Pergamon

PII: S0043-1354(96)00389-2

Wat. Res. Vol. 31, No. 7, pp. 1738–1750, 1997

Printed in Great Britain
0043-1354/97 $17.00 + 0.00

DEVELOPING NUTRIENT TARGETS TO CONTROL BENTHIC CHLOROPHYLL LEVELS IN STREAMS: A CASE STUDY OF THE CLARK FORK RIVER

W. K. DODDS[1]*, V. H. SMITH[2] and B. ZANDER[3]

[1]Division of Biology, Kansas State University, Manhattan, KS 66506, U.S.A., [2]Environmental Studies Program and Department of Systematics and Ecology, University of Kansas, Lawrence, KS 66045, U.S.A., [3]United States Environmental Protection Agency Region 8, Suite 500, 999 18th St. Denver, CO 80202, U.S.A.

(Received April 1996; accepted in revised form November 1996)

Abstract—Approaches for assessing the effects of lowering nutrients on periphyton biomass in streams and rivers are poorly developed in contrast to those for lakes. Here we present two complementary approaches to assess target nutrient concentrations in streams, given desired mean and maximum standing crops of benthic algal chlorophyll. In the first approach, a reference portion or reach of the river that typically exhibits acceptable levels of benthic chlorophyll is identified (i.e. seasonal mean and maximum values do not exceed desirable levels), and the target levels for instream nutrient concentrations are defined by mean nutrient levels in the reference region. In the second approach, regression and graphical analyses of a large stream database are used to identify acceptable levels of instream total N and total P. The first approach supplies site-specific nutrient targets, whereas the second places nutrient control into a broader, more comparative perspective. In order to link these target concentrations to specific nutrient control measures, we describe a spreadsheet model that can be used to translate changes in external loading by point sources into predicted new instream nutrient concentrations. These quantitative methods are applied here to the control of nuisance algal growth in the Clark Fork River, Montana. We suggest that, in general, maintenance of mean instream total N concentrations below 350 $\mu g\ l^{-1}$ and total P below 30 $\mu g\ l^{-1}$ will result in mean benthic algal chlorophyll *a* density below nuisance levels of 100 mg m^{-2} in most streams. © 1997 Elsevier Science Ltd

Key words—eutrophication, periphyton, streams, rivers, nitrogen, phosphorus, benthic algae

INTRODUCTION

Eutrophication of most fresh waters is dependent upon supplies of nitrogen and phosphorus (Vollenweider, 1968). In the case of lakes and reservoirs, a strong quantitative framework has been developed, over the past three decades, that allows the prediction of algal biomass and other water quality parameters from nutrient loading and water column nutrient concentrations (OECD, 1982; Smith, 1982; Canfield, 1983; Reckhow and Chapra, 1983; Ryding and Rast, 1989). These tools are employed with great success in water quality management of lakes worldwide (Sas, 1989; Cooke *et al.*, 1993).

In contrast, the development of a comparable quantitative framework for algae in flowing waters has lagged far behind. Although nuisance algal growth in nutrient-enriched streams and rivers is common in North America and elsewhere (Wong and Clark, 1976; Lembi *et al.*, 1988; Welch *et al.*, 1988), the general quantitative relationships between nutrient supplies and algal biomass in lotic systems are not well characterized. It is clear from laboratory studies (Klewer and Holm, 1980; Horner *et al.*, 1983) and from nutrient enrichment studies in outdoor artificial streams (Stockner and Shortreed, 1976; Bothwell, 1985; Watson *et al.*, 1990), that nutrients stimulate the growth of stream periphyton. However, empirical models for periphyton biomass in natural streams and rivers are rare (Welch *et al.*, 1988; Biggs and Close, 1989; Lohman *et al.*, 1992) and may be specific to localized areas. Consequently, managers find it difficult to make informed decisions regarding the nutrient control of nuisance algal biomass in lotic systems. The purposes of this paper are to present a series of approaches that can be used to help establish nutrient control criteria in rivers and streams where eutrophication has been deemed excessive, and to illustrate the application of these techniques to the Clark Fork River, Montana (Fig. 1).

Benthic chlorophyll *a* routinely in excess of 100 mg m^{-2} is observed upstream from Alberton in the Clark Fork of the Columbia River, Montana (Watson, 1989; Watson *et al.*, 1990; Ingman, 1992a, b). These densities are generally undesirable from a water quality point of view (Horner *et al.*, 1983; Welch *et al.*, 1988). The State of Montana was

*Author to whom correspondence should be addressed. [Fax: +1 913 532 6653].

interested in four eutrophication management questions: (1) what concentrations of dissolved inorganic nitrogen (DIN) and soluble reactive phosphorus (SRP) and ratios of N:P are required to yield chlorophyll *a* densities of 50, 100 and 200 mg m^{-2}?; (2) should total phosphorus and nitrogen be managed instead of DIN and SRP, when assessing potential algal density?; (3) will P removal make a difference in total algal biomass and relative abundance of *Cladophora* (a common nuisance filamentous green alga) in apparently N-limited upstream reaches?; (4) would nearly complete removal at upstream point sources only during the summer irrigation season result in significant reductions in river-wide periphyton levels? We assume that similar questions, typically, will be asked by those attempting to control eutrophication in other flowing waters and suggest that the general approach presented here provides a quantitative framework that has broad applicability.

METHODS

Site description

The Clark Fork River drains approximately 57,000 km^2 of western Montana (Fig. 1) and this paper considers roughly the upper 370 km. There are nine major tributaries and nine towns along this portion of the river. Land use is predominantly forest, rangeland and agriculture in this basin (Ingman, 1992a).

Previous assessments of nutrient limitation on the Clark Fork include artificial stream measurements of the effect of N and P additions (Watson, 1989), and nutrient uptake measurements made using *Cladophora* collected from the river (Lohman and Priscu, 1992). Nutrient deficiency has also been evaluated using cellular N and P concentrations in *Cladophora* (Lohman and Priscu, 1992) and ratios of DIN:SRP (Watson *et al.*, 1990). These studies together indicated that N limitation is common in the Clark Fork system; at times a balance may occur between N and P limitations; and N limitation appears to decrease downstream.

Fig. 1. The Clark Fork River basin, Montana.

Analysis and modeling

Models relating TN, TP, DIN, SRP and other variables to benthic chlorophyll *a* were constructed from a cross-sectional database derived from over 200 distinct sites or rivers throughout North America, Europe and New Zealand (Table 1). Means were determined for 2–3 month periods from all seasons, but summer samples were the most common. Values of chlorophyll *a* in these studies were derived from artificial (about 1/3 of the data) and natural substrates. Stepwise multiple regression was used to construct a series of eutrophication models. Different combinations of variables were forced in different orders (SAS, 1988) and the 95% prediction confidence levels for chlorophyll *a* were calculated for each combination of variables. Models for seasonal mean and maximum chlorophyll *a* are reported here because they are likely to be most relevant to users and to those concerned with controlling stream eutrophication. A graphical probabilistic method based on the procedures of Heiskary and Walker (1988) was also developed. In this method, the percentages of chlorophyll *a* values (mean and maximum) exceeding a series of defined levels (50, 100, 150 and 200 mg m^{-2} chlorophyll *a*) were calculated for a series of specified ranges of nutrient concentrations. Other factors influencing benthic algal biomass (latitutde, temperature, stream gradients, discharge and light) were also investigated using nonparametric correlation analysis, but none of these variables were found to be as useful as a predictor of stream chlorophyll *a* as TN or TP (Dodds *et al.*, unpubl.), so they were not included in our final model development.

We defined nuisance levels of benthic algal chlorophyll *a* as mean values exceeding 100 mg m^{-2} and maximum values exceeding 150 mg m^{-2}. Similar criteria have been used previously (Horner *et al.*, 1983; Welch *et al.*, 1988), and we consider benthic chlorophyll *a* of values less than these critical limits to be indicative of acceptable conditions with regard to benthic algal biomass. Our analysis does not apply to macrophyte growth, a separate water quality parameter that we have not addressed here.

The bulk of the data on nutrient inputs to the Clark Fork were derived from monitoring programs conducted by the State of Montana Department of Health and Environmental Sciences (Ingman, 1992a, b). These data include measurements of DIN, SRP, total N (TN), and total P (TP) both for the river itself and for major point sources and tributaries to the river. Information was also available on benthic chlorophyll *a* levels (Watson, 1989; Watson *et al.*, 1990; G. Ingman, personal communication; V. Watson, personal communication), and the composition and abundance of attached algal species (Bahls, 1989; Weber, 1991; Weber, 1993).

A spreadsheet model was developed to predict the fate and transport of nutrient loadings throughout the study area. This one-dimensional, steady-state, spreadsheet model can be used to calculate the effects of changes in external loadings on instream nutrient concentrations. It uses a mass-balance approach to calculate the spatial distributions of TN and TP over a section of the river, as well as to predict changes in water chemistry that result from external load reductions at different locations. The model portrays conditions during June, July and August for a broad geographic area (Fig. 1) from Warm Springs down to Superior (approximately 370 river km). The results of the model are reported here as a longitudinal profile of instream concentrations and loadings of TN.

The spreadsheet model consists of 28 river segments ranging from 0.5 to 80 km in length. The length of each segment is defined by the locations of known inputs (i.e. point sources and major tributaries) and the locations of existing sampling stations. The TN concentrations in each

Table 1. Location, number of sites, source and type of data collection sites. N = natural substrate; A = artificial substrate

Location	No.	Type	Source
Washington 11, Montana 4, Sweden 6	21 rivers	N	Welch et al., 1988
Ozark, Missouri, 2 yr	21 streams	N	Lohman et al., 1992, Lohman, 1988
Pennsylvania, Michigan, Idaho. Oregon, (four streams each)	16 streams	N	Bott et al., 1985
New Zealand	16 streams	N	Biggs 1995
Missouri urban, forest and pasture catchements, 2 yr	13 streams	N	Smart et al., 1985; Smart, 1980; Jones et al., 1984
New Zealand	9 streams	N	Biggs and Close, 1989
New Zealand	9 streams	A	Biggs, 1988
North America (Southeastern U.S. to Arctic)	8 streams	N	Meyer et al., 1993
Spokane River, Washington, 2 yr, both substrate types	8 sites	N, A	Funk et al., 1983; Nielsen et al., 1984
New Zealand, 2–4 sites each stream	7 streams	N	Welch et al., 1992
North Carolina	7 streams	N	Smith (unpublished)
Virginia and New Hampshire	6 streams	N	Hornberger et al., 1977
Idaho	6 sites	A	Delong and Brusven, 1992
Quebec	5 streams	N	Naiman, 1980; Naiman, 1981; Naiman, 1983
Artificial streams	5 treatments	A	Horner et al., 1990
Montana	4 sites	A	Marcus, 1980
Kansas artificial streams	4 treatments	A	Tate, 1990
Oregon Cascades	3 streams	N	Lyford and Gregory, 1975
British Columbia	3 streams	N	Stockner and Shortreed, 1976
Upper Mississippi River navigation pools, four sites each, 2 yr	2 pools	A	Luttenton, 1982; VanSteenberg, 1983
Fertilized and unfertilized Arctic stream	1 stream	N	Peterson et al., 1985
Tennessee, 2 yr	1 stream	N	Rosemond, 1994
New Jersey, six sites	1 stream	A	Flemer, 1970
Michigan, four sites	1 stream	A	Lenon et al., 1979
New Zealand grazed and ungrazed	1 stream	A	Biggs and Lowe, 1994
Danish lowland	1 stream	N	Sand-Jensen et al., 1988
British Columbia	1 stream	A	Perrin et al., 1987

segment were calculated from all known nutrient inputs (point sources, plus direct inflows from the segment above) and nutrient losses (loss downstream, dilution by tributaries). An additional correction factor was calculated for each segment to estimate changes in nutrient levels that could not be measured directly (e.g. nonpoint sources, gain or loss associated with groundwater exchange, biotic uptake or release, sedimentation and burial). The correction factor was calculated as the proportion of nutrients gained or lost in the segment that could not be accounted for by known inputs and losses. Thus, for cases in which the water in a segment exhibited a 50% decrease in TN concentration during its transit through the segment that could not be accounted for by known inputs and exports, we assumed that 50% of TN entering the segment similarly would be lost under all future management scenarios.

A summer period of 21 June–21 September was chosen for our spreadsheet modeling because: (1) it is the time period in which past data suggest that benthic algal biomass problems are most likely to occur; (2) the use of annual mean values would not adequately reflect the period when critical low flows may occur; and (3) this is the time period used in previous stream water modeling efforts for the State of Montana. The model was calibrated using longterm (1988–1992), summer mean TN concentrations available from the State of Montana for both normal summer flows and for critical low flows (7Q10 hydrologic regimes). The 7Q10 flow regime is minimum discharge summed over a 7 day period having an expected return time of 10 yr.

RESULTS

Dissolved inorganic nutrient criteria

The State of Montana wished to know what concentrations of streamwater, DIN and SRP and ratios of N:P were necessary to yield chlorophyll *a* densities of 50, 100 and 200 mg m^{-2}. General relationships between benthic chlorophyll and dissolved inorganic nutrients are characterized by extremely high variance. Seasonal mean values for SRP or DIN explain a very low proportion of the variance in observed chlorophyll *a* (Table 2 and Fig. 2A, B) from our entire database although values from the Clark Fork alone show a closer relationship. Thus, these data suggest that practical regulations for general external nutrient loading for stream eutrophication control should not be based upon instream SRP and DIN levels, because the prediction uncertainty inherent in such an approach may preclude the satisfactory management of benthic chlorophyll *a*. To illustrate the weakness of modeling approaches based on dissolved nutrients, we used empirical models for DIN and SRP (equations (10) and (20), Table 2) to estimate three desired levels of control (50, 100 and 200 mg m^{-2}). Note that the 95% confidence levels for the predicted seasonal mean chlorophyll *a* densities span 1–2 orders of magnitude (Table 3).

The relationships between the summer mean chlorophyll *a* values found in the Clark Fork in 1990 (V. Watson, personal communication) and their associated mean DIN and SRP values (G. Ingman, personal communication) are plotted with values compiled from other sites in Fig. 2. Using data from the Clark Fork alone to extrapolate to lower values of DIN and SRP would be difficult because chlorophyll *a* measurements were taken from artificial substrates and a limited range of DIN and SRP concentrations. These data do suggest that lower inorganic nutrients may lead to lower chlorophyll *a* densities in the Clark Fork.

Total N and P criteria

Another question posed by Montana was: should instream total P and N concentrations be managed instead of DIN and SRP when assessing potential algal biomass? Our analyses revealed that both total N and total P are related more strongly with benthic algal biomass than are dissolved inorganic N or P

(Table 2). We used three complementary approaches to predict the instream TN and TP concentrations that should correspond to desired chlorophyll *a* densities in the Clark Fork: (1) the regression models (Table 2) were used to calculate acceptable target concentrations of TN and TP and their associated confidence intervals; (2) a graphical probabilistic approach was used to predict target TN and TP levels; and (3) measurements of water quality from six reference sites consistently exhibiting acceptable levels of benthic algal biomass were averaged to develop independent estimates of target instream TN and TP concentrations.

Regression model results. Chlorophyll *a* densities and TN and TP concentrations in the Clark Fork River lie well within the range of reported values throughout the world (Fig. 2C, D). Regression analyses of the full data set revealed that the greatest amount of variance in benthic chlorophyll *a* was accounted for by instream concentration of TN, followed by TP (Table 2).

We subsequently determined target instream TN and TP concentrations corresponding to the three critical mean and maximum chlorophyll *a* values (50, 100 and 200 mg m^{-2}) with a three step process. (1) The appropriate TN and TP values were calculated using the Redfield ratio of 7.23 g N:1 g P, by mass (Ryding and Rast, 1989); these values were assumed to result in balanced algal growth. Setting either TN or TP in these combined models is necessary, because otherwise an enormous number of solutions can satisfy the equations. (2) Equations (7) and (17) (Table 2, mean and maximum chlorophyll *a*, respectively) were used, and TN was varied by iteration to find the values of TN and associated TP that would result in the desired target chlorophyll levels. The associated 95% confidence interval was calculated for each of the TN values corresponding with the targets. (3) Equations (4) and (14) (Table 2, mean and maximum chlorophyll *a*, respectively) were used with TN set according to the iterated TP value using the Redfield ratio, and the TP values that correspond to the values of 50, 100 and 200 mg chlorophyll *a* m^{-2} were found by iteration.

Considerable statistical variance exists in the TN-TP-chlorophyll *a* models that we developed (Tables 4 and 5). However, these confidence limits are on average 40% less than those for the DIN and SRP chlorophyll *a* predictions (Table 3). If instream mean TN and TP concentrations in the Clark Fork are reduced by external nutrient loading controls such that a target mean chlorophyll *a* of 50 mg m^{-2} is attained, then the 95% confidence intervals suggest that the resulting mean chlorophyll *a* generally would not exceed 170 mg m^{-2}. Maximum chlorophyll *a* levels could still, periodically, exceed 150 mg m^{-2}. If the seasonal mean TN concentration is reduced to 275 $\mu g\, l^{-1}$, equations (7) and (17) predict that maximum chlorophyll *a* will be 100 mg m^{-2} (Table 4). Similar results were obtained using equations (4) and (14); with TP up to 35 $\mu g\, l^{-1}$, and TN at 252 $\mu g\, l^{-1}$, the maximum chlorophyll *a* would be about 100 mg m^{-2} (Table 5).

Probabilistic modeling approaches and results. This probabilistic analysis was based on methods developed by Heiskany and Walker (1988) to assess the risk of exceeding user-specified chlorophyll *a* levels. We calculated the frequency with which the three critical chlorophyll *a* levels posed in question 1 (50, 100 and 200 mg m^{-2}) were exceeded in our water quality database across defined ranges of instream mean TN concentrations (Fig. 3A, B). This analysis revealed that neither the seasonal mean nor the maximum densities of benthic chlorophyll *a* exceed

Table 2. Regressions with mean and maximum chlorophyll *a* as the dependent variables. Units are: mean chlorophyll *a* and maximum chlorophyll *a* in mg m^{-2}; TP, TN, DIN and SRP in $\mu g\, l^{-1}$; TN:TP ratio by mass. The value for the coefficient of regression for each independent variable is given in parentheses. Statistical significance for each variable is indicated by the superscripts.* $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$. This analysis is based upon values compiled for 205 streams or sites throughout North America and New Zealand

Dependent variable	Independent variables				Adjusted r^2
log (mean Chl *a*)	intercept	1st	2nd	3rd	
(1)	1.091***	logTP (0.2786***)	—	—	0.089
(2)	0.7950*	logTP (0.6021)	$(\log TP)^2$ (−0.08113)	—	0.086
(3)	−0.1238	logTP (1.3418***)	$(\log TP)^2$ (−0.19749***)	TN:TP (0.00727*)	0.412
(4)	−0.4285	logTP (0.92178***)	$(\log TP)^2$ (−0.16468**)	logTN (0.37408**)	0.429
(5)	0.01173	logTN (0.59490***)	—	—	0.348
(6)	−2.36566*	logTN (2.28959**)	$(\log TN)^2$ (−0.29390*)	—	0.371
(7)	−3.22360**	logTN (2.82630**)	$(\log TN)^2$ (−0.431247**)	logTP (0.25464**)	0.430
(8)	1.02412***	logSRP (0.2327**)	—	—	0.053
(9)	0.83531***	logDIN (0.29576***)	—	—	0.140
(10)	0.37366	logDIN (0.42093**)	logSRP (−0.03812)	—	0.166
log (max Chl *a*)					
(11)	1.4995***	logTP (0.28651***)	—	—	0.071
(12)	1.25846**	logTP (0.55015)	$(\log TP)^2$ (−0.06611)	—	0.067
(13)	0.21620	logTP (1.47096***)	$(\log TP)^2$ (−0.22211***)	TN:TP (0.007238*)	0.368
(14)	0.00652	logTP (1.10067***)	$(\log TP)^2$ (−0.19286**)	logTN (0.3129*)	0.370
(15)	0.47022	logTN (0.60252***)	—	—	0.284
(16)	−1.80885	logTN (2.22712*)	$(\log TN)^2$ (−0.28175)	—	0.299
(17)	−2.70217*	logTN (2.78572**)	$(\log TN)^2$ (−0.43340**)	logTP (0.30568**)	0.354
(18)	1.71065***	logSRP (0.09080)	—	—	0
(19)	1.47894***	logDIN (0.22488***)	—	—	0.068
(20)	1.06686***	logDIN (0.41661**)	logSRP (−0.177211)	—	0

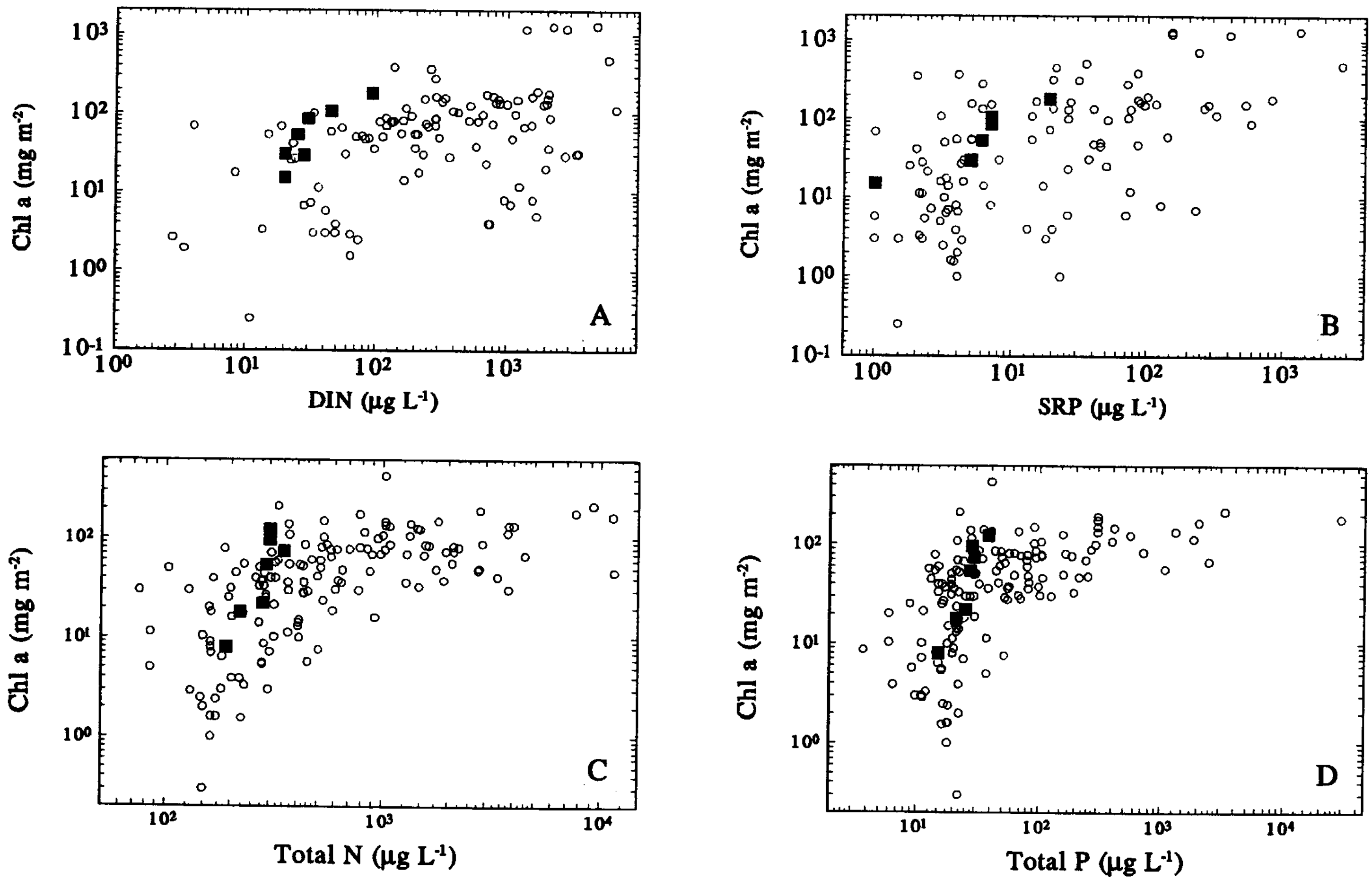

Fig. 2. Mean summer (June–September) chlorophyll *a* for the Clark Fork River and seasonal mean data from other sites throughout the world as a function of DIN (A); SRP (B); TN (C); and TP (D). The filled squares are data points from the Clark Fork (courtesy of V. Watson and the Montana Department of Health and Environmental Sciences).

Table 3. Estimates of SRP and DIN levels required to yield mean and maximum chlorophyll a levels of 50, 100 and 200 mg m^{-2}. Calculated from relationships presented on Table 1. — = not possible to calculate from our model

Target Chl a (mg m^{-2})	SRP (mg l^{-1})	Lower 95% confidence (mg m^{-2})	Upper 95% confidence (mg m^{-2})	DIN (mg l^{-1})	Lower 95% confidence (mg m^{-2})	Upper 95% confidence (mg m^{-2})
Mean 50	0.79	2.44	889	0.83	4.75	505
100	15.6	3.2	1383	8.67	7.92	883
200	307	—	—	90.3	11.1	1292
Max. 50	0.0008	1.98	1333	0.01	3.79	693
100	1.54	3.51	3152	0.21	7.55	1338
200	—	—	—	4.53	14.7	2776

100 mg m^{-2} in the majority of cases, when TN was kept below 200 $\mu g\ l^{-1}$ (Fig. 3A, B). Similarly, when mean TN concentrations remained at or below 500 mg m^{-2}, mean benthic chlorophyll a densities exceeded 150 mg m^{-2} in only 5% of the cases. A similar approach using TP plots suggested mean chlorophyll of 100 mg m^{-2} were present in over half the cases where TP is $< 50\ \mu g\ l^{-1}$ (Fig. 3C). Maximum chlorophyll exceeded 150 mg m^{-2} at least half the time when TP exceeded 50 $\mu g\ l^{-1}$ (Fig. 3D). These plots also further confirmed a clearer relationship between TN and chlorophyll a than between TP and chlorophyll a.

Reference station approach. In this analysis we used a reference station approach, analogous to that used by Ingman (1992b), to develop target instream DIN and SRP levels for the Clark Fork River. In this approach, six sampling stations (stations 8.5, 9, 13, 15.5, 24 and 25 from Ingman, (1992a, b)) were selected that exhibit subjectively determined acceptable levels of benthic chlorophyll a. Target nutrient concentrations then were calculated as the 1988–1992 means of summer (21 June–21 September) TN and TP. The mean summer TN and TP concentrations averaged 318 and 20.5 $\mu g\ l^{-1}$, respectively, during this period. These mean concentrations from the reference reaches are consistent with the target values obtained from both the regression and the probabilistic analyses.

Nutrient removal and Cladophora *dominance*

The State of Montana wished to know if P removal would make a difference in total algal biomass and relative *Cladophora* abundance in apparently N-limited upstream reaches. As is true of many over-enriched streams, nuisance levels of *Cladophora* biomass constitute a water quality variable of concern in the Clark Fork River. We, thus, sought to find the nutrient conditions that correlated most strongly with *Cladophora* growth. No consistent relationship could be found between *Cladophora* abundance and instream TN:TP ratios, but the lowest *Cladophora* abundance was observed primarily at extremely high TN and TP concentrations (Fig. 4B, C). Therefore, it is difficult to predict on the basis of our current database whether a policy of P removal will alter the relative abundance of *Cladophora* in upstream reaches of the Clark Fork. However, a loading management policy favoring high instream TN and TP concentrations designed to reduce *Cladophora* abundance would lead, in turn, to excessive total levels of chlorophyll a and would not be a viable management option.

Reach specific management options

The final issue of concern to the State of Montana was that if nearly complete removal at upstream point sources only occurred during the summer irrigation season, it would result in significant reductions in river-wide periphyton levels. Our one-dimensional, steady-state, spreadsheet model was used to create predictions for: (1) baseline TN concentrations with no loading controls imposed; (2) a scenario in which a 100% removal of the Deer Lodge inputs was imposed; and (3) control at Butte and 100% removal of Deer Lodge inputs. The second two control scenarios were modeled because they were identified previously as the most viable management approaches by the State of Montana. Both sets of predictions were run first for typical summertime flow conditions and then for conditions of critical low flows over a 7 day period expected to occur every 10 yr on average (the 7Q10 conditions). Here, we present only the typical summertime flow results.

Under normal summer baseline flow conditions, the 350 $\mu g\ l^{-1}$ TN target is exceeded at many stations under normal summer flow conditions (Fig. 5A). A 100% removal of nutrient inputs from Deer Lodge alone results in compliance with a 350 $\mu g\ l^{-1}$ TN target at all stations downstream (Fig. 5B). A 100% elimination of the Deer Lodge inputs coupled with a

Table 4. Levels of instream TN and TP required to reach target mean and maximum chlorophyll a. Levels calculated from equations (7) and (17) (Table 2) with TP set to Redfield ratio

Target Chl a (mg m^{-2})	TN ($\mu g\ l^{-1}$)	TP ($\mu g\ l^{-1}$)	Lower 95% (mg m^{-2})	Upper 95% (mg m^{-2})
Mean 50	450	62.3	6.56	170
100	1600	221	26	420
200	3000	415	15.5	436
Max. 50	145	20.1	7.8	407
100	275	38.1	29	1380
200	650	90	63	3310

Table 5. Levels of instream TN and TP required to reach target mean and maximum chlorophyll *a*. Levels are calculated from equations (4) and (14) (Table 2) with TN set to Redfield ratio

Target Chl *a* (mg m^{-2})	TN (μg l^{-1})	TP (μg l^{-1})	Lower 95% (mg m^{-2})	Upper 95% (mg m^{-2})
Mean 50	470	65	9	234
100	1423	197	11	295
200	7570	1020	33	1072
Max. 50	115	16	7	352
100	252	35	15	710
200	650	90	28	1356

7.0 mg l^{-1} effluent TN limit at the Butte WWTP would extend compliance with a 350 μg l^{-1} TN target to all stations upstream of Deer Lodge and downstream of the mouth of Warm Springs Creek (Fig. 5C). Results from critical low flows show that more stringent loading controls are required to reach the TN target (data not shown). Similar results were also obtained for TP reduction simulations. Less stringent controls are likely to lead to a lack of compliance at one or more downstream stations.

DISCUSSION

Utility of models using dissolved inorganic nutrients and TN and TP

The poor correspondence between levels of algal biomass and DIN and SRP seen in Fig. 2(A) and (B) is a direct consequence of the biotic processes that control upstream concentrations of DIN and SRP. At any given time, the concentrations of dissolved inorganic nutrients are determined by the balance between uptake and regeneration (remineralization), which act simultaneously to control dissolved nurients at approximately equilibrium levels and are very resistant to perturbations (Dodds, 1993). Dissolved inorganic nutrients are also poorly correlated with the biomass or activity of algae in lakes (Brylinsky and Mann, 1975). Lake managers use TP rather than SRP to make recommendations for target chlorophyll *a* levels (e.g. OECD 1982; Reckhow and Chapra, 1983; Ryding and Rast, 1989; Sas, 1989). Similarly, we believe that using SRP in streams probably would not be fruitful in the absence of explicit site-specific models for nutrient uptake and remineralization.

Although some published data on uptake rates of inorganic N and P are available for the Clark Fork, we have been unable to obtain the corresponding direct estimates of N and P remineralization rates. A similar lack of data on these two important parameters is also likely to be the case for most, if not all, of the rivers in the world. Thus, it is extremely difficult to estimate, with confidence, how changes in nutrient loading would influence instream DIN or SRP concentrations, and the high variance (Fig. 2A, B) makes it difficult to relate changes in DIN or SRP to subsequent changes in benthic algal biomass.

The most parsimonious explanation for the stronger relationship observed here between chlorophyll *a* and TN and TP is that instream TN and TP concentrations are more indicative of the nutrients that are ultimately biologically available for benthic algal growth than are DIN and SRP. We tested a variety of other factors in our global data set (including DIN, SRP, latitude, stream slope, mean and maximum flows and temperature), and none were correlated as strongly with benthic chlorophyll *a* as TN and TP (Dodds *et al.* unpublished). Thus, management criteria based on TN and TP probably offer the best option for managers wishing to control nuisance algal growth in streams and rivers. We recognize that water column TN and TP values include N and P in algal cells suspended from the benthos. However, much of the TN and TP is ultimately biologically available. Even if all the TN and TP (Bradford and Peters, 1987) is not available, TN and TP are probably the best predictors of available nutrients.

Other investigators have also used water column nutrients to predict algal biomass in streams. For example, local TP-based models have been presented for Missouri (Lohman *et al.*, 1992) and New Zealand (Biggs and Close, 1989). As far as we know, the general application of these models as a management tool for benthic chlorophyll has not been explored. In addition, suspended chlorophyll has been positively correlated with TP in streams as a function of stream catchment area (Van Nieuwenhuyse and Jones, 1996), but this model cannot be applied to benthic algal biomass.

Possible targets for TN and TP

The results presented above provide three separate, but complementary, estimates of target concentrations for TN and TP in the Clark Fork. Regression methods suggest that 275 μg l^{-1} TN generally will yield acceptable (100 mg m^{-2} mean and 150 mg m^{-2} maximum) chlorophyll *a* levels. The probabilistic approach suggests that when TN is in the range of 200–500 μg l^{-1}, chlorophyll will be acceptable in most cases. The reference station approach demonstrated that, generally, an acceptable site had an average TN concentration of 318 μg l^{-1}. Given the strong similarities among the target values derived from these three different approaches, we adopted a value of 350 μg l^{-1} TN as a provisional target level that would allow for some external input of TN to the Clark Fork system and yet should avoid frequent episodes of excessive benthic algal growth.

A corresponding phosphorus target value of 38 μg l^{-1} TP (Table 4) is suggested, when the Redfield

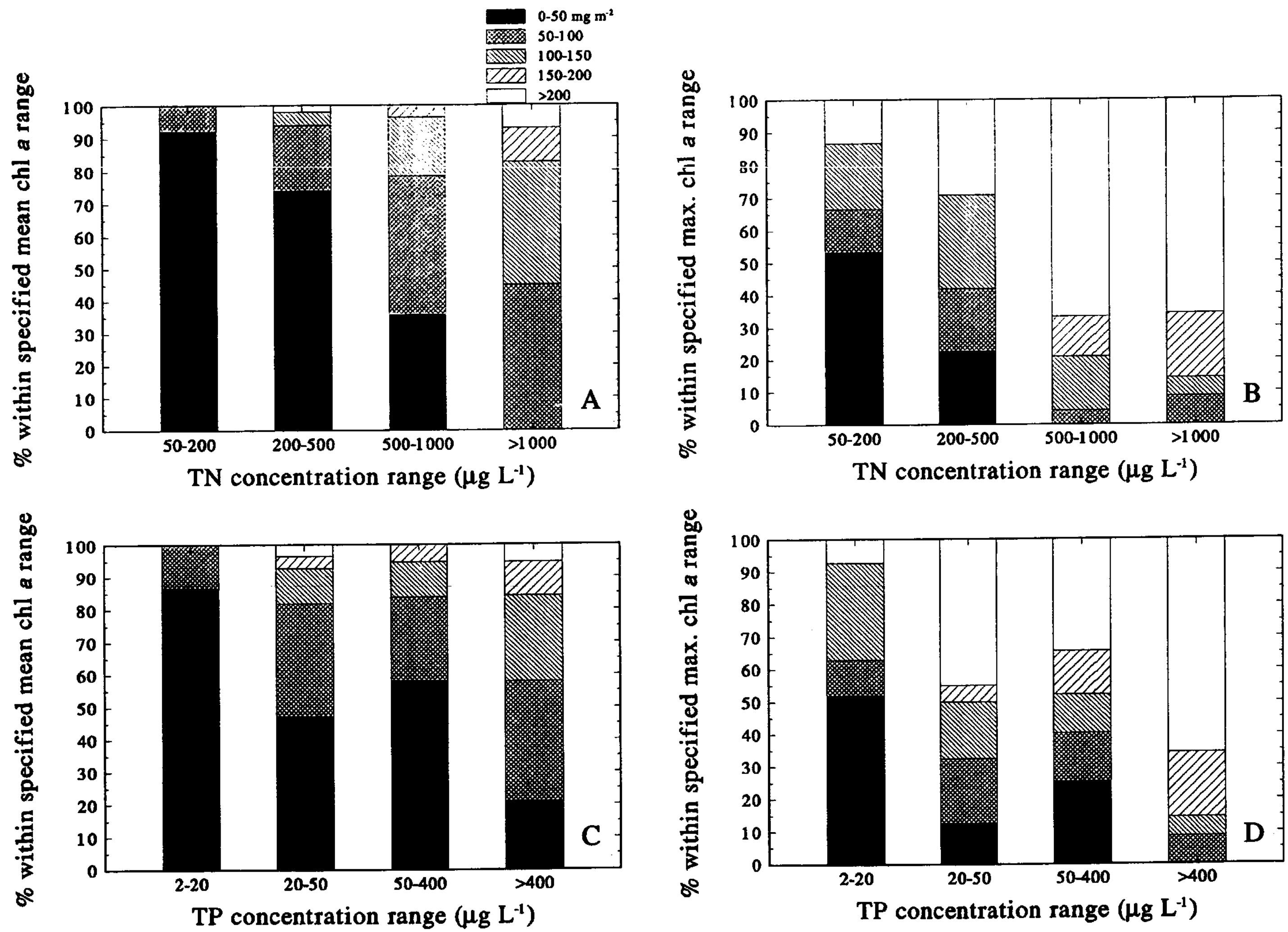

Fig. 3. Percentage of mean or maximum chlorophyll *a* values falling within predefined ranges for each of four separate TN or TP ranges. Mean (A) and maximum (B) as functions of TN, and mean (C) and maximum (D) as functions of TP.

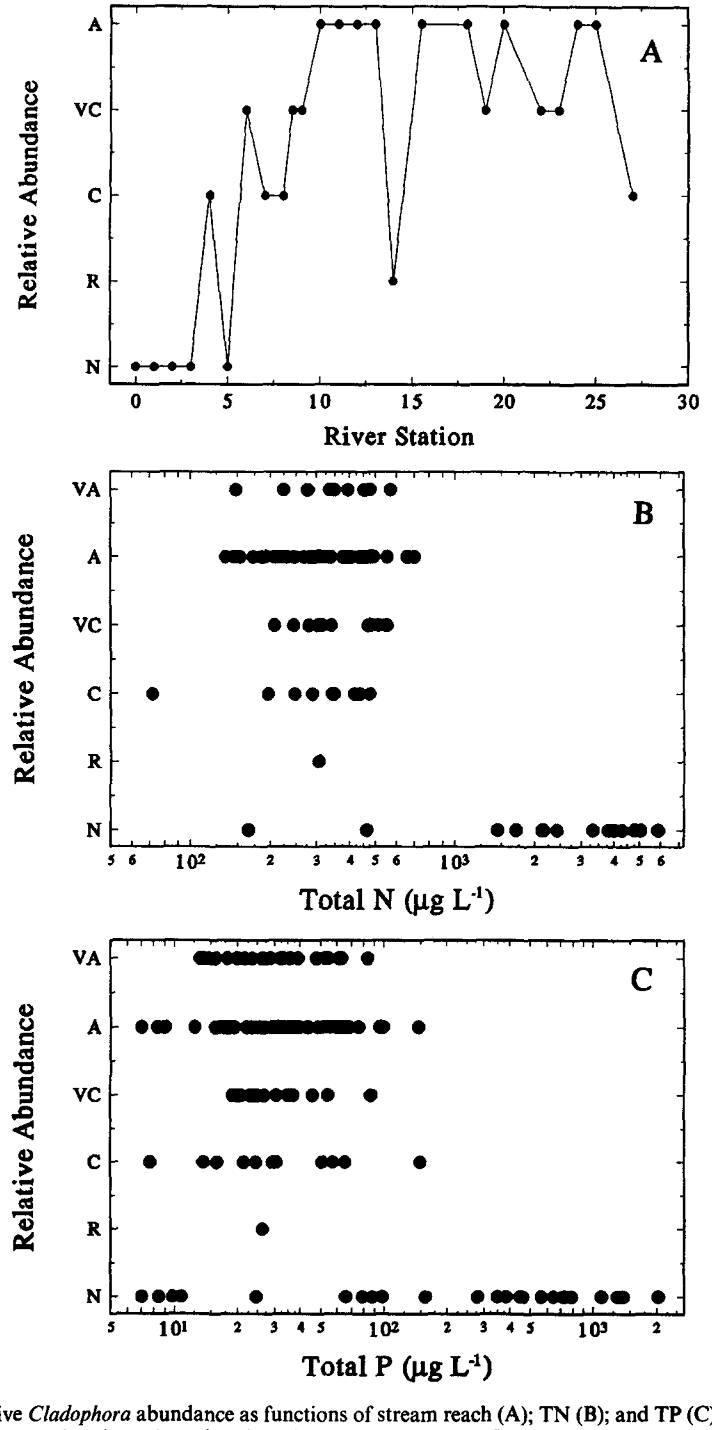

Fig. 4. Relative *Cladophora* abundance as functions of stream reach (A); TN (B); and TP (C). Abundance scores: VA = very abundant, A = abundant, VC = very common, C = common, R = rare, and N = not present.

ratio is used to set TP, and TN is varied to reach target chlorophyll *a* concentrations according to equation (17) (Table 2). A similar approach using equation (14) (Table 2), setting TN by the Redfield ratio, and instead varying TP to reach target chlorophyll *a* concentrations suggests that 35 $\mu g\, l^{-1}$ TP correspond with maximum values around 100 mg m^{-2} chlorophyll *a* (Table 5). In contrast, the target TP value (20.5 $\mu g\, l^{-1}$) derived from actual instream nutrient measurements at the reference stations was somewhat lower. We, thus, conclude that a provisional target of 30 $\mu g\, l^{-1}$ TP may control peak algal biomass below 150 mg m^{-2} chlorophyll *a*.

Management of Cladaphora *growth.*

Cladophora is a common nuisance alga in enriched streams, rivers and lakes (Whitton, 1970; Dodds and Gudder, 1992), and control of its growth is very difficult (Dodds, 1991). In the Clark Fork River, *Cladophora* was occasionally abundant at some stations (Fig. 4A). *Cladophora* appears to be N-limited in the Clark Fork (Lohman and Priscu, 1992), and research suggests that this is also true in many other systems (Dodds and Gudder, 1992). However, we were unable to develop models that could be used to predict management scenarios that would lower algal biomass and simultaneously reduce *Cladophora* dominance. Control of *Cladophora* in the

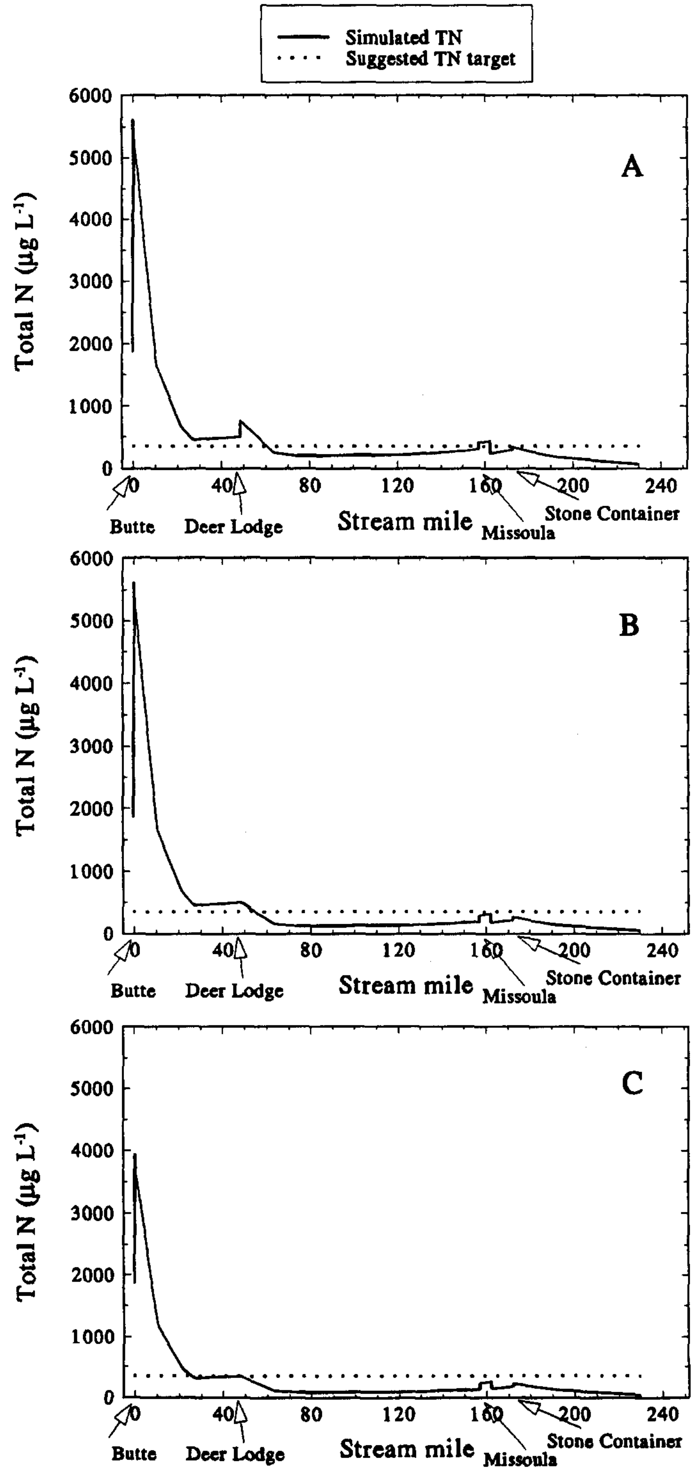

Fig. 5. TN loading scenarios used to calculate instream TN concentrations during normal summer low flows for no loading controls (A); complete removal of sewage from the city of Deer Lodge (B); and city of Butte limited to 7 mg l^{-1} and complete removal at Deer Lodge (C).

Clark Fork will be as difficult as it has been in other systems.

Instream summer loading policies

We conclude that significant reductions are needed in certain segments of the Clark Fork to reach instream targets of 350 $\mu g\,l^{-1}$ for TN and 30 $\mu g\,l^{-1}$ for TP. The spreadsheet nutrient model developed here can be used to define different combinations of controls that would maintain these targets throughout the basin under critical flow conditions. The spreadsheet model can also be used to evaluate the relative importance of point and nonpoint sources of nutrients, and to aid in the design of control strategies. For example, one alternative control strategy may be to augment flow during periods of low flow to dilute inflowing nutrients. The spreadsheet model could easily be modified to simulate such a control strategy. However, the model currently includes only nutrient loading and concentrations and does not predict instream biological responses such as periphyton growth levels.

SUMMARY

We conclude that strategies to control external nutrient loading and manage stream eutrophication should not be based on instream DIN and SRP levels, because instream dissolved inorganic nutrient concentrations are poorly related to benthic algal biomass. Management of external nutrient loading instead should be based upon instream TN and TP concentrations. Our three approaches (regression, probabilistic and reference reaches) converge on similar suggested targets for nutrient control. A general probabilistic analysis using a global data set suggests that, if TN values are maintained below a target concentration of 350 $\mu g\,l^{-1}$, chlorophyll *a* should not exceed 100 mg m^{-2} in most cases (Fig. 3). The validity of this target TN concentration is supported independently by water chemistry data from six reference reaches of the Clark Fork that generally do not exhibit subjectively determined excessive levels of chlorophyll *a*; the mean TN for these reference reaches is 317 $\mu g\,l^{-1}$ TN.

If the instream TP concentrations are maintained below 13% of the target TN value (target = 45.5 $\mu g\,l^{-1}$ TP), a high probability exists that the effectiveness of TN control measures will not be impaired by the nuisance growth of nitrogen-fixing algal species. By comparison, the mean TP concentration from the six reference reaches on the Clark Fork was 20.5 $\mu g\,l^{-1}$. The regression calculation using the global data set suggests that keeping TP below 35 $\mu g\,l^{-1}$ and TN below 252 $\mu g\,l^{-1}$ will control algal biomass below 100 mg m^{-2} chlorophyll *a* (Table 4). We suggest 30 $\mu g\,l^{-1}$ TP as a target level that will control algal biomass.

We stress that our recommended TN and TP targets may not apply to all systems. The final target values of TN and TP that are chosen for water quality protection in streams in other geographical locations ultimately must be based upon the objective criteria that local managers consider to be most relevant. In addition, the proposed levels of TN and TP may be unattainable in some ecoregions depending upon local geology, sediment contamination and other factors.

An additional benefit of using TN and TP values to control nutrients in the river as opposed to DIN and SRP levels involves consideration of nonpoint sources. Most information linking land use practices with nutrient input to aquatic systems uses TN and TP, not DIN or SRP. Building a regulatory framework for stream nutrients around TN and TP input will assist in future efforts to control nonpoint sources.

Acknowledgements—This reserch was supported by the Montana Department of Health and Environmental Sciences through a grant from Stone Container Corporation. Discussions with V. Watson, G. Ingman and J. Priscu improved the focus of this work, P. McCormick and three anonymous reviewers provided valuable critiques of the manuscript. K. Lohman helped conceptualize and construct the global data set, J. Jones provided data. D. Gudder provided technical assistance. This is contribution no. 96-439-J from the Kansas Agricultural Experiment Station.

REFERENCES

Bahls L. L. (1989) An assessment of water quality in the Clark Fork River and its tributaries based on the structure and composition of summer algae associations in the stream-bottom periphyton community. Report to the State of Montana Department of Health and Environmental Sciences, Water Quality Bureau, Helena, Montana, December 1989.

Biggs B. J. F. (1988) Artificial substrate exposure times for periphyton biomass estimates in rivers. *New Zeal. J. Mar. Freshwater Res.* **22,** 507–515.

Biggs B. J. F. (1995) The contribution of flood disturbance, catchment geology and land use to the habitat template of periphyton in stream ecosystems. *Freshwater Biol.* **33,** 419–438.

Biggs B. J. F. and Close M. E. (1989) Periphyton biomass dynamics in gravel bed rivers: the relative effects of flows and nutrients. *Freshwater Biol.* **22,** 209–231.

Bothwell M. L. (1985) Phosphorus limitation of lotic periphyton growth rates: an intersite comparison using continuous-flow troughs (Thompson River system, British Columbia). *Limnol. Oceanogr.* **30,** 527–542.

Bott T. L., Brock J. T., Dunn C. S., Naiman R. J., Ovink R. W. and Petersen R. C. (1985) Benthic community metabolism in four temperate stream systems: an inter-biome comparison and evaluation of the river continuum concept. *Hydrobiologia* **123,** 3–45.

Bradford M. E. and Peters R. H. (1987) The relationship between chemically analyzed phosphorus fractions and bioavailable phosphorus. *Limnol. Oceanogr.* **32,** 1123–1137.

Brylinsky M. and Mann K. H. (1975) The influence of morphometry and of nutrient dynamics on the productivity of lakes. *Limnol. Oceanogr.* **20,** 666–667.

Canfield D. E. Jr (1983) Prediction of chlorophyll *a* concentrations in Florida lakes: the importance of phosphorus and nitrogen. *Wat. Res. Bull.* **19,** 255–262.

Cooke G. D., Welch E. B., Peterson S. A. and Newroth P. R. (1993) *Restoration and Management of Lakes* *and Reservoirs*. 2nd edn, 548 p. Lewis Publishers, Boca Raton.

Delong M. D. and Brusven M. A. (1992) Patterns of periphyton chlorophyll *a* in an agricultural nonpoint source impacted stream. *Wat. Res. Bull.* **28,** 731–741.

Dodds W. K. (1991) Factors associated with the dominance of the filamentous green alga *Cladophora glomerata*. *Wat Res.* **25,** 1325–1332.

Dodds W. K. (1993) What control levels of dissolved phosphate and ammonium in surface waters? *Aquat. Sci.* **55,** 132–142.

Dodds W. K. and Gudder D. A. (1992) The ecology of *Cladophora*. *J. Phycol.* **28,** 415–427.

Flemer D. A. (1970) Primary productivity of the North Branch of the Raritan River, New Jersey. *Hydrobiologia* **35,** 273–296.

Funk W. H., Gibbons H. L., Duffner R. M., Notestine T. and Nielsen T. (1983) Water quality of the upper Spokane River and evaluation of methods for measurement of the effect of effluent upon primary and secondary producers. Research Project Technical Completion Report, College of Engineering, Civil and Environmental Engineering. Washington State University, Pullman, WA.

Heiskary S. A. and Walker W. W. (1988) Developing phosphorus criteria for Minnesota lakes. *Lake Reserv. Mngmt.* **4,** 1–9.

Hornberger G. M., Kelly M. G. and Cosby B. J. (1977) Evaluating eutrophication potential from river community productivity. *Wat. Res.* **11,** 65–69.

Horner R. R., Welch E. B. and Veenstra R. B. (1983) Development of nuisance periphytic algae in laboratory streams in relation to enrichment and velocity. pp. 121–134. In *Periphyton of Freshwater Ecosystems* (Edited by Wetzel R. G.). Dr W. Junk Publishers, The Hague.

Horner R. R., Welch E. B., Seeley M. R. and Jacoby J. M. (1990) Responses of periphyton to changes in current velocity, suspended sediment and phosphorus concentration. *Freshwater Biol.* **24,** 215–232.

Ingman G. L. (1992a) A rationale and alternatives for controlling nutrients and eutrophication problems in the Clark Fork river basin. State of Montana, Department of Health and Environmental Sciences. Section 525 of 1987 Clean Water Act amendments.

Ingman G. L. (1992b) Assessment of phosphorus and nitrogen sources in the Clark Fork river basin. State of Montana, Department of Health and Environmental Sciences. Section 525 of 1987 Clean Water Act amendments.

Jones J. R., Smart M. M. and Burroughs J. N. (1984) Factors related to algal biomass in Missouri Ozark streams. *Verh. Internat. Verein. Limnol.* **22,** 1867–1875.

Klewer J. A. and Holm H. W. (1980) The phosphorus-chlorophyll *a* relationship in periphytic communities in a controlled ecosystem. *Hydrobiologia* **94,** 173–176.

Lembi C. A., O'Neal S. W. and Spencer D. F. (1988) Algae as weeds: economic impact, ecology, and management alternatives. pp. 455–481. In *Algae and Human Affairs*. (Edited by Lembi C. A. and Waaland J. R.). Cambridge, England.

Lenon H. L., King R. H., Wujek D. E. and Bailey R. E. (1979) Survey and evaluation of the water quality, Tittabawassee River, Near Midland, Michigan 1978–1979. Department of Biology Limnological Services Technical Report no. 2. Central Michigan University, Mt Pleasant, MICH.

Lohman K. (1988) Nutrient sources and the influence of nutrients on periphyton in Northern Ozark border streams. PhD thesis, University of Missouri, Columbia, Missouri. 209 pp.

Lohman K. and Priscu J. C. (1992) Physiological indicators of nutrient deficiency in *Cladophora* (Chlorophyta) in the Clark Fork of the Columbia River, Montana. *J. Phycol.* **28,** 443–448.

Lohman K., Jones J. R. and Perkins B. D. (1992) Effects of nutrient enrichment and flood frequency on periphyton biomass in Northern Ozark streams. *Can. J. Fish. Aquat. Sci.* **49,** 1198–1205.

Luttenton M. R. (1982) An investigation of factors affecting the distribution and abundance of phycoperiphyton in Navigation Pool No. 5, Upper Mississippi River. Master's thesis, University of Wisconsin, LaCrosse, WISC.

Lyford J. H. Jr and Gregory S. V. (1975) The dynamics and structure of periphyton communities in three Cascade Mountain streams. *Verh. Internat. Verein. Limnol.* **19,** 1610–1616.

Marcus M. D. (1980) Periphytic community response to chronic nutrient enrichment by a reservoir discharge. *Ecology* **61,** 387–399.

Meyer J., Crocker T., D'Angelo D., Dodds W., Findlay S., Oswood M., Repert D. and Toetz, D. (1993) Stream research in the Long-Term Ecological Research network. LTER Publication no. 15. Seattle, WA.

Naiman R. J. (1980) The Matamek research program: annual report for 1979. Woods Hole Oceanographic Institution, Woods Hole, MS.

Naiman R. J. (1981) The Matamek research program: annual report for 1980. Woods Hole Oceanographic Institution, Woods Hole, MS.

Naiman R. J. (1983) The annual pattern and spatial distribution of aquatic oxygen metabolism in Boreal Forest Watersheds. *Ecol. Monogr.* **53,** 73–94.

Nielsen T. S., Funk W. H., Gibbons H. L. and Duffner R. M. (1984) A comparison of periphyton growth on artificial and natural substrates in the Upper Spokane River. *Northwest Sci.* **58,** 243–248.

OECD (Organization for Economic Cooperation and Development) (1982) Eutrophication of waters. Monitoring assessment and control. Final report. OECD Cooperative Programme on monitoring of Inland Waters (Eutrophication Control), Environment Directorate, OECD, Paris, 154 p.

Perrin C. J., Bothwell M. L. and Slaney P. A. (1987) Experimental enrichment of coastal stream in British Columbia: effects of organic and inorganic additions on autotrophic periphyton production. *Can. J. Fish. Aquat. Sci.* **44,** 1247–1256.

Peterson B. J., Hobbie J. E., Hershey A. E., Lock M. A., Ford T. E., Vestal J. R., McKinley V. L., Hullar M. A. J., Miller M. C., Ventullo R. M. and Volk G. S. (1985) Transformation of a tundra river from heterotrophy to autotrophy by addition of phosphorus. *Science* **229,** 1383–1386.

Reckhow K. H. and Chapra S. C. (1983) *Engineering Approaches for Lake Management*, Vol. 1, p. 340. Ann Arbor Science, Butterworth Publishing Co., Woburn, MASS.

Rosemond A. D. (1994) Multiple factors limit seasonal variation in periphyton in a forest stream. *J. N. Am. Benthol. Soc.* **13,** 333–345.

Ryding S.-O. and Rast W. (1989) *The Control of Eutrophication of Lakes and Reservoirs*. Vol. 1, p. 314. UNESCO, Paris and The Parthenon Publishing Group, United Kingdom.

Sand-Jensen K., Møller J. and Olesen B. H. (1988) Biomass regulation of microbenthic algae in Danish lowland streams. *Oikos* **53,** 332–340.

SAS (1988) Users Guide 6.03. SAS Institute, Cary, NC. 1028 pp.

Sas H. (1989) *Lake Restoration by Reduction of Nutrient Loading: Expectations, Experiences, Extrapolations*. Academia Verlag, Richarz.

Smart M. M. (1980) Stream-watershed relationships in the Missouri Ozark plateau province. PhD dissertation, University of Missouri, Columbia, Missouri.

Smart M. M., Jones J. R. and Sebaugh J. S. (1985)

Stream-watershed relations in the Missouri Ozark plateau province. *J. Environ. Qual.* **14,** 77–82.
Smith V. H. (1982) The nitrogen and phosphorus dependence of algal biomass in lakes: an empirical and theoretical analysis. *Limnol. Oceanogr.* **27,** 1101–1112.
Stockner J. G. and Shortreed K. R. S. (1976) Autotrophic production in Carnation Creek, a coastal rainforest stream on Vancouver Island, British Columbia. *J. Fish. Res. Board Can.* **33,** 1553–1563.
Tate C. M. (1990) Patterns and controls of nitrogen in tallgrass prairie streams. *Ecology* **71,** 2007–2018.
Van Nieuwenhuyse E. E. and Jones J. R. (1996) Phosphorus–chlorophyll relationship in temperate streams and its variation with stream catchment area. *Can. J. Fish. Aquat. Sci.* **53,** 99–105.
Vansteenburg J. B. (1983) Effects of the dairyland power cooperative electrical generating facility on the phycoperiphyton in Navigation Pool no. 9, Upper Mississippi River. Master's thesis. University of Wisconsin, LaCrosse WISC.
Vollenweider R. A. (1968) Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and phosphorus as factors in eutrophication. Technical report DAS/CSI/68.27, Environmental Directorate, Organization for Economic Cooperation and Development (OECD), Paris, 154 p.
Watson V. (1989) Maximum levels of attached algae in the Clark Fork River. *Proc. Mont. Acad. Sci.* **49,** 27–35.
Watson V., Berlind P. and Bahls L. (1990) Control of algal standing crop by P and N in the Clark Fork River. *Proc. 1990 Clark Fork River Symposium.*
Weber E. E. (1991) An assessment of water quality in the Clark Fork River and its major tributaries, based on the structure and composition of summer algae associations in the periphyton community. Report to the State of Montana Department of Health and Environmental Sciences, January 1991.
Weber E. E. (1993) An assessment of biological integrity and impairment of aquatic life in the Clark Fork River and its major tributaries based on the structure and composition of algae associations in the periphyton community. Report to the State of Montana Department of Health and Environmental Sciences, July 1993.
Welch E. B., Quinn J. M. and Hickey C. W. (1992) Periphyton biomass related to point-source nutrient enrichment in seven New Zealand streams. *Wat. Res.* **26,** 669–675.
Welch E. B., Jacoby J. M., Horner R. R. and Seeley M. R. (1988) Nuisance biomass levels of periphytic algae in streams. *Hydrobiologia* **157,** 161–168.
Whitton B. A. (1970) The biology of *Cladophora* in freshwaters. *Wat. Res.* **4,** 457–476.
Wong S. L. and Clark B. (1976) Field determination of the critical nutrient concentrations for *Cladophora* in streams. *J. Fish. Res. Board Can.* **33,** 95–96.