal Biomass

SRP and NO_2^- + NO_3^-. Concentrations of SRP and NO_2^- + NO_3^- varied appreciably among streams and with distance downstream and time (Figure 6). Concentrations of SRP typically ranged from 20 to 200 μg/liter, whereas concentrations of NO_2^- + NO_3^- were highest in Storm Creek (overall average 1006.8 μg/liter) followed by Graburn (99.7 μg/liter), Nine Mile (91.9 μg/liter) and Battle Creek (25.5 μg/liter). While variable among streams, concentrations of NO_2^- + NO_3^- typically decreased with distance downstream. For instance, in Graburn Creek, NO_2^- + NO_3^- decreased from 260–300 μg/liter in upper reaches to <15 μg/liter in lower reaches whereas concentrations of NO_2^- + NO_3^- decreased from about 2000 in the upper to 600 μg/liter in the lower reaches of Storm Creek (Figure 6).

Epilithic biomass. Epilithic biomass, measured as Chl *a* concentration, varied significantly with distance downstream in Battle Creek (August: ANOVA on $\log_{10}$-transformed data: $F_{(9,30)} = 9.9$, $P < 0.0001$), Graburn

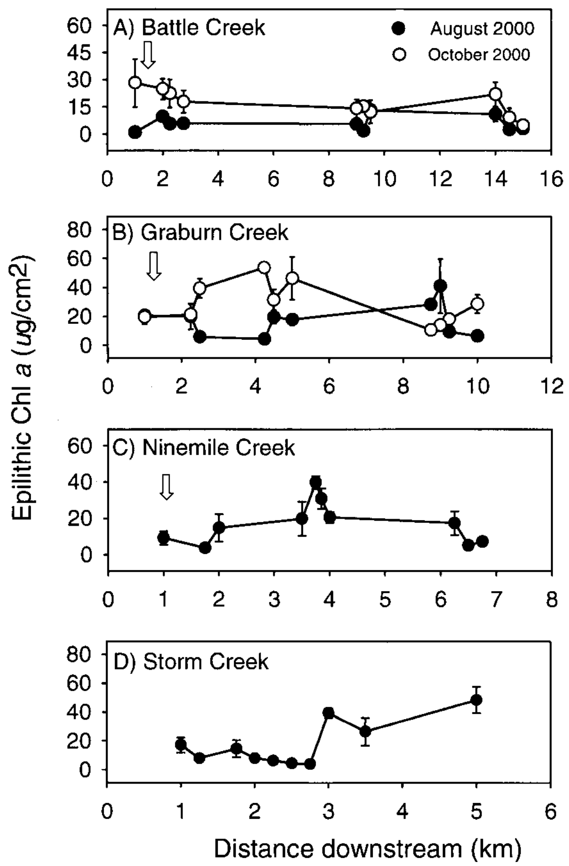

Figure 7. Longitudinal variation in mean (± 1SE) algal biomass (μg Chl *a*/cm^2) from upper stone surfaces in Battle, Graburn, Nine Mile, and Storm creeks, August and October 2000. Arrows indicate approximate location of livestock enclosures.

(August: $F_{(9,30)} = 2.7$, $P < 0.05$, October: $F_{(9,30)} = 4.4$, $P < 0.001$), Nine Mile (August: ANOVA: $F_{(9,30)} = 5.5$, $P < 0.001$), and Storm creeks (August: $F_{(9,30)} = 10.0$, $P < 0.0001$) but not in Battle Creek in October (ANOVA, $F_{(9,30)} = 2.0$, $P > 0.05$) (Figure 7). In contrast with Battle and Graburn creeks, epilithic Chl *a* generally increased with distance downstream in Storm Creek (Figure 7).

Regression analyses showed that epilithic Chl *a* was unrelated ($P > 0.05$) to SRP in Battle and Storm creeks, but related with SRP in Graburn and Nine Mile creeks where second-order polynomials explained 51% and 47% of variance in epilithic biomass based on SRP concentrations, respectively. Epilithic Chl *a* was also unrelated ($P > 0.05$) with concentrations of $NO_2^- + NO_3^-$ in Battle and Storm creeks but positively and nonlinearly related to concentrations of $NO_2^- + NO_3^-$ in Graburn and Nine Mile creeks (Figure 8).

Nutrient Limitation

Epilithic communities in the Cypress Hills were N-limited or nutrient replete depending upon stream and season (Figure 9). While epilithic algae at Battle and Nine Mile creeks were N-limited in both summer and fall, algal communities in Graburn Creek were nutrient unlimited in the summer but N-limited in the fall. Epilithic communities in Strom Creek were nutrient replete in the fall. While the absence of P limitation is consistent with high SRP concentrations, N limitation occurred at both high and low $NO_2^- + NO_3^-$ levels (Table 4).

Discussion

We used a within-watershed design to compare the effects of four grazing treatments on riparian forage biomass, instream vegetation cover and bank stability, water quality, and epilithic biomass. Our stream-reach design (Platts and Nelson 1985) differs markedly from watershed-scale approaches typically used to evaluate effects of agricultural activities on water chemistry and biotic communities (Cooke and Prepas 1998, Chambers and others 2001). However, the within-watershed or stream-reach approach can provide insights on the effects of livestock grazing that would otherwise be precluded due to the absence of replicate streams.

Bank Stability

Erosion of steambanks is arguably one of the dominant negative effects of livestock grazing that can be remedied by reducing or excluding livestock from stream channels (Kauffman and others 1983, Sheffield and others 1997). For example, Sheffield and others (1997) reported the streambank erosion declined by 77% when alternative water sources reduced use of streams by livestock in Virginia. Similarly, Kauffman and others (1983) reported significantly greater streambank losses in grazed areas compared to nongrazed areas in Oregon. We predicted that exclusion of livestock from buffers and the livestock-absent enclosures would result in significant improvements in bank stability. Consistent with this prediction, our results indicated that exclusion of livestock from streams in the Cypress Hills significantly increased the proportion of stable streambanks by about 50% in buffers and the no-livestock grazing treatments following exclusion of livestock for two years. In contrast, the proportion of stable streambanks did not differ markedly between 1998 and 2000 when livestock were allowed access to streams in the early, late, and all-season grazing treatments. Increases in the proportion of stable stream-

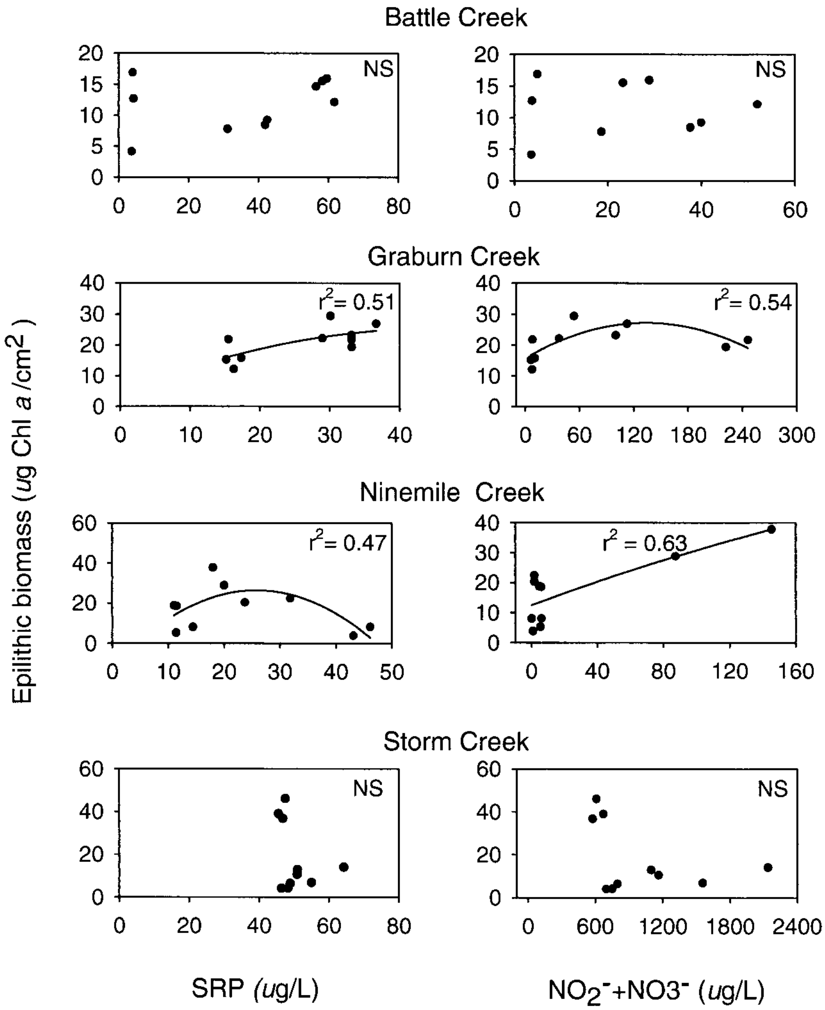

Figure 8. Linear and nonlinear regression analyses of algal biomass (μg Chl a/cm^2) with soluble reactive phosphorus (SRP) and nitrate + nitrite from Battle, Graburn, Nine Mile, and Storm creeks, 2000.

banks arise because of rapid growth of vegetation along riparian banks and in stream channels that stabilized previously eroded regions.

Our data indicated that the increased stability of streambanks resulted primarily from the establishment of grasses, rather than shrubs, willow, and aspen within riparian areas. These results further support the notion that the reestablishment of grassy riparian zones may be a useful restoration option (Lyons and others 2000). In the long term, the predominance of grassy riparian zones within the Cypress Hills following the reduced livestock grazing may likely arise because beavers remove the majority of mature aspen within up to 40–70 m from streams and moose and elk intensively browse young and aspen and willow.

Riparian and Instream Vegetation

Intensive livestock grazing can result in dramatic reductions in riparian vegetation and, if present, vegetation within stream channels. Our visual surveys of the Cypress Hills suggested that high utilization by livestock reduced riparian and instream vegetation. Based on these observations, we predicted that exclusion of livestock from buffers and the livestock-absent enclosures would result in significant improvements in riparian biomass and the extent to vegetation covered stream channels. Consistent with our observation, exclusion of livestock for the two summer–fall periods typically resulted in a three- to fivefold increase in riparian vegetation biomass and a twofold increase in the extent to which vegetation covered stream channels.

We also hypothesized that differences in the timing and length of the three grazing treatments (i.e., early versus late season grazing and two versus four month grazing periods) would produce detectable differences in streambank, riparian, and stream conditions. In contrast, our results indicated that bank stability, riparian vegetation, and instream vegetation

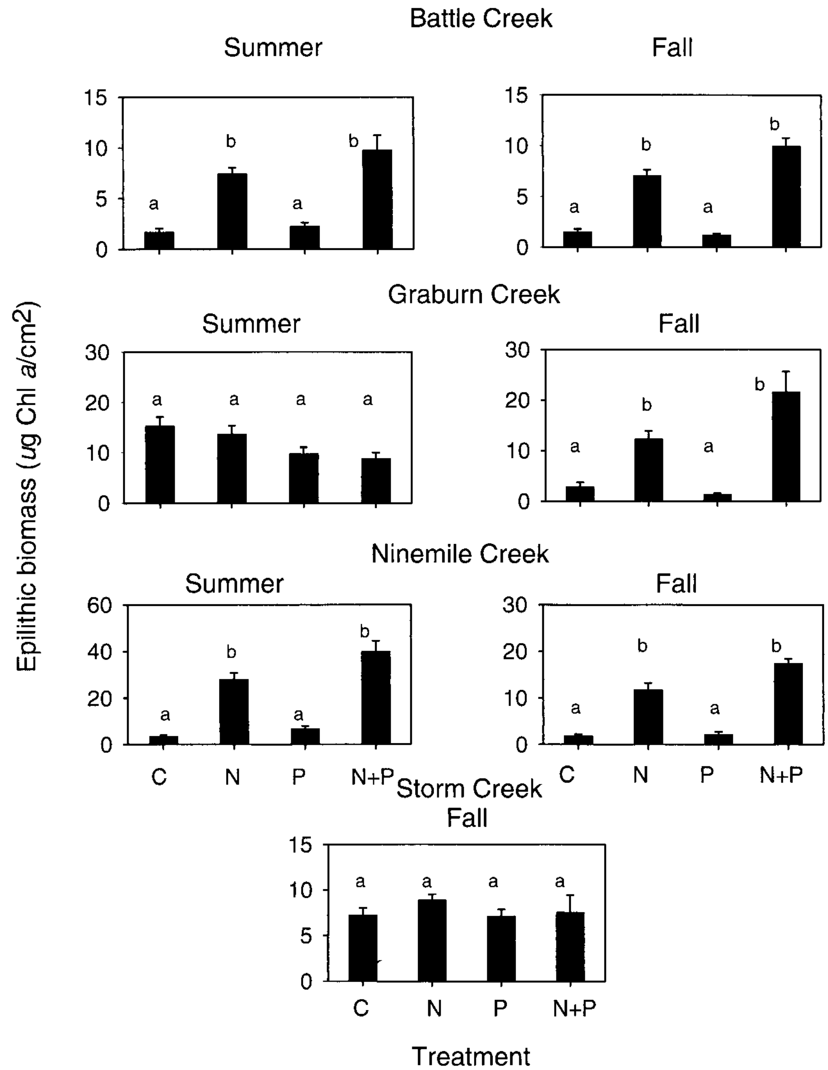

Figure 9. Mean (± 1SE) epilithic Chl *a* concentrations from nutrient diffusing substrata deployed in Battle, Graburn, Nine Mile, and Storm creeks in summer and fall 2000. C = nutrient absent controls, N = nitrogen enriched, P = phosphorus enriched, N + P = nitrogen and phosphorus enriched. Histograms sharing the same letter are not significantly different.

Table 4. Mean concentrations of soluble reactive phosphorus (SRP) and nitrite + nitrate (NO_3^-) when nutrient-diffusing substrata (NDS) were deployed in the summer (28 June–11 July) and fall (7–17 August) from four creeks in the Cypress Hills, Alberta[a]

	Summer		Fall	
Creek	SRP	$NO_2^- + NO_3^-$	SRP	$NO_2^- + NO_3^-$
Battle	51.0	3.1	49.3	3.8
Graburn	28.6	48.2	35.5	259.4
Nine Mile	32.1	7.0	43.6	7.7
Storm	—	—	47.1	983.8

[a]Data are average concentrations based on values obtained when NDS were deployed and retrieved. —no data (NDS not deployed).

cover did not differ consistently among the three livestock-present treatments. We expect that the predominance of nondetectable differences among the three livestock-present treatments arise from preferential use of riparian areas by livestock for forage, water, and rest compared with upland areas. For example, Platts and Nelson (1985) reported that streamside forage was more heavily utilized than adjacent range forage in Idaho. The absence of differences in bank and riparian conditions may arise because livestock preferentially graze riparian and instream vegetation prior to utilizing adjacent upland areas and that cattle can largely deplete these forage sources within two months.

Water Chemistry

Livestock grazing can increase concentrations of N and P in streams by: (1) increasing the propensity of overland flow during precipitation events; (2) decreasing denitrification rates within the riparian zone; (3) reducing uptake by riparian vegetation; (4) increasing N and P inputs to stream channels, riparian zones and shallow groundwater from voided wastes; and (5) mobilizing N and P in stream sediments and stream banks (Platts and Nelson 1985, Platts and Rhinne 1985, Armour and others 1991, Lal 1997, Sheffield and others 1997, Quinn and others 1997). We predicted that differences in the four livestock grazing treatments would result in detectable improvements in water quality. Further, we expected that differences in the timing and duration of the early season, late season, and all-season grazing treatments would result in a gradient in grazing pressure. Our results from the two-year livestock enclosure experiment showed that livestock grazing only markedly affected TP and SRP in one of the three study streams. In general, livestock grazing did not result in detectable effects on TP, SRP, TN, $NO_2^- + NO_3^-$, or DOC. Effects on dissolved oxygen, conductivity, and turbidity were also either not detectable or relatively minor.

The absence of detectable effects of livestock grazing on stream water quality likely arises from: (1) low rates of precipitation during the study period, (2) comparatively high inputs of concentrations of SRP and $NO_2^- + NO_3^-$ from groundwater, (3) minimal establishment of trees in riparian zones of buffer and livestock-absent enclosures that could have reduced N and P inputs compared to livestock-present treatments, and (4) statistical analyses based on low sample sizes.

Low rates of precipitation during the summer–fall periods in 1999 (overall range in Battle, Graburn, and Nine Mile creeks was 135–172 mm) and 2000 (overall range 73–132 mm) also reduced the opportunity for overland flow and leaching of urine and feces deposited within riparian zones to shallow groundwater. Warm weather during the summer–fall periods reduced soil moisture conditions, and much of the precipitation that fell may have been adsorbed at the soil–air interface rather than percolating through soils and contributing to nutrient inputs to streams.

Groundwater springs were present in about half of the stream enclosures and contributed appreciable quantities of N and P to streams throughout much of the summer–fall period despite minimal local precipitation. Our finding that concentrations of SRP and $NO_2^- + NO_3^-$ in streams were unrelated to daily loadings, however, suggests that springs adjacent to streams probably comprise only a small portion of all groundwater inputs. Low to moderate concentrations of N and P in groundwater may arise from organically rich dark brown and black chernozems, dark and gray luvisols in plateau grasslands, and the presence of similar soils, legumes (*Trifoleum* spp.), and limestone-rich conglomerates adjacent to and within riparian zones. Gray luvisols are known to contain moderate amounts of phosphorus (Marston 1989), but inefficiently retain P (Xiao and others 1991) compared with other soil types (Frossard and others 1989).

Impacts of livestock grazing on water quality can be moderated by the presence of vegetated riparian buffers (Platts and Rhinne 1985, Osborne and Kovacic 1993, Lowrance and others 1997, Sovell and others 2000). It has been reported that forested vegetated riparian strips reduced concentrations of nitrogen in groundwater by 68%–100% and in surface runoff by 78%–98% (Petersen et al. 1992). Osborne and Kovacic (1993) reported that vegetated riparian buffers consisting of grass or forest reduced concentrations of nitrate in shallow groundwater by up to 90%. Pinay and others (1998) also reported significant decreases in groundwater nitrate along groundwater flowpaths crossing the riparian zones during the growing season.

Our results from livestock enclosures showed significantly greater grass biomass in livestock-absent enclosures (i.e., buffers and the no-livestock grazing treatment) compared to livestock-present treatments and adjacent areas accessible to livestock. However, exclusion of livestock did not result in appreciable increases in density of willow or aspen within riparian zones although some regeneration was observed (Korpela 2001). The extent to which the reestablishment of willow, trembling aspen, and grasses reduce N and P in shallow groundwater is unknown and can not be evaluated without a more detailed assessment of groundwater flow paths (Pinay and others 1998) and the extent to which groundwater is modified within the hyporheic zone.

We quantified longitudinal patterns in SRP and $NO_2^- + NO_3^-$, expecting that the all-season livestock grazing treatment would result in substantial increases in nutrients and that such increases could affect longitudinal patterns in nutrient availability. Increased concentrations of SRP and $NO_2^- + NO_3^-$ could also affect epilithon if epilithon biomass is strongly related to the availability of SRP and $NO_2^- + NO_3^-$. Our data showed that nutrient levels were typically highest in upper watershed reaches and declined dramatically with distance downstream. However, the all-season grazing treatment cannot account for the elevated nutrient levels in upper watershed of Battle and Graburn creeks because concentrations of SRP and $NO_2^- + NO_3^-$ in the all-season grazing treatment were not appreciably higher than that in the livestock-absent or early and late season grazing treatments located immediately upstream. High nutrient levels in the upper watersheds probably arises because they are dominant recharge sites of groundwater that contain high concentrations of SRP and moderate, but variable, concentrations of $NO_2^- + NO_3^-$. To some extent, declines in nutrient levels could also result from sequestering of nutrients in beaver dams (e.g., Devito and Dillon 1993), which are increasingly abundant in lower reaches, combined with high concentrations of dissolved oxygen (7 mg/liter), reducing the likelihood of nutrient release from anoxic sediments during the summer–fall period. Alternatively, increased densities of willows lower in the watershed could remove appreciable an amount of nutrients from shallow groundwater that would otherwise enter stream channels.

The lack of detectable effects of livestock grazing on stream water chemistry could also arise from low sample sizes, especially after the removal of autocorrelated data from analyses. Irrespective of the cause, our results suggest that detecting the effects of livestock grazing on water quality in hydrologically complex systems, such as the Cypress Hills Plateau, using a within-watershed approach, is inherently difficult and that mechanistic-based approaches focusing on providing detailed an understanding of nutrient inputs, transformations, and exports may prove to be a more rigorous approach.

Epilithon

Comparisons of benthic Chl *a* at Graburn Creek showed variable and nondetectable effects of livestock grazing in 1999 but significantly lower epilithic Chl *a* in the all-season grazing treatment compared with other treatments in September 2000. Lower epilithic Chl *a* in the all-season treatment compared to other treatments was also observed in Battle Creek although differences were not statistically significant. While the actual mechanism is unknown, reduced algal biomass in the all-season grazing treatments could arise from physical disruption of algal communities because livestock appeared to spend a considerable amount of time in streams drinking or grazing instream vegetation.

Increased concentrations of SRP and $NO_2^- + NO_3^-$ could also affect epilithic biomass dynamics if epilithic Chl *a* is strongly related to the availability of SRP and $NO_2^- + NO_3^-$. Our comparisons at the watershed scale indicated that epilithic biomass was moderately well related to SRP and $NO_2^- + NO_3^-$ ($r^2 = 0.47$–0.63) in two streams and unrelated to SRP and $NO_2^- + NO_3^-$ in the remaining two streams. Spatial variation in irradiance, abundance of invertebrate scrapers, and subtle variation in water depth and velocity among sampling sites likely explains why empirical relationships were not more powerful (e.g., Rosemond 1993, Bourassa and Cattaneo 1998, Biggs 2000).

Potential reductions in stream N and P concentrations due to the establishment of riparian vegetation (grasses, shrubs, and trees) have important consequences to algal biomass dynamics. Our data indicate marked declines in SRP and $NO_2^-+NO_3^-$ from headwater sites to locations 5–14 km downstream and that longitudinal variation in benthic Chl *a* in Graburn and Nine Mile creeks were significantly related to concentrations of SRP and $NO_2^-+NO_3^-$. While results from nutrient-diffusing substrata indicate seasonal changes in nutrient limitation status (Francoeur and others 1999, Wold and Hershey 1999), algal communities in Battle and Nine Mile creeks were N limited in summer and fall whereas communities Graburn Creek shifted from nutrient-replete conditions in summer to N limitation later in the year. These results suggest that reductions in N and P inputs in headwater reaches could result in nutrient limitation throughout much of the Battle, Graburn, and Nine Mile watersheds. Our understanding of the effects of cattle grazing on algal biomass in the Cypress Hills is incomplete because the extent to which increases in algal accrual, due to removal of riparian vegetation and increased light and water temperatures, may be offset by physical disruption of epilithic mats by livestock is poorly understood.

Management Implications

Results of the present study suggest that alternative livestock grazing practices need to be developed in the Cypress Hills to improve bank stability and to restore riparian and instream vegetation. Even though our overall sampling design was generally insufficient to detect differences in water quality among the four livestock grazing treatments perhaps due to low statistical power, changes in livestock grazing practices could still result in improvements in water quality.

Reduced use of riparian zones by livestock in the Cypress Hills Provincial Park could be accomplished by excluding livestock (i.e., fencing riparian areas), reducing stocking densities, or adopting deferred or rest-rotation, grazing practices (Platts and Nelson 1985, Humphrey and Patterson 2000, Sovell and others

2000). The development of off-stream water supplies (e.g., Godwin and Miner 1997, Sheffield and others 1997) and placement of salt blocks at sites located away from streams should also reduce use of riparian zones.

Our data indicate that neither the early nor late season grazing treatments resulted in appreciable improvements in bank stability, riparian biomass, or the extent to which stream channels were vegetated. These results suggest that a deferred grazing approach, where livestock are excluded from riparian areas during early or late summer, would unlikely result in appreciable improvements in bank stability or vegetation communities in riparian zones or stream channels. While total exclusion of livestock from riparian zones is an option, establishing and maintaining fences is costly, would require the development of off-stream watering, and is generally viewed as an option only if all other approaches fail. Reducing overall stocking densities in the Cypress Hills Provincial Park is also problematic because livestock use riparian areas disproportionately compared with plateau grasslands. Thus, overall stocking units would have to be reduced greatly before they translated into an appreciable reduction in the use of riparian zones.

In contrast, a rest-rotation approach may be acceptable to ranchers, environmentalists, and park administrators for ecological and economic reasons. First, provincial and federal parks in Canada are often managed within a natural disturbance paradigm where disturbances, such as fire, are recognized as natural processes that maintain ecological integrity (Scrimgeour and Wicklum 1996). The plateau grasslands of the Cypress Hills are known to have evolved with herbivory from bison (*Bison bison*). One view is that resident populations migrated throughout the plateau grasslands and slopes throughout the year and may have resulted in some areas being grazed more intensely in some years compared with others. While livestock are currently removed from the Cypress Hills during the winter, primarily because of difficulties in providing an accessible water supply, installation of frost-free, nose pumps could allow grazing during the winter. Thus, a yearlong, rest-rotation approach where livestock are rotated among large fields at three- to six-year intervals could be used to approximate patterns in herbivory by bison. The rotation period would also allow establishment of grassy riparian zones. From a financial perspective, the rest-rotation approach is also appealing because it ensures financial returns to local ranchers who rely on income from livestock grazing in the Cypress Provincial Park. Adoption of a natural disturbance based model requires a better understanding of how bison historically used the grasslands in the Cypress Hills and the adjacent plains grasslands.

A disturbance-based rest-rotation model designed to mimic herbivory by bison, however, is fundamentally different from developing a best management practice model that is adopted to minimize environmental effects of livestock grazing. If stocking rates are high, as could be expected if based on historical densities of bison, the rest-rotation approach would result in periods where riparian and stream habitats would be degraded when livestock are present, followed by periods of recovery after livestock have been removed. Proponents of natural disturbance-based models would consider these changes to be acceptable because to some extent, they mimic natural processes. Irrespective of the approach, the ability to develop manage the Cypress Hills Provincial Park may be strongly affected by the extent to which park administrators, ranchers, and environmentalists agree on what ecosystem attributes need to be protected, rather than which management model is most appropriate.

Acknowledgments

Unless otherwise stated, the interpretation and statements contained in this document are those of the authors. We thank the Cypress Hills Research Steering Group for their input on the design and objectives of the study. Beverley Gingras, Dan Sturgess, Ross Warren, Dayle Soppet, and Elsabe Kloppers assisted with fieldwork, and Michelle Hiltz provided advice and completed some of the statistical analyses. Greg Wagner (University of Calgary, Calgary, Alberta) assisted with transportation. Annual precipitation data for Elkwater were kindly provided by Jon DeWalt (Alberta Environment).

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