Appendix 10



# Long-term response of periphyton and macrophytes to reduced municipal nutrient loading to the Bow River (Alberta, Canada)

**Al Sosiak**

**Abstract:** The biomass of periphyton and aquatic macrophytes (*Potamogeton vaginatus* and *Potamogeton pectinatus*) in the Bow River was sampled over 16 years to assess the response of these plants to improved phosphorus (1982–1983) and nitrogen removal (1987–1990) at Calgary's two municipal wastewater treatment plants. These improvements in treatment reduced total phosphorus loading to the Bow River by 80%, total ammonia loading by 53%, and nitrite + nitrate loading by 50%. No change in periphytic biomass was detected after enhanced phosphorus removal where total dissolved phosphorus (TDP) in river water remained relatively high (10–33 $\mu g \cdot L^{-1}$). However, periphytic biomass declined at sites further downstream with TDP < 10 $\mu g \cdot L^{-1}$. Regression analysis predicted that nuisance periphyton biomass (>150 $mg \cdot m^{-2}$) occurred at TDP > 6.4 $\mu g \cdot L^{-1}$ (95% confidence interval: 1.9–7.6 $\mu g \cdot L^{-1}$). Macrophyte biomass was inversely correlated with discharge and was lower during high-discharge years. Biomass also declined following enhanced nutrient removal, with the greatest decrease following reduced nitrogen discharge. These results provide the first evidence for a response of periphyton and aquatic macrophytes to enhanced nutrient removal from municipal wastewater.

**Résumé** : La biomasse du périphyton et des macrophytes aquatiques (*Potamogeton vaginatus* et *Potamogeton pectinatus*) a été mesurée sur une période de 16 ans dans la rivière Bow pour déterminer la réaction de ces plantes à l'amélioration du retrait du phosphore (1982–1983) et de l'azote (1987–1990) par les deux usines municipales de traitement des eaux usées de Calgary. Ces améliorations de traitement ont réduit l'apport de phosphore total à la rivière Bow de 80 %, celui de l'ammoniaque total de 53 % et celui des nitrites + nitrates de 50 %. Aucun changement n'a pu être détecté dans la biomasse du périphyton après le retrait amélioré du phosphore lorsque la concentration de phosphore dissous total (TDP) est demeurée relativement élevée (10–33 $\mu g \cdot L^{-1}$) dans l'eau de la rivière. Cependant, la biomasse du périphyton a baissé à des sites plus en aval où la concentration de TDP était < 10 $\mu g \cdot L^{-1}$. Une analyse de régression permet de prédire que la biomasse de périphyton qui est considérée nuisible (>150 $mg \cdot m^{-2}$) apparaît aux concentrations de TDP > 6,4 $\mu g \cdot L^{-1}$ (intervalle de confiance de 95 % : 1,9–7,6 $\mu g \cdot L^{-1}$). La biomasse des macrophytes est en relation inverse avec le débit et est plus faible les années de fort débit. La biomasse a aussi diminué après le retrait plus efficace des nutriments; la diminution la plus importante s'est produite à la suite de la réduction de l'apport d'azote. Ces résultats constituent les premières indications que le périphyton et les macrophytes aquatiques réagissent à un retrait plus efficace des nutriments dans les eaux usées municipales.

[Traduit par la Rédaction]

## Introduction

Nutrients from municipal wastewater discharges have caused eutrophication of lakes and rivers across Canada. To reduce this nutrient discharge, tertiary treatment has been installed at municipal wastewater treatment plants (WWTPs) serving about 34% of the Canadian population (Chambers et al. 1997). Reductions in phosphorus loading from municipal wastewater have controlled or reversed eutrophication in various Canadian lakes (reviewed in Chambers et al. 1997). However, there have been fewer studies of the response of aquatic plants in flowing water to reduced municipal nutrient loading. Although the response of periphyton to nutrient enrichment has been examined in stream channel experiments (Bothwell 1989; Horner et al. 1990), no studies appear to have evaluated the response of periphyton to reduced municipal loading in natural stream ecosystems. Chambers (1993) found no evidence of reduced growth of aquatic macrophytes in the South Saskatchewan River owing to reduced municipal phosphorus loading from Saskatoon. Terrell and Canfield (1996) reported similar findings after the elimination of both phosphorus and nitrogen discharge from a WWTP to a cove on a Florida river.

The Bow River is a major tributary of the Saskatchewan–Nelson river system, which originates in the Rocky Mountains of southwestern Alberta, Canada. Nutrient levels are low (Table 1) and little growth of submersed macrophytes occurs upstream from the two WWTP outfalls in Calgary (M1, Fig. 1). However, lush growth of periphyton, giant pondweed (*Potamogeton vaginatus*), and sago pondweed (*Potamogeton pectinatus*) occurred downstream of the WWTP outfalls in coarse gravel to cobble substrate

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**A. Sosiak.** Science and Standards Division, Alberta Environment, Room 301, 2938 11 St. NE, Calgary, AB T2E 7L7, Canada (e-mail: Al.Sosiak@gov.ab.ca).

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**Table 1.** Physical,[a] chemical, and biological characteristics of selected Bow River sampling sites, May–October 1979–1996.

| | | Bowness | | Stier's Ranch | | Ronalane | |
|---|---|---|---|---|---|---|---|
| Variable | Units | Mean | Range | Mean | Range | Mean | Range |
| Depth | m | 0.64 | — | 0.65 | — | NA | — |
| Width | m | 122 | — | 59 | — | NA | — |
| Current velocity | $m \cdot s^{-1}$ | 1.3 | — | 2.6 | — | NA | — |
| Duration of ice cover[b] | days | 148 | October 22 – May 2 | Rarely freezes | — | 136 | October 15 – April 20 |

| | | Bowness | | Stier's Ranch | | Ronalane | |
|---|---|---|---|---|---|---|---|
| Variable | Units | Median | Range | Median | Range | Median | Range |
| Bed material size | mm | 100 | — | 14 | 14–300[c] | 18 | — |
| Daily mean discharge | $m^3 \cdot s^{-1}$ | 96.7 | 39.1–387 | 95.8 | 43.9–450 | 57.7 | 12.2–817 |
| Ammonia, total as N | $mg \cdot L^{-1}$ | 0.02 | 0.01–0.10 | 0.20 | 0.02–1.50 | 0.05 | 0.01–0.26 |
| Nitrite + nitrate, dissolved as N | $mg \cdot L^{-1}$ | 0.030 | 0.003–0.100 | 0.680 | 0.180–2.00 | 0.03 | 0.003–0.98 |
| Nitrogen, total as N | $mg \cdot L^{-1}$ | 0.22 | 0.02–2.32 | 1.28 | 0.50–5.76 | 0.53 | 0.20–7.81 |
| Phosphorus, total dissolved as P | $mg \cdot L^{-1}$ | 0.003 | 0.003–0.040 | 0.026 | 0.003–0.280 | 0.011[d] | 0.003–0.090 |
| Phosphorus, total as P | $mg \cdot L^{-1}$ | 0.008 | 0.003–0.121 | 0.049 | 0.006–0.350 | 0.030 | 0.007–0.645 |
| Silica, reactive | $mg \cdot L^{-1}$ | 3.10 | 2.10–4.59 | 2.80 | 0.01–5.50 | 0.60 | 0.01–5.10 |
| Turbidity | NTU | 1.8 | 0.2–8.0 | 3.7 | 0.3–45.0 | 8.5 | 0.5–520 |
| Water temperature | °C | 12.5 | 0.8–18.2 | 13.0 | 2.8–21.0 | 17.0 | 1.4–25.1 |
| Macrophyte biomass | $g \cdot m^{-2}$ | Absent | — | 425.5 | 0–3897[c] | 69.9 | — |
| Periphytic Chl *a* | $mg \cdot m^{-2}$ | 25.1 | 0.7–233.2 | 152.4 | 3.4–889.7 | 79.5 | 3.9–493.3 |

**Note:** NA, not available; NTU, nephelometric turbidity units.
[a]Source: River Engineering Branch, Alberta Environment, 9th Floor, 9820-106 Street, Edmonton, AB T5K 2J6, Canada.
[b]Source: Kellerhals et al. (1972).
[c]Range at macrophyte sampling sites M1–M8.
[d]Median TDP concentration following enhanced phosphorus removal (1990–1996): 0.005 $mg \cdot L^{-1}$.

(Table 1) (Charlton et al. 1986). Drifting aquatic macrophytes and algae clogged intakes for irrigation systems, interfered with boating and angling (Alberta Environment 1979) and caused low levels of dissolved oxygen at night (Cross et al. 1986). Few submersed macrophytes were found downstream from Carseland (Fig. 1), except below the Bassano Reservoir, during a basin-wide survey by Alberta Environment (AENV) (A. Sosiak, unpublished data).

In 1982–1983, the City of Calgary installed chemical phosphorus removal at both the Bonnybrook (near site M2, Fig. 1) and the Fish Creek WWTPs (site M5, Fig. 1; new total phosphorus (TP) approval limit: 1 $mg \cdot L^{-1}$) and later added biological removal. During the next 6 years, macrophyte biomass declined at some sampling sites, but periphytic biomass did not change significantly (Sosiak 1990). Although phosphorus levels had been reduced, there was concern that ammonia in the Bow River could reach toxic levels with the rapid population growth that has occurred in Calgary (Fig. 2). The City of Calgary then also decreased ammonia and nitrate discharge to the Bow River from both WWTPs by modifications to the activated sludge process (1987), increased aeration (1990), and biological nitrogen removal (in stages beginning in 1990). The Bonnybrook WWTP (alone) must comply with a new ammonia limit of 5 $mg \cdot L^{-1}$ during July–September and 10 $mg \cdot L^{-1}$ during other months.

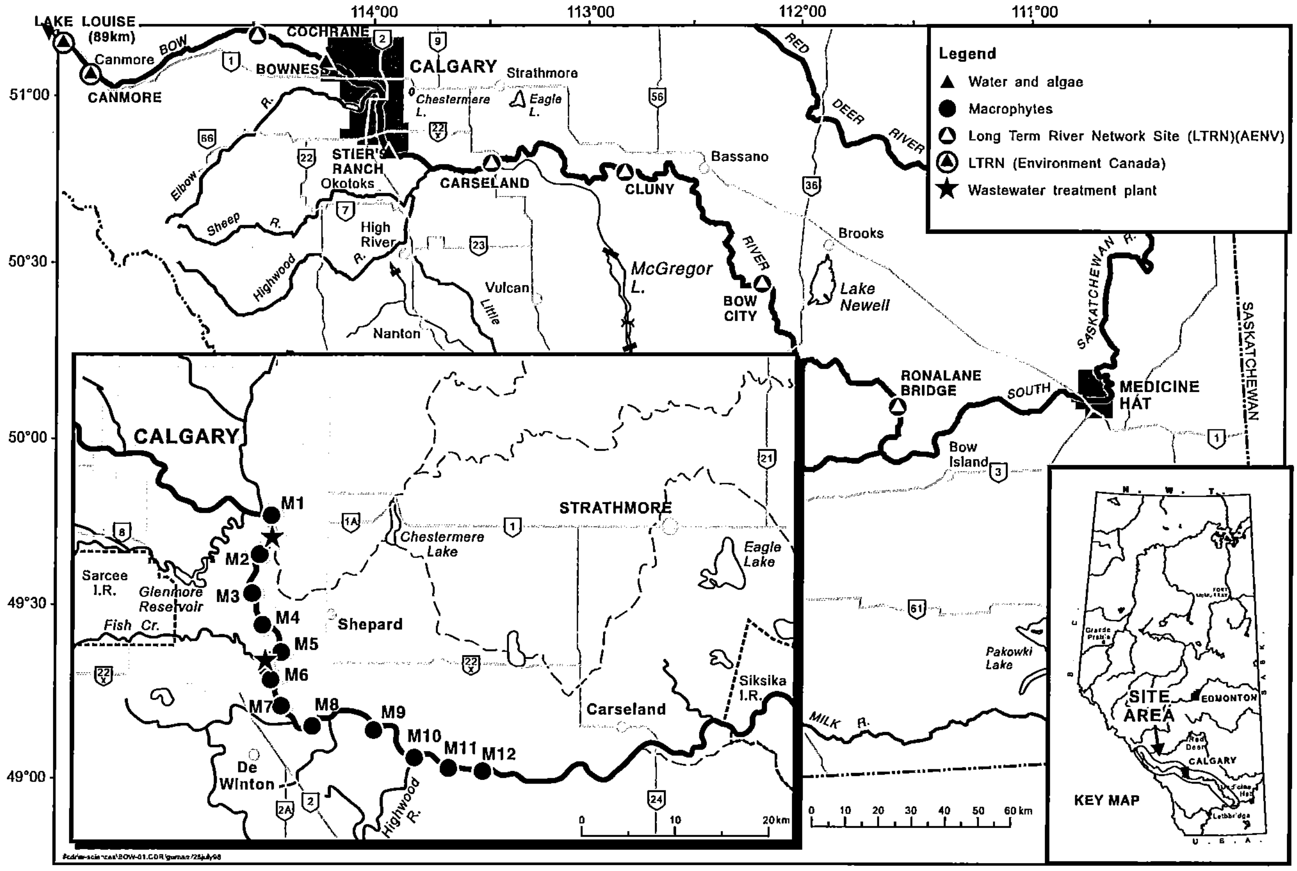
Aquatic macrophyte and periphytic biomass, as dry weight and chlorophyll *a* (Chl *a*), respectively, were sampled over 16 years by AENV to determine the long-term response of these aquatic plants in the Bow River to reduced nutrient discharge from the WWTPs in Calgary. In this paper, I first evaluate changes in nutrients and aquatic plant biomass following enhanced phosphorus removal and then the response to both phosphorus and nitrogen removal. I then develop predictive equations for aquatic plant biomass.

## Materials and methods

### Sampling sites and methods

Grab samples for chemical constituents and field measurements (dissolved oxygen, pH, conductivity, temperature) were collected at least monthly at four sites upstream from WWTP discharge in Calgary and at four sites downstream from WWTP discharge during 1979–1996 (Fig. 1). The site at Stier's Ranch was the first fully mixed location downstream from the WWTP discharge. The site at Cluny was only sampled during 1985–1996. Most sites were long-term river monitoring sites (LTRN) sampled by AENV for a wide range of variables; however, two headwater sites were sampled by Environment Canada. During 1980–1986, the duration and frequency of periphyton sampling varied each year among sites ($n$ = 6–22). After 1986, periphyton at all sites and other variables at Bowness and Stier's Ranch were only sampled during May–October ($n$ = 6), the time period when shoreline areas at all sites are always ice free. Single composite samples of surficial sediments were collected each fall from sites at Bowness, Stier's Ranch, and Ronalane during

**Fig. 1.** Bow River sampling sites in Alberta, Canada.



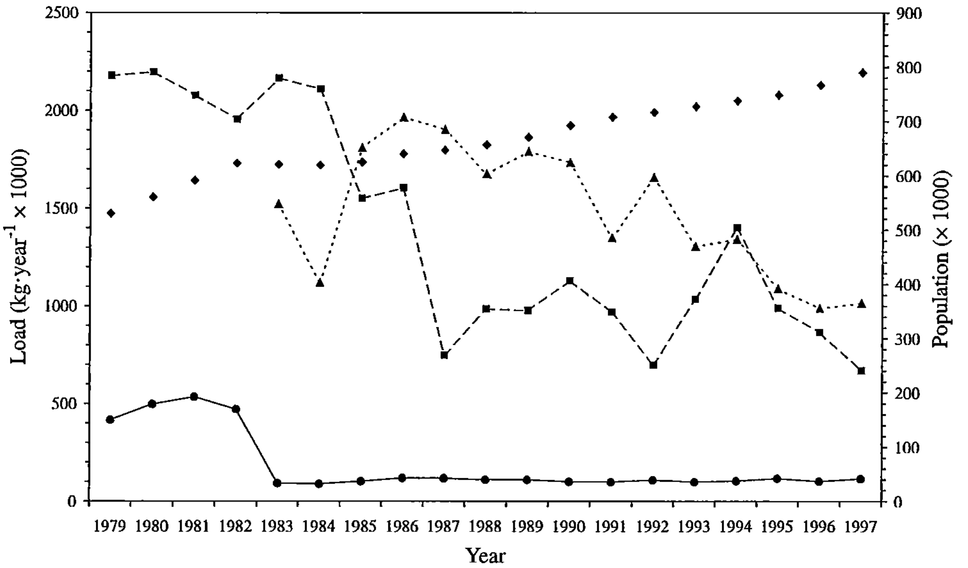

1981 and 1987–1996 using an airlift sampler (Cross et al. 1986), which used nitrogen gas pulsation to suspend fine sediments from the mainly cobble and boulder riverbed. Sediment samples were analyzed for forms of phosphorus but not nitrogen.

Mean daily discharge on sampling days and monthly mean discharges were provided by the Water Survey of Canada from stage recorders on the Bow River at Lake Louise (Water Survey of Canada Station No. 05BA001), Banff (05BB001), Ghost Dam (05BE006), Bearspaw (05BH008), Calgary (05BH004), Carseland (05BM002), Bassano Dam (05BM004), and near the mouth of the river (05BN012). Because there is no flow gauge immediately below Calgary, daily and monthly discharges were calculated from the sum of discharges in the Bow River at Calgary, the Elbow River below Glenmore Dam (05BJ001), Fish Creek at Priddis (05BK001), and City of Calgary WWTP return flows minus diversion to the Western Headworks Canal (05BM003) plus the Highwood River near the mouth (05BL024) for macrophyte sites below the Highwood River.

Epilithon (periphyton growing on stones or similar objects) was routinely sampled during 1980–1996 from Bowness to Ronalane (Fig. 1), a distance of 371 km. The routine sampling site for periphyton upstream from the WWTP discharge at Bowness (sampled in 1980, 1981, and 1993) was moved 5 km downstream in 1983 to Bowness Park (Fig. 1), where all other variables were sampled during 1979–1996. During 1980–1983, all periphytic samples for Chl *a* analysis were collected by brushing 9–12 rocks of known surface area, randomly collected from areas at wading depth, and filtering a single subsample of slurry through a glass fiber filter (Whatman GF/C). The surface area of these rocks was determined so that results could be expressed per unit area. The material retained on the filter was covered with anhydrous magnesium carbonate and stored at –4°C. During 1984–1996, periphyton samples were collected from nine rocks by scraping a 4-cm$^2$ area delimited by a template with a scalpel. Three samples, each a composite of material from three rocks, were collected per site on each sampling date and analyzed separately. Additional periphytic samples were collected for species identification using the same method monthly during 1981 and 1982 and quarterly in 1994.

In early September 1981–1996, five aquatic macrophyte samples were randomly collected from right and left bank sites at 12 shoreline sampling locations located from 2.5 km upstream to 33.0 km downstream of both municipal plants (Fig. 1). Sites were spread over 50.3 km, or 69% of the macrophyte distribution (A. Sosiak, unpublished data). Previous studies indicated that maximum biomass occurred in mid- to late September (Charlton et al. 1986). Plants within a 30.48 × 30.48 cm frame were uprooted and collected in a 3-mm-mesh sampling net. In the laboratory, debris was removed and samples less than 200 g wet weight were oven-

**Fig. 2.** Combined mass load of TP, nitrate, and total ammonia contributed to the Bow River from the Calgary WWTPs compared with the City of Calgary's population, 1979–1997. Circles, TP; triangles, $NO_3$-N; squares, $NH_3$-N; diamonds, Calgary's population.




dried at 105°C for 24 h. Samples over 200 g were centrifuged for 1 min to remove excess moisture and then weighed and converted to dry weight by dividing wet weight by 7.73 (Charlton et al. 1986). This conversion factor was periodically verified.

### Sample analysis

Analytical techniques and detection limits for all variables tested statistically in this study are given in Table D1.[1] Chemical constituents were analyzed at the Water Quality Laboratory of Environment Canada in Calgary during 1979–1984. After this laboratory was relocated in 1984, all chemical constituents sampled by AENV were analyzed by Chemex Labs Alberta, Inc., of Calgary (now Maxxam Analytics, Inc.). In all years, periphytic pigments were extracted in 90% acetone at the Monitoring Branch, AENV laboratory in Edmonton, homogenized, and analyzed using spectrophotometry with corrections for total pheophytin.

The City of Calgary (C. Collins, Bonnybrook Wastewater Treatment Plant, 4302-15th St. SE, Calgary, AB T2G 3M8, Canada, personal communication) provided census data and nutrient discharge estimates for both municipal WWTPs during 1979–1997. Nitrate was not measured at the City of Calgary laboratory before 1983.

Periphytic species were identified and counted as single-celled individuals, filaments, or colonies and biomass estimates were prepared by Dr. M. Agbeti of Bio-Limno Research and Consulting (Edmonton, Alta.). Aliquots of diluted samples were allowed to settle overnight in sedimentation chambers. A minimum of 500 algal cells or colonies was counted for each sample using an inverted microscope. Algal wet biomass was calculated from recorded abundance and published cell biomass values from other rivers.

### Data analysis

The water quality data collected in this study were often not normally distributed and exhibited significant seasonality, serial correlation, and flow dependency. Accordingly, nonparametric statistical methods that compensate for these factors were used in all analyses except regression following procedures in the water quality statistics package WQHYDRO (Aroner 1995). Trend analysis was used to determine whether water chemistry changed significantly following enhanced nutrient removal and whether macrophyte and periphytic biomass declined as anticipated.

Except for macrophytes, which were only sampled once per year, time periods before and after changes in wastewater treatment were first tested for seasonality using the Kruskal–Wallis test for seasonality. Step trend analysis was then used to test whether enhanced phosphorus and nitrogen removal at the WWTPs resulted in immediate incremental changes in plant biomass and other variables. Data with significant seasonal variation were tested for step trends using the seasonal Wilcoxon–Mann–Whitney test (time intervals in Table 2), and changes in medians were calculated using the seasonal Hodges–Lehman estimator. For data without significant seasonal variation, the Anderson–Darling analysis of variance, followed by an experimentwise Kruskal–Wallis multiple comparison test, was used to test step trends. The Anderson–Darling analysis of variance is not adversely affected by differences in variance between samples, which often occurred in these data.

All variables were then tested for monotonic trends, or gradual increasing or decreasing biomass or concentration, following the installation of enhanced phosphorus removal in 1982–1983. Data with significant seasonal variation were tested for monotonic trends using the seasonal Kendall test, with correction for significant serial correlation as required. Serial correlation, or correlation between consecutive observations, was tested using Kendall tau correlation analysis on a time series with trend and seasonality removed. Data that did not display significant seasonal variation were tested for

[1] Copies may be purchased from the Depository of Unpublished Data, CISTI, National Research Council of Canada, Montreal Road, Ottawa, ON K1A 0S2, Canada.

**Table 2.** Incremental changes (step trends) in selected variables in the Bow River after the installation of enhanced phosphorus and nitrogen removal at Calgary WWTPs.

| | Significant change in discharge, biomass, or concentration[a] | | | | | | |
|---|---|---|---|---|---|---|---|
| Site | Discharge ($m^3 \cdot s^{-1}$) | Macrophyte biomass ($g \cdot m^{-2}$)[b] | Periphytic Chl *a* ($mg \cdot m^{-2}$) | TP ($mg \cdot L^{-1}$) | TDP ($mg \cdot L^{-1}$) | Ammonia ($mg \cdot L^{-1}$) | Nitrite + nitrate ($mg \cdot L^{-1}$) |
| **Changes after the installation of enhanced phosphorus removal[c]** | | | | | | | |
| Bowness, macrophyte site 1 | −6.20 | NS | NS[d] | NS | +0.002 | NA | NS |
| Stier's Ranch, macrophytes (sites 2–12) | −11.85 | RB: −177<br>LB: −273 | NS | −0.188 | −0.144 | +0.150 | NS |
| Carseland | −12.60 | — | NS | −0.117 | −0.110 | −0.035 | NS |
| Bow City | −21.63 | — | NS | −0.108 | −0.028 | NS | NS |
| Ronalane | −20.35 | — | NS | −0.046 | −0.021 | NS | NS |
| **Changes after enhanced nitrogen removal[e]** | | | | | | | |
| Bowness, macrophyte site 1 | +18.95 | +72.6 | NS | NS | NS | −0.005 | +0.013 |
| Stier's Ranch, macrophytes (sites 2–12) | +16.70 | RB: −329<br>LB: −354 | NS | NS | NS | −0.073 | −0.229 |
| Carseland | +26.15 | — | −82.08 | −0.005 | −0.005 | −0.030 | −0.210 |
| Cluny | +21.60 | — | −97.46 | NS | −0.005 | NS | −0.132 |
| Bow City | +34.95 | — | −15.03 | NS | −0.004 | −0.005 | NS |
| Ronalane | +19.72 | — | −47.41 | NS | −0.003 | NS | NS |

**Note:** Bowness site is upstream from the Calgary WWTPs and the other sites are downstream. NS, not significant; NA, data not tested because of change in detection limits; RB, right bank facing downstream; LB, left bank facing downstream.

[a]Seasonal or nonseasonal Hodges–Lehmann estimator.

[b]Median of estimates from sites with significant change in biomass.

[c]Intervals tested: macrophytes, 1981–1982 versus 1983–1989; discharge and chemistry, 1979–1982 versus 1983–1985.

[d]Site 05BH0110 at Bowness; flow-adjusted and tested 1980, 1981 versus 1993.

[e]Intervals tested: macrophytes, 1983–1989 versus 1990–1996; discharge and chemistry, 1983–1988 versus 1989–1996.

monotonic trends using the Mann–Kendall test. Sen slopes were calculated to provide an estimate of the approximate magnitude of significant monotonic trends.

Intermediate trend statistics were calculated for monthly median values and then combined in an annual statistic. A $\chi^2$ test of trend homogeneity was used to ensure that monthly trends followed one consistent direction over the year. Where trends were not homogeneous, trend tests were calculated just for the May–October time period, which had consistent homogeneity of trends. As recommended by Ward et al. (1990), a 0.10 level of significance was used in all trend analysis.

Previous work (Chambers et al. 1991) found that current velocity was an important factor regulating macrophyte biomass in the Bow River. Since there was a small but significant increase in discharge at the macrophyte sampling sites during 1983–1996, which could scour macrophytes and partially account for declining biomass, macrophyte data were also adjusted for discharge and again tested for trends. Macrophyte biomass (dependent variable) each year was flow-adjusted by robust regression (Aroner 1995) against total discharge over the season (independent variable), and residuals from the model with the best fit were tested for trends. This procedure is based on the assumption that variations in discharge do not appreciably affect the physiological response of macrophytes to nutrients and other variables. Periphytic Chl *a* from the old sampling site at Bowness was also flow-adjusted to compensate for the large variation in discharge during the three years (1980, 1981, and 1993) that this site was sampled. Variables were not otherwise flow-adjusted prior to analysis.

Spearman rho rank correlation was used to determine whether periphytic Chl *a* and macrophyte biomass were significantly correlated with physical and chemical variables. Results for the various sites from Bowness to Ronalane were merged for correlation and regression analysis. Maximum periphytic Chl *a* during May–October 1980–1996 was correlated with the median concentration of other variables sampled during the same time period. Median macrophyte biomass from the different sites each year ($n$ = 5) (1981–1996) was combined and correlated with median chemistry from Bowness (macrophyte site 1), Stier's Ranch (sites 2–10), or Carseland (sites 11 and 12) and total discharge during the macrophyte growing season (May–September). To rigorously evaluate effects of discharge on maximum periphytic Chl *a*, this variable was correlated with total May–October discharge, median discharge on sampling days, and, as an index of scouring potential, the number of days each season that discharge was high enough to initiate bed movement. Bed mobility threshold flows were calculated by River Engineering Branch, AENV, using Shield's mobility number (Shields 1936) for five sites from Bowness to Bow City with sufficient bed material information. To eliminate spurious correlation due to significant mutual correlation among three variables, partial Spearman rank correlation coefficients between macrophyte biomass and chemical variables were calculated as required.

Step-up linear multiple regression analysis ($\alpha$ = 0.05) (SYSTAT version 8.0) was then used to determine the relationship between aquatic plant biomass, physical variables, and nutrients. Maximum periphytic Chl *a* and macrophyte biomass were regressed against May–October (medians) of all variables that were significantly correlated with biomass.

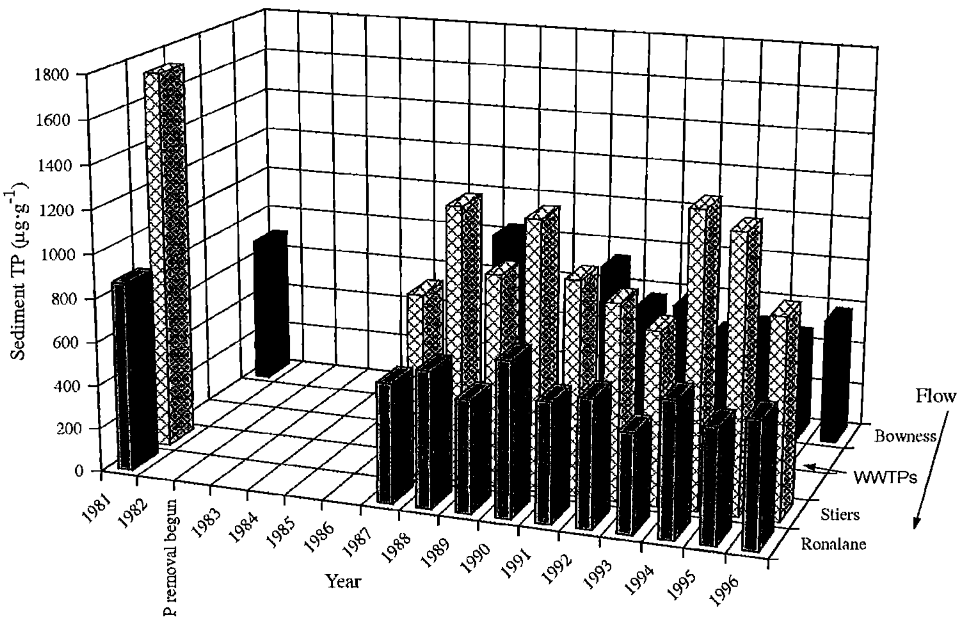
All chemical data, except the various forms of nitrogen during regression with periphytic Chl *a*, were transformed to natural logarithms (ln) to normalize the distribution of each

**Table 3.** Significant monotonic trends in periphytic Chl *a*, and physical and chemical variables in the Bow River during 1983–1996.

| | Sen slope (units/year) for significant trends (seasonal Kendal tau test, except as indicated[a]) | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Site | Discharge ($m^3 \cdot s^{-1}$) | Periphytic Chl *a* ($mg \cdot m^{-2}$) | TP ($\mu g \cdot L^{-1}$) | TDP ($\mu g \cdot L^{-1}$) | Total ammonia ($\mu g \cdot L^{-1}$) | Nitrite + nitrate ($\mu g \cdot L^{-1}$) | TN ($\mu g \cdot L^{-1}$) | Reactive silica ($\mu g \cdot L^{-1}$) |
| Lake Louise | NS | — | NS | NS | –<0.01[a] | +0.50 | +2.62 | NS |
| Canmore | NS | — | –<0.20 | –0.25 | –1.96 | +2.43 | +4.55 | +28.95 |
| Cochrane | +1.452 | — | NS | NS | –<0.01 | +2.40 | –4.88 | NS |
| Bowness | +1.901 | NS | NS | +<0.01 | –<0.01[a] | +2.11 | –13.09 | +42.67 |
| Stier's Ranch | +2.098 | NS | NS | NS | –10.01 | –47.27 | –54.44 | +62.65 |
| Carseland | +4.012 | –11.066 | NS | –0.61[a] | –2.50 | –39.81 | –59.41 | +92.99 |
| Cluny | NS | –7.904[a] | NS | –0.70 | NS | –21.31 | –30.22 | +78.08 |
| Bow City | +4.417 | –2.033[a] | NS | –0.67[a] | NS | NS | –23.27 | NS |
| Ronalane | +2.991 | –5.281[a] | NS | –0.73 | NS | NS | –16.35 | NS |

**Note:** The Bowness site is upstream from the Calgary WWTPs and the other sites are downstream. NS, not significant.
[a]Kendal tau test (nonseasonal).

**Fig. 3.** TP in surficial sediments in the Bow River, 1981–1996. WWTP notation illustrates the location of the discharge from WWTPs between the Bowness and Stier's Ranch sites.




variable. Macrophyte biomass was transformed to ln + 1 to allow for zeros, and periphytic Chl *a* was transformed using the Box–Cox transformation ($Y^{0.25}$). Normal-probability plots of residuals were inspected for linearity to determine if the assumption of normality was satisfied. Plots of residuals against predicted values were inspected for randomness to determine if the assumption of constant variance was satisfied. One outlier was removed from the macrophyte analysis. Variance inflation factors (VIF) for each independent variable were calculated to ensure that none were significant in the multiple regression equation due to appreciable multicollinearity (VIF > 10) with other independent variables and were thus redundant.

The best multiple regression equation for periphyton was then used to predict the concentration of total dissolved phosphorus (TDP) and nitrite + nitrate that caused nuisance periphytic biomass in the Bow River, here defined as maximum periphytic Chl *a* exceeding 150 $mg \cdot m^{-2}$, as recommended by Welch et al. (1988). To estimate 95% confidence intervals (CI) for these threshold concentrations, the confidence region for biomass was inverted to calculate the range of values for [TDP], and then [nitrite + nitrate], that could plausibly correspond to a periphytic Chl *a* of 150 $mg \cdot m^{-2}$. A TDP:TP regression equation was then developed to permit comparison with other TP criteria (Dodds et al. 1997).

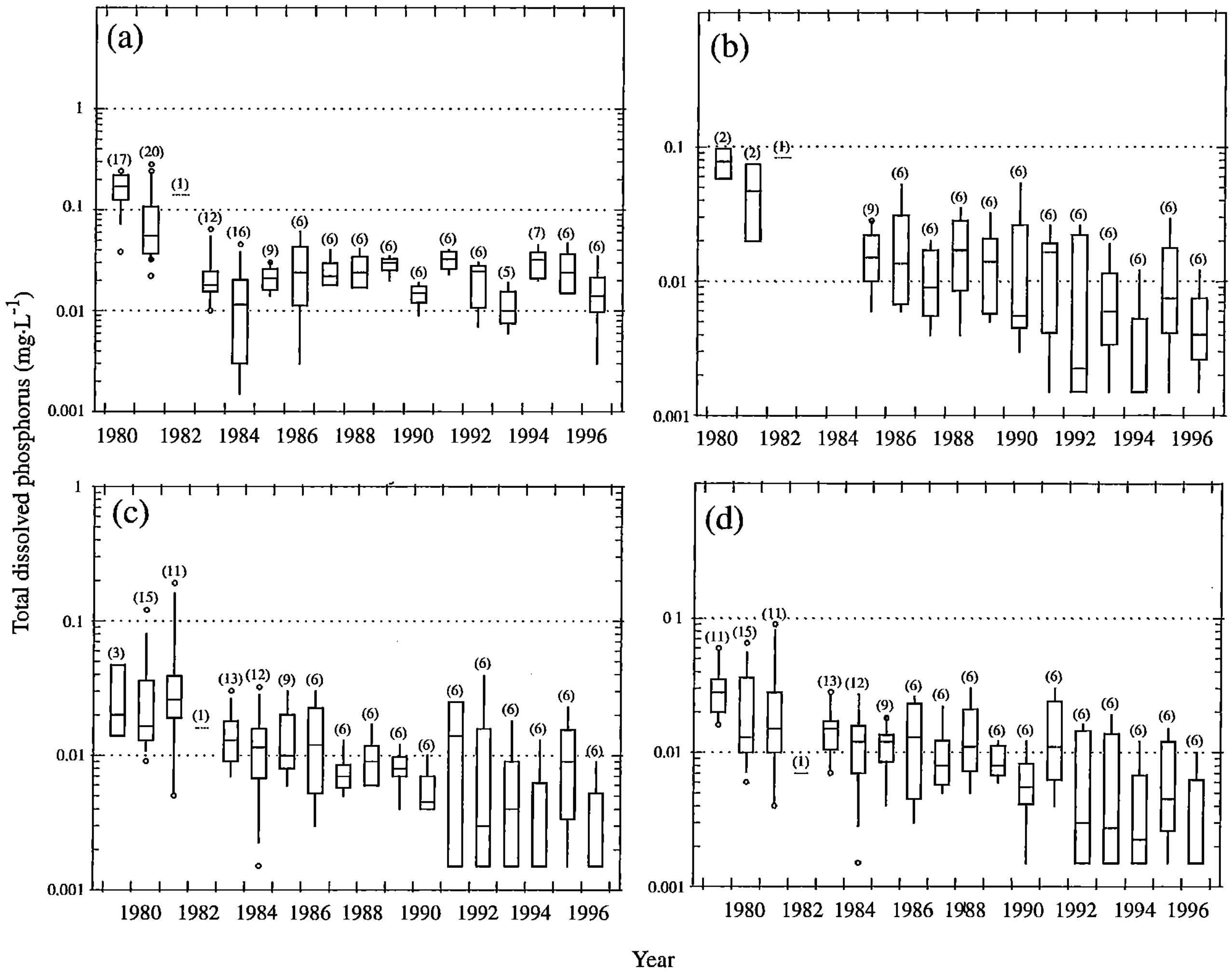
## Results

### Phosphorus

Both [TP] and [TDP] declined during 1989–1996 at Canmore, perhaps due to improved municipal treatment at Banff in 1989. Furthermore, there was a slight increase in TDP at Bowness. Otherwise, there were no significant monotonic trends in phosphorus detected at any of the four sites upstream from the municipal outfalls in Calgary (Tables 2 and 3).

TP loading to the Bow River from the two WWTPs in Calgary decreased by 80% after enhanced phosphorus re-

**Fig. 4.** TDP during May–October at selected sites: (*a*) Stier's Ranch, (*b*) Cluny, (*c*) Bow City, and (*d*) Ronalane. Bars in boxes are medians, tops and bottoms of boxes represent the 25th and 75th percentiles, and whiskers represent the 10th and 90th percentiles (values outside these percentiles plotted). In some cases, the medians coincide with the 25th percentile. Numbers above bars present sample size.




moval was installed in 1982–1983 (Fig. 2). Both [TP] and [TDP] decreased significantly in the Bow River after 1982–1983 at all sites downstream from Calgary (Table 2). TP declined in fine surficial river sediments from 1981 to 1987 at two of these sites (Stier's Ranch and Ronalane) but not upstream from the WWTPs, at Bowness (Fig. 3).

Median [TDP] (May–October) decreased 67% from 55 $\mu g \cdot L^{-1}$ in 1981 to 18 $\mu g \cdot L^{-1}$ in 1983 immediately downstream from Calgary (Stier's Ranch) but remained between 10 and 33 $\mu g \cdot L^{-1}$ (Fig. 4*a*), the highest in the basin. While there was no trend in median TDP at Stier's Ranch after 1983, TDP declined thereafter at the four sites further downstream (Table 3) and was below 10 $\mu g \cdot L^{-1}$ after 1991 at the three lower sites (Cluny, Bow City, and Ronalane) (Figs. 4*b*–4*d*). Median TDP (May–October) at Ronalane after 1991 (<5 $\mu g \cdot L^{-1}$) was at least 70% lower than in 1981 (15 $\mu g \cdot L^{-1}$).

## Nitrogen

Ammonia loading to the Bow River from the two WWTPs in Calgary was relatively steady from 1979 to 1984 and then declined by 53% over the next 3 years with improved treatment, with the greatest decrease in 1987 (Fig. 2). During May–October 1980–1989, median [total ammonia] was highest in the Bow River downstream from Calgary (Stier's Ranch), generally 0.2–0.3 $mg \cdot L^{-1}$, but thereafter the median declined there the most (step trend, Table 2). After 1989, median [total ammonia] was 0.11–0.28 $mg \cdot L^{-1}$ at Stier's Ranch (Fig. 5*a*), 0.01–0.06 $mg \cdot L^{-1}$ at Carseland, and $\leq 0.04$ $mg \cdot L^{-1}$ at all other sites.

Municipal nitrate loading to the Bow River peaked in 1986 and then gradually declined by 50% between 1986 and 1996 (Fig. 2). Median [nitrite + nitrate] in the Bow River downstream from Calgary (Stier's Ranch) peaked in 1985 at 1.2 $mg \cdot L^{-1}$ (Fig. 5*b*) and then both nitrite + nitrate and ammonia declined significantly ($P < 0.05$, monotonic trend, Table 3) at Stier's Ranch and Carseland. No significant monotonic trends in nitrite + nitrate were detected further downstream at Bow City and Ronalane. After 1989, median [nitrite + nitrate] was 0.41–0.81 $mg \cdot L^{-1}$ at Stier's Ranch and Carseland and $\leq 0.5$ $mg \cdot L^{-1}$ at all other sites. Although me-

**Fig. 5.** (*a*) Total ammonia and (*b*) nitrite + nitrate during May–October at Stier's Ranch.

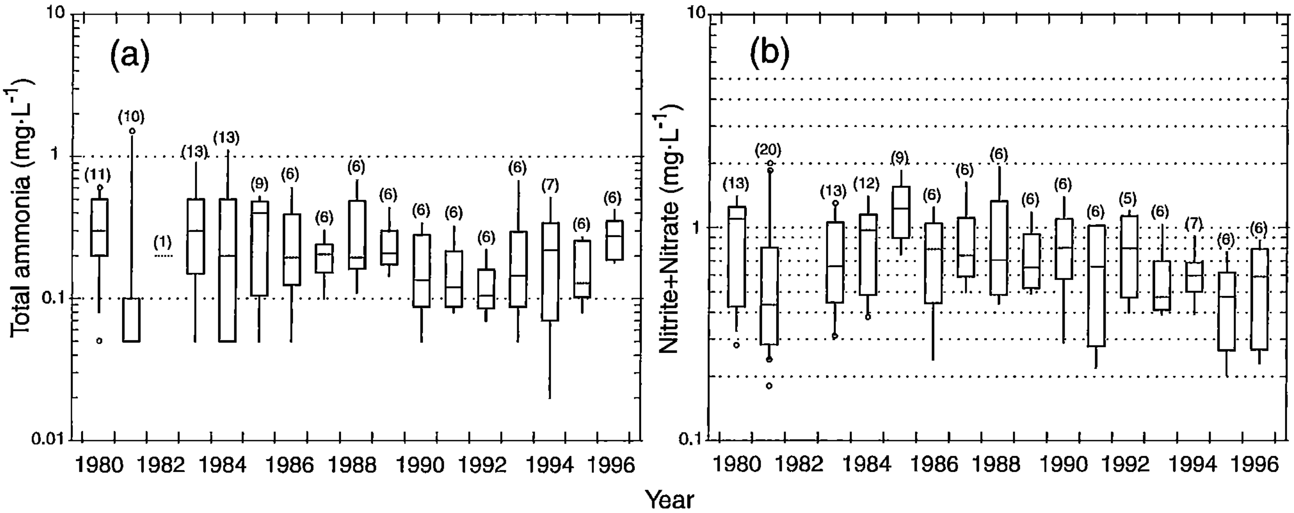
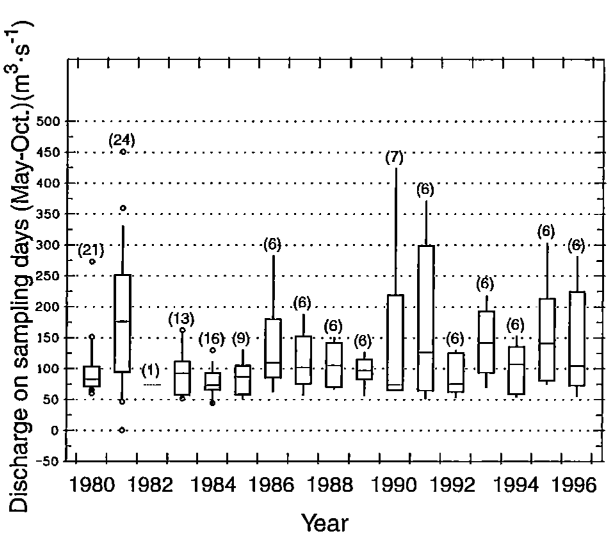



**Fig. 6.** Discharge on sampling days during May–October at Stier's Ranch.




dian biologically available N:P ratios (as ($NO_2$ + $NO_3$ + $NH_4$ + $NH_3$)/TDP) generally ranged from 6 to 12 before 1983 at all sites downstream from Calgary, most N:P ratios were well above 20 at these sites after 1989.

### Discharge

On average, the highest river discharge during the study period occurred in 1981 (Fig. 6), before enhanced nutrient removal was installed. However, discharge high enough to initiate bed movement (≥413 $m^3 \cdot s^{-1}$ at Stier's Ranch) occurred at regular intervals throughout the study period. Such discharge occurred on fewer days in the reach that includes Stier's Ranch (11 days in 1981, 1986, 1990, and 1995), in the main area of macrophyte growth, than at other sites on the lower Bow River (Carseland, Cluny, Bow City: 188–438 days during May–October 1980–1996, except 1983, 1984, and 1987). The lowest discharge occurred during a drought in 1984 and 1985. Following this drought, increasing trends in discharge were detected over the entire year during 1983–1996 at every site except Lake Louise, Canmore, and Cluny (Table 3).

### Reactive silica, temperature, and turbidity

Median [reactive silica] was relatively high (May–October median >2.0 $mg \cdot L^{-1}$) from the headwaters of the Bow River to Stier's Ranch, and there were increasing trends in silica at most sites between Canmore and Cluny (Table 3). Median [reactive silica] was much lower at the two sites furthest downstream (Bow City and Ronalane), often below 0.5 $mg \cdot L^{-1}$, but no trends in silica were detected at these sites. Significant declining trends in water temperature were detected only at two sites (Cluny and Bow City) after 1983. No trends in turbidity were detected at any sites.

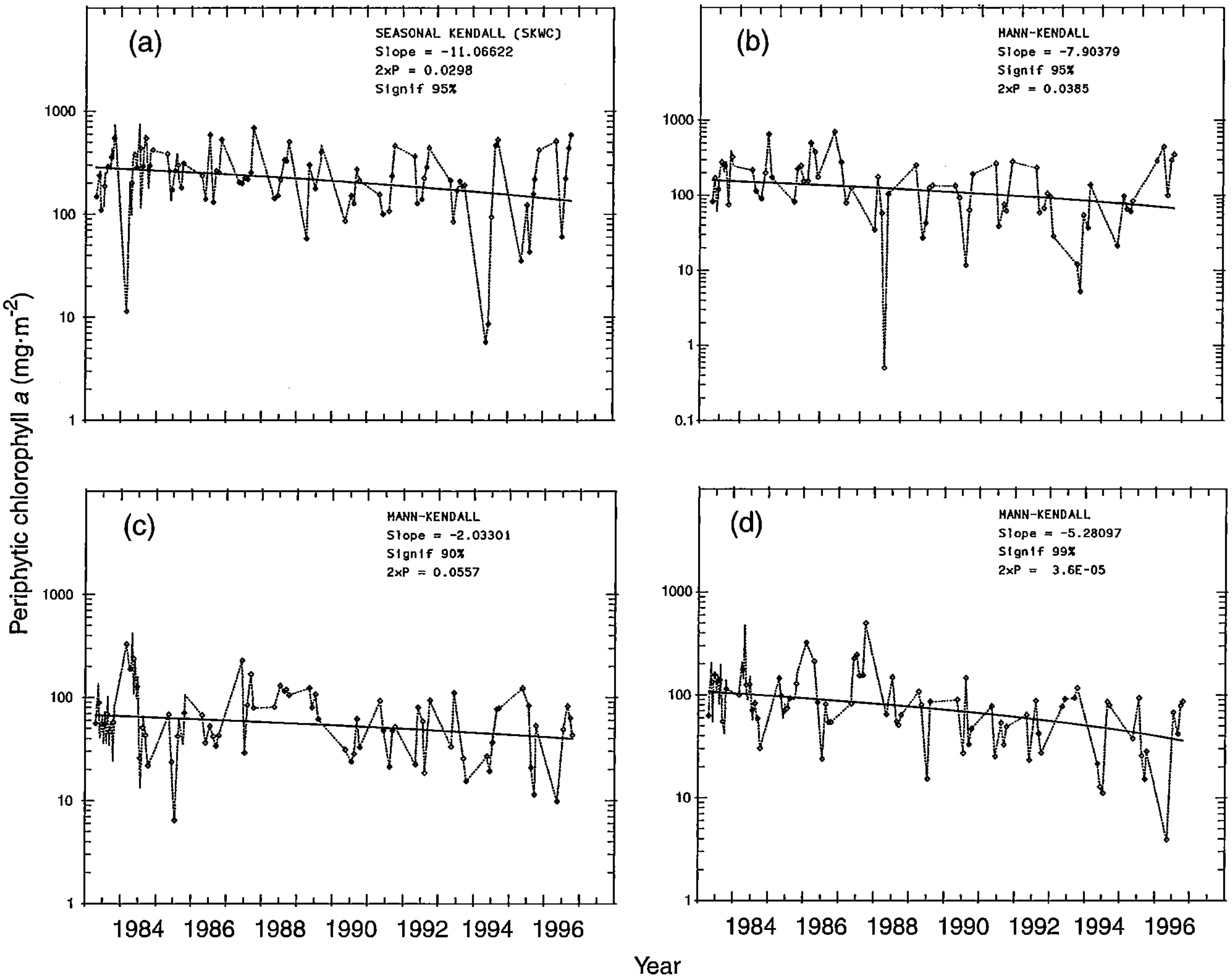
### Periphytic Chl *a* and algal species composition

No significant step trend in periphytic Chl *a* was detected at any site during the 3 years (1983–1985) following the installation of enhanced phosphorus removal (Table 2). Furthermore, there was no change in Chl *a* 3 years before and after the change in sampling method in 1984. Accordingly, there was no indication that this change in sampling method biased results.

There was no monotonic trend in periphytic Chl *a* detected during 1983–1996 upstream from the WWTP discharge at Bowness or downstream at Stier's Ranch. However, this variable gradually declined (monotonic trend) during 1983–1996 at all four downstream sites (Carseland, Cluny, Bow City, and Ronalane) (Table 3; Fig. 7).

Algal species were only identified during 3 years (Fig. 8), but the available data suggest that the species composition of the periphyton has changed since enhanced nutrient removal was installed. The abundance (cells per unit area) of cyanobacteria and diatoms declined significantly since 1980–1981 at two of three sites with declining trends in periphytic Chl *a* (Bow City and Ronalane). Furthermore, the abundance of chlorophyta declined significantly at Stier's Ranch. Although *Cladophora* or *Stigeoclonium* were dominant chlorophyta at all sites from Stier's Ranch to Ronalane in 1980 and 1981

**Fig. 7.** Trend plots of periphytic Chl *a* during 1983–1996 at selected sites: (*a*) Carseland, (*b*) Cluny, (*c*) Bow City, and (*d*) Ronalane. Sen slope represented by a horizontal line.




(Charlton et al. 1986), these taxa were not found at any site in 1994. Periphytic algal species composition and biomass estimates in 1994 are given in Table D2.[1]

Biomass estimates ($\mu g \cdot cm^{-2}$) were only available for May–October 1994 ($n$ = 3 per site) but indicate that biomass was then primarily diatoms at these sites, although cyanobacteria and green algae were also abundant at some sites.[1] Cyanobacteria (primarily *Phormidium*) were especially numerous (cells per unit area) at Carseland in 1994 (Fig. 8), but cyanobacterial biomass was at most 23% of the periphyton, and median Chl *a* was lower at this site in 1994 than in 1981 and 1982 (Fig. 7*a*).

### Macrophyte biomass

Macrophyte biomass declined significantly after 1983 at all of the sites along the right bank (facing downstream) and all but one of the monitoring sites on the other bank of the river (monotonic trends, Fig. 9). Biomass was highest along the right bank in the plumes from the two WWTPs, which both discharge along this side of the river. Furthermore, significant declines in biomass were still detected at nearly half of these sites ($P < 0.10$, $n = 9$) after the data were flow-adjusted to eliminate any influence of an increasing discharge trend.

Macrophyte biomass declined significantly at some sites ($n = 8$) over 7 years (1983–1989) after enhanced phosphorus removal was installed at the WWTPs. However, the greatest decrease in biomass ($n = 20$) (Table 2), especially at sampling sites M10–M12 (Fig. 1), occurred at all sites during declining nitrogen but stable phosphorus levels (1990–1996) (Table 3). Macrophyte biomass was negligible at the upstream control site (site 1) during periodic visits from 1981 to 1989 (Fig. 9). However, biomass and nitrite + nitrate increased at this site after 1989, in spite of an increase in discharge (Table 2), which presumably increased scouring.

The species composition of the macrophyte community was not routinely recorded throughout the entire study. However, periodic observations indicated that *P. vaginatus* remained dominant after 1983 at sites 1–8, and either *P. vaginatus* or *P. pectinatus* was dominant at sites 9–12, as in 1981 and 1982 (Charlton et al. 1986) before enhanced nutrient removal was installed. *Zannichellia palustris* and

**Fig. 8.** Cell counts for periphyton at five sites on the Bow River, 1980, 1981, and 1994: Bowness (1) (different sampling location at Bowness in 1994 than in 1980 and 1981), Stier's Ranch (2), Carseland (3), Bow City (4), and Ronalane (5).

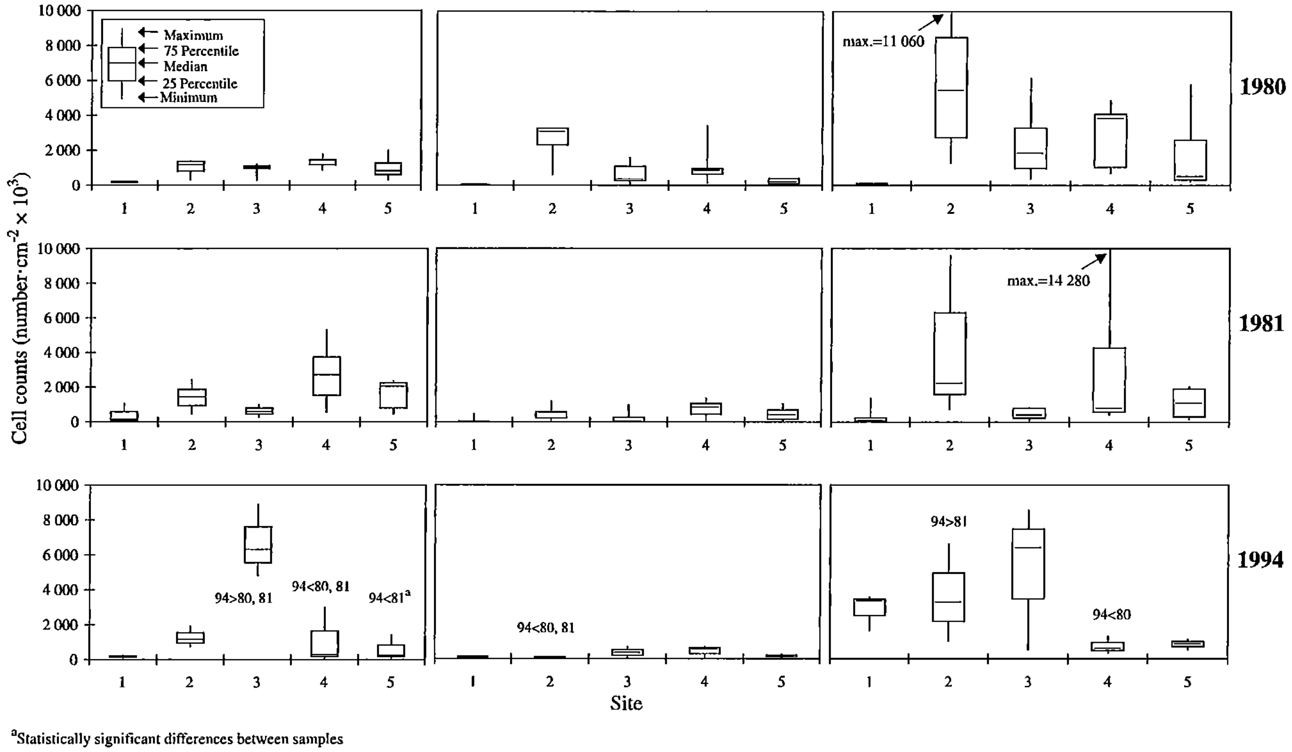



**Table 4.** Correlation between aquatic plant biomass and physical and chemical variables in the Bow River.

| | Spearman rank correlation (rho)[a,b] between plant biomass and other variables[c] | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| Variable | Season discharge | Turbidity | Water temperature | TP | TDP | TN | Ammonia | Nitrite + nitrate |
| Maximum periphytic Chl *a* | −0.009 | **−0.337** | −0.112 | **0.414** | **0.636** | **0.717** | **0.544** | **0.662** |
| Fall macrophyte biomass | **−0.577** | 0.046 | **0.195** | **0.398** | **0.294** | **0.393** | **0.174** | **0.251** |

[a]Bold values are statistically significant ($P < 0.10$, two-tailed test).
[b]Partial Spearman rank correlation used to eliminate spurious correlation where the tested variable, biomass, and a third variable were all significantly correlated.
[c]May–October medians.

*Potamogeton crispus* also occurred infrequently in sheltered areas.

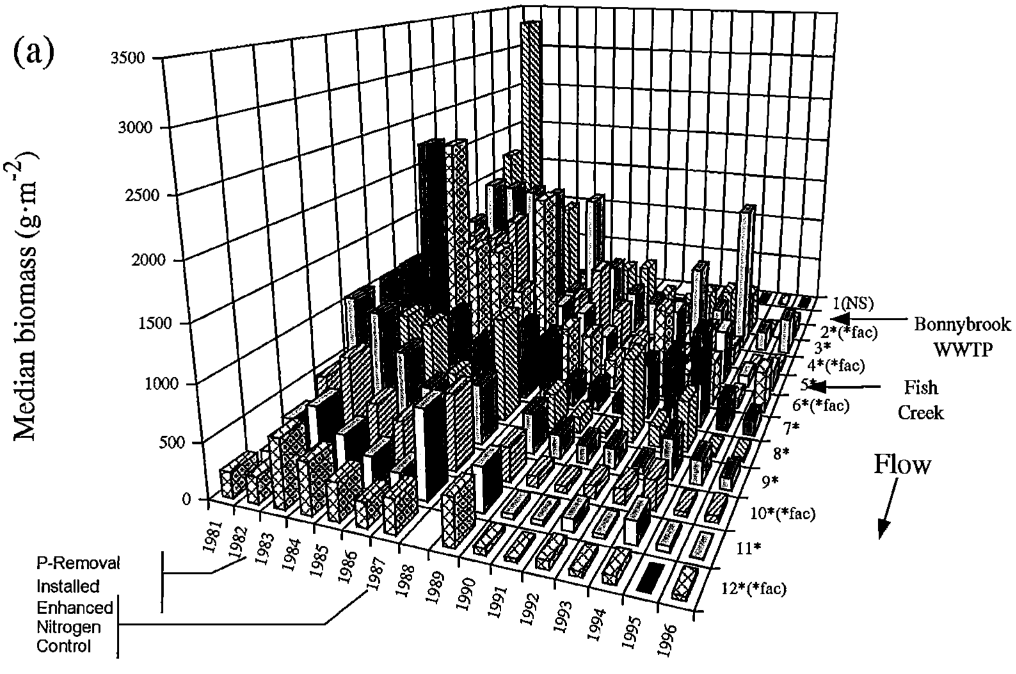
### Correlation and predictive equations for biomass

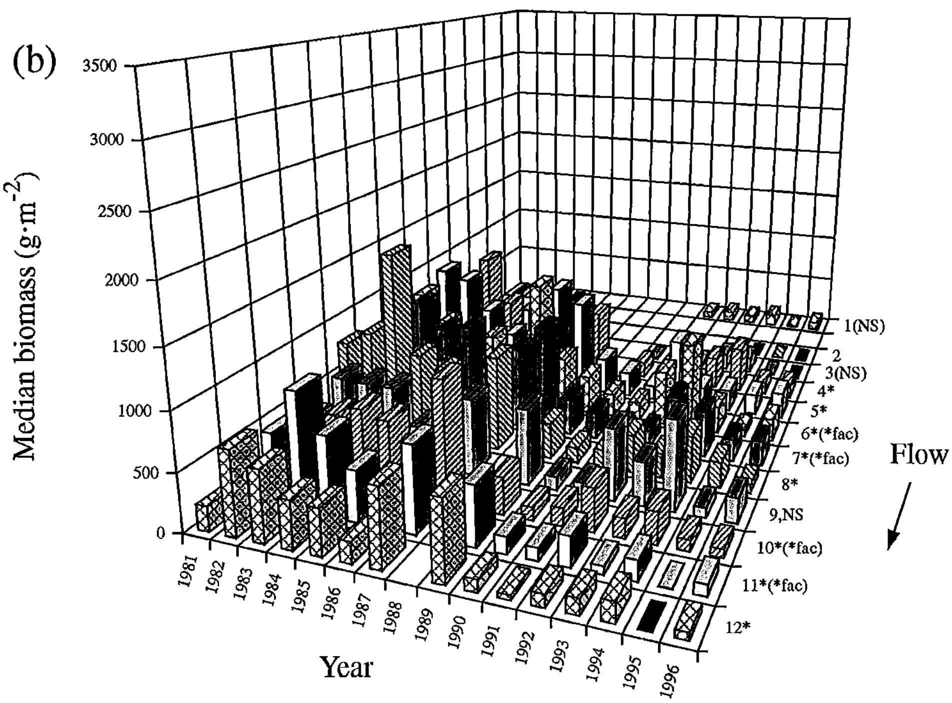
Maximum periphytic Chl *a* was strongly correlated with total nitrogen (TN), TDP, and nitrite + nitrate and correlated to a lesser extent with ammonia and TP (Table 4) but was not significantly correlated with temperature or discharge ($P > 0.699$) evaluated using three different methods ($\rho \leq -0.044$). There was a negative correlation between periphytic Chl *a* and turbidity.

The following multiple regression equation indicates that median TDP ($P = 0.001$) with nitrite + nitrate ($P < 0.001$) explained 55% of the variability in maximum periphytic Chl *a*: maximum periphytic Chl *a* $((\mathrm{mg \cdot m^{-2}})^{0.25}) = 4.590 + 1.397$ nitrite + nitrate ($\mathrm{mg \cdot L^{-1}}$) + 0.274 TDP (ln $\mathrm{mg \cdot L^{-1}}$); $r^2 = 0.55$, $n = 90$. Independent variables in the periphyton and macrophyte equations have been arranged in declining order of statistical significance, as indicated by $t$ values. Other independent variables were not statistically significant.

To solve the multiple regression equation for TDP at nuisance periphytic biomass, [nitrite + nitrate] was first set at 0.211 $\mathrm{mg \cdot L^{-1}}$. This was the [nitrite + nitrate] at a maximum periphytic Chl *a* of 150 $\mathrm{mg \cdot m^{-2}}$, as predicted by a regression equation (maximum periphytic Chl *a* $((\mathrm{mg \cdot m^{-2}})^{0.25}) = 2.979 + 2.600$ nitrite + nitrate − 0.6199 nitrite + nitrate$^2$ ($\mathrm{mg \cdot L^{-1}}$); $r^2 = 0.56$, $n = 90$). Similarly, [TDP] of 4.8 $\mu\mathrm{g \cdot L^{-1}}$ was used to solve the multiple regression equation for nitrite + nitrate (where maximum periphytic Chl *a* $((\mathrm{mg \cdot m^{-2}})^{0.25}) = 5.389 + 0.1084\mathrm{TDP} - 0.0459\mathrm{TDP}^2$ (ln $\mathrm{mg \cdot L^{-1}}$); $r^2 = 0.45$, $n = 90$). The multiple regression equation was then used to predict that nuisance levels of maximum periphytic Chl *a* will occur at a median [TDP] of 6.4 $\mu\mathrm{g \cdot L^{-1}}$ (CI: 1.9–7.6 $\mu\mathrm{g \cdot L^{-1}}$) and a median [nitrite + nitrate] of 0.267 $\mathrm{mg \cdot L^{-1}}$ (CI: 0.090–

**Fig. 9.** Macrophyte biomass in the Bow River, 1981–1996: (*a*) right bank and (*b*) left bank, both facing downstream. *Significant declining trend; NS, not significant; fac, flow-adjusted concentration; black bars indicate zero values.




0.302 mg·L$^{-1}$). This median [TDP] is equivalent to a median [TP] of 18 μg·L$^{-1}$ (where TDP (ln mg·L$^{-1}$) = −0.3268 + 1.169TP (ln mg·L$^{-1}$); $r^2$ = 0.86, $n$ = 90).

With any discharge correlation eliminated, macrophyte biomass was most strongly correlated with TP and TN but was also significantly correlated with TDP, nitrite + nitrate, ammonia, and water temperature (Table 4). Macrophyte biomass was negatively correlated with discharge.

The following equation indicates that discharge ($P$ < 0.001), nitrite + nitrate ($P$ < 0.001), TDP ($P$ < 0.001), and water temperature ($P$ = 0.031) explained 57% of the variability in macrophyte biomass: macrophyte biomass (ln + 1 g·m$^{-2}$) =

44.596 – 1.909 discharge (ln $m^3$) + 0.754 nitrite + nitrate (ln $mg \cdot L^{-1}$) + 0.622 TDP (ln $mg \cdot L^{-1}$) + 1.722 temperature (ln °C); $r^2 = 0.57$, $n = 294$.

## Discussion

### Periphyton

After 1983, the lower Bow River was phosphorus limited and the biomass of periphyton (mainly diatoms) decreased where [TDP] declined below 10 $\mu g \cdot L^{-1}$. Although there have been various enrichment studies in stream channels and in situ, declining periphytic biomass has not been previously reported following controls on phosphorus discharge to a river. The regression equation developed for the Bow River predicted that nuisance levels of maximum periphytic Chl *a* of 150 $mg \cdot m^{-2}$ will occur at the relatively low median TDP of 6.4 $\mu g \cdot L^{-1}$ (1.9–7.6 $\mu g \cdot L^{-1}$). This TDP would be equivalent to about 18 $\mu g \cdot L^{-1}$ TP, well below the 30 $\mu g \cdot L^{-1}$ that Dodds et al. (1997) found would prevent the same maximum periphytic Chl *a*, based on an analysis of results from 200 different sites or rivers from North America, Europe, and New Zealand. However, Chételat et al. (1999) found that nutrient-enriched sites on 13 rivers in Ontario and Quebec had higher periphytic Chl *a* and were usually dominated by *Cladophora* at TP over about 20 $\mu g \cdot L^{-1}$.

Periphytic biomass in the Bow River decreased at TDP similar to levels that have limited growth in experimental stream systems. Bothwell (1989) found that there was a linear increase in the biomass of diatom mats where 1–30 µg soluble reactive phosphorus (SRP)$\cdot L^{-1}$ was added to water in stream channels containing 1.3–2.4 $\mu g \cdot L^{-1}$. Similarly, Horner et al. (1990) reported increased periphytic biomass accrual in laboratory streams containing ≤2 $\mu g \cdot L^{-1}$ to which an additional 15 µg SRP$\cdot L^{-1}$ was added and further increases due to improved nutrient uptake at higher water velocity up to 60 $cm \cdot s^{-1}$. Subsequent work with a calibrated model of channel periphyton accrual has shown that biomass declined only after SRP was reduced below 10 $\mu g \cdot L^{-1}$ (Walton et al. 1995).

Although SRP is widely used in eutrophication studies, TDP was measured in the current study because previous work in the Bow River (Cross et al. 1986) and elsewhere (Bradford and Peters 1987) determined that this was the best indicator of biologically available phosphorus, based on algal assays. In the Bow River, SRP was 0.65 of TDP in summer and 0.78 of annual average TDP (Cross et al. 1986).

During the same time period, there was no significant change in periphytic biomass at an upstream control site (Bowness) or at higher median [TDP] (10–33 $\mu g \cdot L^{-1}$) at Stier's Ranch, where no trend in TDP was detected after 1983. TDP at Stier's Ranch was probably still sufficient for a high rate of periphyton growth. Bothwell (1989) calculated that a maximum areal biomass occurred at an additional 28 µg SRP$\cdot L^{-1}$ (about 36 µg TDP$\cdot L^{-1}$ in the Bow River) and found that periphytic biomass was not phosphorus limited at higher [SRP] (>30–50 $\mu g \cdot L^{-1}$).

Although there was little change in TP loading from Calgary's municipal plants after 1983, a large decline in [TDP] in the lower Bow River occurred after 1989. The reasons that TDP declined in the lower Bow River after 1989 are not known. The Town of High River ceased discharge of wastewater to the Bow River basin in September 1989, but this source contributed only 2408 kg in 1990, or just 2.3% of the combined annual loading from Calgary WWTPs.

There was no evidence that variables other than phosphorus caused the observed decline in periphytic biomass. Wellnitz et al. (1996) reported that the addition of nitrate increased the biovolume of periphyton in Rocky Mountain streams, while the addition of phosphorus alone increased periphytic Chl *a*, ash-free dry mass, and cell density. In the Bow River, periphytic Chl *a* was positively correlated with ammonia and nitrite + nitrate. However, in spite of decreased nitrogen loading from Calgary, no monotonic trends in these forms of nitrogen were detected at two sites with declining periphytic biomass (Bow City and Ronalane). Absence of trends in these variables suggests that municipal loading of nitrogen had less impact at these sites, as nitrogen is lost along the river by denitrification and biological uptake upstream. Although reactive silica at Bow City and Ronalane often fell below levels that limit diatom abundance in lakes (0.5 $mg \cdot L^{-1}$; Wetzel 1983), no trends in silica were detected at these sites.

Similarly, none of the physical variables included in this analysis appeared to account for the decline in periphytic biomass. Although season discharge was significantly correlated with periphytic biomass at some sites (Carseland, Bow City, and Ronalane alone), this relationship was not statistically significant overall. Presumably, the characteristics of individual sites, such as slope, bed mobility, and algal species composition, influence the degree to which biomass is affected by discharge. Furthermore, although positive trends in discharge were detected at Bowness and Stier's Ranch, trends in periphytic biomass were not detected at these sites. Chételat et al. (1999) found no relationship between periphyton biomass and water velocity over the range 10–107 $cm \cdot s^{-1}$ in 13 Ontario and Quebec rivers.

Some authors have reported light limitation of periphytic biomass (Kjeldsen 1996) and community structure (Wellnitz et al. 1996) in streams, but Bothwell (1988) found that seasonal variation in solar insolation had no effect on growth rate at lower nutrient levels. Although solar insolation was not measured in the lower Bow River, there have been no changes in riparian vegetation or turbidity that could account for declining periphytic biomass.

There has been no evidence to date of decreased secondary production following the decline in macrophyte and periphytic biomass in the Bow River downstream from Calgary. No year-to-year trends were detected in the zoobenthic community in annual monitoring at Cochrane, Carseland, or Ronalane during 1983–1987 (Anderson 1991). Furthermore, the condition of trout (estimated weight of 500-mm trout) collected just downstream from Calgary during 1988–1992 did not decrease after enhanced nutrient removal was installed (Table D3).[1] Since no change in periphytic biomass has been detected immediately downstream from Calgary, changes in higher trophic levels would not be expected there. However, declining invertebrate and fish production may occur over time further downstream where periphytic biomass has declined.

Changes in sampling methods or frequency do not appear to account for declining trends in periphytic biomass or nutrients. Periphytic Chl *a* did not change significantly from

1983 to 1984, when the sampling method to remove periphyton from rocks was changed, and a single sampling method was used during 1984–1996 when declining trends were detected at sites along the lower Bow River. Furthermore, Charlton (1986) did not find any significant difference between the two sampling methods in periphytic Chl *a* collected from various habitats in the Muskeg River, except in very shallow areas (<15 cm depth). However, the change to a scraping method may have reduced the diversity of algal species sampled on rocks that were not smooth (Aloi 1990) compared with results before 1984. Trends in periphytic biomass and phosphorus were also present when only the May–October data were tested. Accordingly, the analysis was not affected by changes in the duration of sampling at some sites.

**Macrophytes**

Macrophyte biomass in the Bow River greatly declined over 14 years following the installation of enhanced nutrient removal at Calgary WWTPs, especially at sampling sites furthest downstream. The greatest decrease in macrophyte biomass occurred during 1990–1996 following decreases in nitrogen loading from wastewater and declining nitrogen (total ammonia <0.2 $mg \cdot L^{-1}$, nitrite + nitrate <0.8 $mg \cdot L^{-1}$) but stable TDP in river water at Stier's Ranch. These results and the regression analysis indicate that macrophyte growth was limited more by the availability of nitrogen than of phosphorus, but both nutrients influenced growth.

The regression analysis suggests that discharge had a stronger influence than nutrients on macrophyte biomass in the Bow River. Furthermore, discharge was high enough to initiate brief periods of bed movement and presumably scouring in the area of macrophyte growth during 3 years under enhanced nutrient removal (1983–1996). Chambers et al. (1991) found that macrophyte biomass at sites on the Bow River decreased with increasing current velocity within the weed bed over the range of 0.01 to 1 $m \cdot s^{-1}$ and was rare at speeds over 1 $m \cdot s^{-1}$. However, it would be difficult and impractical to manage macrophyte growth in the Bow River through manipulation of discharge without also controlling nutrient loading. Higher discharge could be used to scour macrophytes, but higher discharge could adversely affect other users of the Bow River. Furthermore, significant macrophyte growth occurred prior to the installation of nutrient removal during 1981 (range of macrophyte biomass: 31–1114 $g \cdot m^{-2}$), when median river discharge was very high.

The reason that macrophyte biomass declined following a decrease in nitrogen loading cannot be determined from the available data, but depletion of nitrogen in river sediments could be an important factor. Aquatic macrophytes in lakes and rivers derive more of their phosphorus and nitrogen supply from sediments through their roots than from the water column (reviewed in Chambers et al. 1989), even in coarse river sediments. Chambers and Prepas (1994) found that nitrogen concentrations in porewater and sediments downstream from a WWTP discharge were greatest at sites with the highest open-water concentrations and highest macrophyte biomass. Since river water can strongly influence riverbed concentration, it is postulated that the nitrogen pool in the bed of the Bow River declined along with the concentration in river water. Rapid depletion of nitrogen in the bed of the Bow River may have caused nitrogen limitation in sediments.

Other studies have found that aquatic macrophytes are more apt to be limited by their supply of nitrogen rather than of phosphorus. Macrophyte growth in lakes was not stimulated by the addition of phosphorus in studies reviewed by Barko et al. (1991), but treatment with nitrogen enhanced the biomass of water hyacinth (*Eichhornia crassipes*) in a lake (Carignan et al. 1994). Feijoo et al. (1996) found a positive correlation between biomass of the submersed macrophyte *Egeria densa* and nitrogen in stream water and sediments. Barko et. al (1991) concluded that pools of exchangeable nitrogen in lake sediments are smaller, more rapidly depleted, and more likely to limit production of submersed macrophytes than pools of phosphorus. In contrast, Carr and Chambers (1998) found that the biomass of *P. pectinatus* in the South Saskatchewan River below Saskatoon was related to sediment phosphorus but not nitrogen concentration, and biomass in their artificial streams did not respond to additional nitrogen alone. The fine-textured sediments in their artificial streams (Carr and Chambers 1998) and the South Saskatchewan River may retain an adequate supply of nitrogen more readily than the rocky substrate typical in the Bow River, although in the same basin.

In addition to reduced municipal discharge, nitrogen loading to the Bow River was also reduced when two fertilizer plants ceased discharge in 1987 and 1990. These two plants contributed about 7128 kg unionized ammonia$\cdot$annum$^{-1}$ and 13 976 kg nitrate$\cdot$annum$^{-1}$ in 1988–1989 (Nahulak and Kabat 1990). However, these industrial loads were only 0.7 and 0.8%, respectively, of the annual municipal loading from Calgary.

Few other studies have monitored the response of aquatic macrophytes to a reduction in nutrient loading from municipal wastewater. Terrell and Canfield (1996) found that large reductions in nitrogen and phosphorus loading from municipal wastewater did not control filamentous algae or submersed vegetation in a cove of the productive Crystal River in Florida, and further nutrient control was not recommended. Chambers (1993) concluded that a decline in percent cover of pondweed (including *P. pectinatus* and *P. vaginatus*) in the South Saskatchewan River was primarily a result of high flow rather than the installation of enhanced phosphorus removal at the WWTP at Saskatoon. In contrast, declining trends in macrophyte biomass in the Bow River were still significant at nearly half of the sites after data were adjusted to eliminate the influence of river discharge.

The biomass of aquatic macrophytes (including the species dominant in the Bow River) also declined in the Highwood River downstream from High River 1 year after the discharge of all treated wastewater ceased in 1989 (fig. 7.3-2 in Golder Associates 1995). Before 1989, dissolved oxygen levels declined to 3.6 $mg \cdot L^{-1}$ (lower in the mixing zone) due to nocturnal respiration by aquatic macrophytes in this river, leading to fish kills. Since then, dissolved oxygen has been no lower than 6.3 $mg \cdot L^{-1}$ in hourly monitoring throughout the summers of 1990–1997.

Studies elsewhere have found no evidence for nutrient limitation of macrophyte growth in running water. Kern-Hansen and Dawson (1981) reported no correlation between macrophyte biomass and either water or sediment nutrient concentration in 19 Danish streams. Canfield and Hoyer (1988) found that aquatic macrophytes in relatively productive streams in Florida were limited by shading rather than nutrient supply. Unlike most of these Florida streams, the channel of the Bow River at the macrophyte sampling sites is wide (59 m at Stier's Ranch), with little shading from riparian vegetation at most sites. Accordingly, shading by riparian vegetation is unlikely to have reduced macrophyte biomass at these sites. Furthermore, no trends in turbidity were detected during this study.

**Management implications**

The results of this study suggest that nuisance growth of aquatic plants caused by municipal loading can be managed through the control of nutrient discharge to some rivers. The response of specific aquatic plant communities to reduced nutrient loading will depend on the species composition of these communities, substrate characteristics, and whether nutrients in sediments and water still exceed levels required for optimal growth. In the Bow River, aquatic macrophyte biomass decreased following a reduction in nitrogen loading which resulted in river median total ammonia of 0.11–0.28 $mg \cdot L^{-1}$ and nitrite + nitrate of 0.47–0.81 $mg \cdot L^{-1}$. Periphytic biomass decreased in the Bow River where TDP in river water was reduced below 10 $\mu g \cdot L^{-1}$ but not at higher TDP (10–33 $\mu g \cdot L^{-1}$).

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