Freshwater Biology (1998) **40,** 145–158

APPLIED ISSUES

Primary nutrients and the biotic integrity of rivers and streams

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SUMMARY

1. Controls to reduce loadings of primary nutrients to maintain biotic integrity in rivers and streams have not been widely implemented because the relation between nutrients and chlorophyll, and its consequences for higher trophic levels, is confounded in lotic ecosystems by their openness, the variable degree of nutrient limitation and by the effect of physical factors.
2. The relationship between primary nutrients and biotic integrity in rivers and streams was tested using biological, physical and chemical information collected since 1982 from similar locations in streams throughout Ohio using standard procedures.
3. There was a negative correlation between nutrients, especially total phosphorus, and biotic integrity. The deleterious effect of increasing nutrient concentration on fish communities in low order streams was detectable when nutrient concentrations exceeded background conditions (total inorganic nitrogen and phosphorus > 0.61 mg L^{-1} and 0.06 mg L^{-1}, respectively).
4. These results suggest that the control of release of toxins and oxygen-demanding wastes to rivers is insufficient to protect aquatic life, and confirm the importance of non-point sources of pollution in catchment planning as well as the combined effect of habitat and riparian quality on nutrient assimilation.

Introduction

Historically, pollution control management in streams and rivers has focused on the abatement of gross impacts from untreated domestic sewage and industrial discharges. Although the negative effects of nutrient enrichment on rivers and streams received attention in the early 1970s (Cole, 1973), strategies to control the loading of primary nutrients have been largely directed at reducing eutrophication of lakes or estuaries (e.g. the 1972 Great Lakes Water Quality Agreement). Although a phosphorus–chlorophyll relationship (Van Nieuwenhuyse & Jones, 1996), the nutrient limitation of periphyton (Wu, Bowker & Antoine, 1996; Chessman, Hutton & Burch, 1992) and bottom-up control in streams (Deegan & Peterson, 1992) have been demonstrated, the control of primary nutrients to maintain biotic integrity has not been widely implemented. Control efforts for streams have lagged behind those for lakes because the relationship between nutrients and chlorophyll, and its possible consequences for higher trophic levels, is confounded in streams by light limitation (i.e. turbidity and shading; Lowe, Golladay & Webster, 1986), the frequency of spates (Lohman, Jones & Perkins, 1992), grazing (McCormick, 1994; Power, 1990a; Stewart, 1987), rapid nutrient cycling (Mulholland *et al.*, 1995), catchment area (Van Nieuwenhuyse & Jones, 1996; Wu *et al.*, 1996), input source and form (Newbold, 1992) and the variable

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nature of nutrient limitation in lotic ecosystems (Newbold, 1992; Pan & Lowe, 1995; Wu *et al*., 1996; Welch *et al*., 1988). However, Van Nieuwenhuyse & Jones (1996) reported a positive curvilinear relationship between mean total phosphorus and chlorophyll for temperate streams, with the greatest increase in chlorophyll concentration per unit total phosphorus occurring at total phosphorus concentrations less than 0.1 mg L^{-1}. The relationship corresponds with Newbold's (1992) assertion that nutrient limitation in streams is restricted to (or most detectable in) near pristine conditions, and suggests that relatively small increments in nutrient concentrations in streams should have measurable effects on biological communities.

The effects on primary and secondary consumers of nutrient additions to steams can be unpredictable (Deegan *et al*., 1997). Maximum salmonid production in streams is strongly related to ionic strength (Kwak & Waters, 1997), and nutrient additions to an arctic river resulted in increased fish growth in one experiment (Deegan & Peterson, 1992), but not in another (Deegan *et al*., 1997). In temperate warmwater streams, where the food web is typically more complex than in coldwater streams, trophic responses to nutrient enrichment are likely to be indirect and unpredictable. Enrichment may alter relative and absolute abundances of periphyton and macroinvertebrates (Stevenson, 1997; Dudley, Cooper & Hemphill, 1986), which may affect secondary consumers through reduced drift (Kerans, 1996; Hildebrand, 1974), changes in food quality (Fuller & Fry, 1991; Hayward & Margraf, 1987) or indirectly by influencing foraging efficiency through increased structural complexity (Power, 1990b; Savino & Stein, 1982). By depleting dissolved oxygen through nocturnal respiration, high standing stocks of periphyton can negatively affect sensitive fish and macroinvertebrates. Alternately, nutrients can be sequestered without measurable changes in community structure (McCormick, 1994; Stewart, 1987), especially in periphytic communities dominated by diatoms (Stevenson, 1997; Bothwell, 1989). Nevertheless, there should be some detectable change in the higher trophic levels of rivers and streams, especially for grazing invertebrates, with increased nutrient concentration (Wallace & Gurtz, 1986; Deegan *et al*., 1997). An important caveat, however, is that nutrients may serve as surrogates for other variables (e.g. habitat degradation, toxic substances) affecting community structure, given that nutrients can saturate algal growth rates at comparatively low concentrations (Bothwell, 1989).

The main objective of this study was to examine the relationship between primary nutrients and the biotic integrity of rivers and streams. Biotic integrity refers to the extent to which a community has a species composition, diversity and functional organization comparable to that expected for the natural habitat of a region (Karr & Dudley, 1981). The Ohio Environmental Protection Agency (Ohio EPA) has been collecting complementary biological, physical and chemical information from streams throughout Ohio using standard procedures since 1982, allowing us to assess this relationship using a correlative approach. Our objectives were fourfold. First, to determine any relationship between primary nutrients and fish or macroinvertebrates. Second, if a relationship exists, the amount of variation explained by either nitrogen or phosphorus compared to potential concomitant variables, especially habitat. Third, to quantify potential threshold levels of nutrients in streams beyond which fish or macroinvertebrate community structure is likely to be significantly degraded and, lastly, to investigate the relationship between macroinvertebrate and fish community structure and increasing concentrations of nutrients.

Methods

The data were obtained from spot samples of water quality, electrofishing and qualitative habitat assessments collected at 1657 sites by Ohio EPA after 1981. At each site, water samples were collected three-to-six times, electrofishing was performed one-to-three times at 3–6-week intervals, and habitats assessed using the Qualitative Habitat Evaluation Index (QHEI, Rankin, 1995) at least once in a given year. The QHEI is a qualitative visual assessment of functional aspects of stream macrohabitats (e.g. amount and type of cover, riparian width, siltation and channel morphology). Macroinvertebrate collections from multiple-plate artificial substratum samplers (Hester & Dendy, 1962) obtained from 901 locations coinciding with the fish and chemical sites were available for analysis. Fish community attributes were collectively measured with the Index of Biotic Integrity (IBI, Karr, 1981) modified for Ohio (Ohio Environmental Protection Agency, 1987; Yoder & Rankin, 1995). Macroinvertebrate community structure was measured by the Invertebrate Community Index (ICI, DeShon, 1995). The ICI is a multimetric measure of the invertebrate community composed of ten metrics based on structure and composition. The individual metrics were scored against expectations derived from least impacted reference sites.

Individual chemical parameters, IBI scores and select component IBI metrics measured at each site were averaged prior to data analysis. Chemical samples excluded prior to averaging included those collected during high stream flow events or those where either ammonia, nitrate, nitrite or phosphorus were not measured. Method detection limits (MDL) in mg L^{-1} for ammonia, nitrite, nitrate and phosphorus were 0.05, 0.02, 0.1 and 0.05, respectively. To comply with model assumption of analysis of variance and linear regression, concentrations of nitrite and nitrate falling below the MDL were randomly assigned values using a uniform distribution (based simply on inspection of distributions of values at and above the MDL) bounded by their MDLs and 0.001 mg L^{-1} for nitrite and 0.01 mg L^{-1} for nitrate. For ammonia and phosphorus, values below the MDL were randomly assigned to a discrete distribution over five intervals from 0.04 mg L^{-1} to 0.005 mg L^{-1} where probability for assignment at each interval was one half lower than the preceding one starting at $\rho = 0.5$. Lower boundaries were chosen *ad hoc* as lower boundaries in the literature are usually reported as vanishingly small or at or below detection limits (e.g. McCormick, 1994Van Nieuwenhuyse & Jones, 1996; Wu *et al*., 1996). In any given year, approximately half of all samples were collected from independent rivers, and therefore subject to independent errors. Two-to-five samples were collected from the same river in approximately 40% of cases, reflecting samples directed at point sources, and should therefore have uncorrelated sampling error. Up to thirty-two samples were collected from single large rivers in the remaining 10% of cases. In testing model aptness, multicolinearity and autocorrelation were not problematic as judged by variance inflation factors and the Durban–Watson test (Neter, Wasserman & Kutner, 1990).

The relationship of fish or macroinvertebrate community structure with the concentration of total inorganic nitrogen (TIN), total phosphorus (TP) and with habitat quality for four size classes of streams were tested in a nutrient and habitat model by the regression of IBI or ICI scores on habitat quality, $\log_{10}(TIN + 1)$, $TP^{-0.5}$ and an interaction term ($\log_{10}$ (TIN*TP)). Stream size classes were based on drainage area as follows: headwaters < 51.8 km^2, wadable streams 51.8 < 518.1 km^2, small rivers 518.1 < 2590.7 km^2 and large rivers $\geqslant 2590.7$ km^2. Potential concomitant water quality variables were handled in two ways. First, cases with mean ammonia-nitrogen (NH_3-N) concentrations $\geqslant 1.0$ mg L^{-1} (9% of all records) were excluded from all regressions because of the chronic toxicity of NH_3-N at the stream temperature and pH typical of Ohio streams (75th percentile values are 25.3 °C and 8.11 for maximum daily stream temperatures and pH, respectively). Those cases are usually sites with defined sources of pollution degrading water quality through a variety of pollutants; therefore, their exclusion lessens the potential influence of concomitant variables (e.g. dissolved oxygen, suspended solids, metals, etc.) on the nutrients and habitat regression model. Second, the degree to which either TP or TIN serve as a proxy for other variables in the nutrient-only model was examined using a stepwise regression model including the following variables: NH_3-N, minimum dissolved oxygen (DO), total suspended solids (TSS), chemical oxygen demand (COD), TP, TIN, TIN*TP and total recoverable Cd, Cr, Cu, Pb and Zn. All individual water quality determinants collected at individual sites were averaged and, with the exception of DO, $\log_{10}(x + 1)$ transformed prior to analysis. Because not all water quality variables were collected consistently, 841 and 523 samples matched respective fish and macroinvertebrate sites for all parameters. Other potential concomitant variables (e.g. polynuclear aromatic hydrocarbons, pesticides, other metals) were not considered because of small sample sizes, variable detection limits and because they rarely have detectable impact (Ohio Environmental Protection Agency, 1992). Chemical rather than biological oxygen demand was used as more records were available.

A categorical approach was also used to delineate potential threshold concentrations of TIN and TP beyond which fish and macroinvertebrates communities are likely to be degraded. This was accomplished by coding individual records to percentile distributions of TIN and TP by stream size (Table 1). An analysis of covariance was conducted using nutrient code as the independent variable and IBI or ICI as the dependent variable adjusted for habitat quality (QHEI). Because of relatively small cell sizes of nutrient code 4 in headwater and wadable categories, codes 4 and 5

Table 1 Scheme used to assign codes to nutrient concentrations (mg L^{-1}) of total inorganic nitrogen (TIN) and total phosphorus (TP) in four size classes of Ohio streams based on percentile distributions. Note that TP concentrations greater than or equal to the 90th percentile were placed in category 4, whereas those for TIN were placed in category 3, owing to the generally more limiting nature of TP at the respective concentrations (i.e. N:P > 30 when TIN ⩾ 90th percentile and TP < 50th percentile).

		Headwaters		Wadable streams		Small rivers		Large rivers	
	Percentile	P	N	P	N	P	N	P	N
	25^{th}	0.06	0.48	0.06	0.61	0.09	1.08	0.14	1.70
	50^{th}	0.17	1.37	0.12	1.65	0.16	2.04	0.27	2.65
	75^{th}	0.50	3.61	0.32	3.63	0.30	3.52	0.42	3.61
	90^{th}	1.70	7.71	0.98	6.35	0.45	4.94	0.60	4.40
Code		Number of observations							
1	both $\leqslant P_{25}$ N_{25}	50		71		52		18	
2	either $\leqslant P_{50}$ N_{50}	239		372		50		46	
3	$\leqslant P_{75}$, $\leqslant N_{90}$	51		111		169		38	
4	$> P_{75}$, $<> N_{90}$	13		18		30		0	
5	both $\geqslant P_{90}$ N_{90}	26		85		51		19	
6	$NH_3 \geqslant 1.0$ mg L^{-1}	72		57		18		1	

were combined. Possible underlying differences in IBI scores related to different nutrient concentrations were examined by a multiple analysis of variance using nutrient code as the independent variable, QHEI as the covariate and the following components of fish community structure as dependent variables:
total number of pollution sensitive species for a given record;
IBI metric score (i.e. 1, 3 or 5) for percentage tolerant fishes;
IBI metric score for percentage omnivorous fishes;
IBI metric score for percentage insectivorous fishes;
IBI metric score for percentage of carnivorous fishes;
the relative weight of all fishes in a sample.

The relationship between nutrients and macroinvertebrate community structure was determined by MANCOVA, as described for fish indices, again using QHEI as the covariate and the following dependent variables:
the number of Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa;
the percentage composition of mayflies;
the percentage composition of dipterans in the midge tribe Tanytarsini;
the percentage composition of other dipterans and non-insects (e.g. oligochaetes, snails, leeches);
abundance of scrapers, and abundance of all organisms as number per square metre on the artificial colonizer block.

Multiple comparisons of means by nutrient code were made using Scheffe's method (Neter *et al*., 1990).

Results

IBI scores were negatively correlated with increasing concentrations of TIN and TP in headwaters and wadable streams and TIN in small rivers, but no relationship was evident for large rivers (Table 2). Similarly, ICI scores were negatively correlated with increasing TIN and TP in headwaters and wadable streams in the nutrient and habitat model (Table 3). Comparison of results from the nutrient and habitat model with those obtained from the water quality model indicate that TIN serves exclusively as a surrogate for other water quality parameters in headwater and wadable streams (Tables 2 and 3). Ammonia-nitrogen accounted for a significant portion of the variation in IBI and ICI scores in headwaters and, to a lesser extent, in wadable streams, suggesting a nutrient effect due to nitrogen may still be important since NH_3-N is the preferred form for uptake by autotrophs. Regression slopes and correlation coefficients were similar between the nutrient and water quality models for TP, demonstrating that TP was not serving as a proxy for the variables tested in the water quality model. Though not proving cause and effect, this provides circumstantial evidence for a link between IBI scores and TP. TP did not explain variation

Table 2 Regression results for nutrient and habitat models (IBI = QHEI + TIN + TP) and for water quality models from stepwise selection (IBI = QHEI + TIN + TP + NH_3 + DO + COD + TSS + Cd + Cr + Cu + Pb + Zn). The interaction term was not significant and was therefore dropped from all models

Nutrients and habitat model						Water quality model – stepwise selection					
Variable	bx	SE	*t*	*P*	Type II R^2	Variable	bx	SE	*F*	*P*	Part. R^2
Headwaters											
Intercept	12.72	2.3	5.54	0.0001	NA	Intercept	17.56	7.66	5.25	0.0231	NA
QHEI	0.35	0.03	10.95	0.0001	0.2423	QHEI	0.33	0.05	45.63	0.0001	0.2656
TIN	–7.43	2.1	–3.54	0.0004	0.0324	NH_3	–11.66	4.51	6.69	0.0105	0.1017
TP	1.06	0.39	2.72	0.0069	0.0193	COD	–13.52	4.2	10.37	0.0015	0.0391
n = 378						TSS	6.92	2.78	6.19	0.0138	0.0137
						DO	0.6	0.26	5.37	0.0216	0.0131
						TP	1.05	0.52	4.09	0.0446	0.0126
						n = 186					
Wadable streams											
Intercept	10.43	1.65	6.31	0.0001	NA	Intercept	23.14	5.04	21.07	0.0001	NA
QHEI	0.32	0.02	15.77	0.0001	0.276	QHEI	0.27	0.03	92.08	0.0001	0.225
TIN	–5.10	1.25	–4.07	0.0001	0.0248	TP	1.95	0.35	31.69	0.0001	0.1563
TP	1.90	0.26	7.35	0.0001	0.0765	COD	–9.68	2.51	14.83	0.0001	0.0311
n = 655						NH_3	–11.78	3.40	12.03	0.0006	0.0182
						Cu	8.49	2.87	8.75	0.0033	0.0135
						Zn	–5.74	1.81	10.06	0.0016	0.0106
						n = 360					
Small Rivers											
Intercept	15.19	2.74	5.54	0.0001	NA	Intercept	22.48	4.68	23.09	0.0001	NA
QHEI	0.38	0.03	13.29	0.0001	0.3374	QHEI	0.32	0.038	69.63	0.0001	0.3119
TIN	–10.6	2.3	–4.61	0.0001	0.0577	Pb	–26	3.73	48.54	0.0001	0.1267
TP	0.42	0.48	0.88	0.3791	0.0022	TSS	6.15	2.18	7.94	0.0053	0.0187
n = 350						TIN	–5.89	2.64	4.93	0.0273	0.0116
						n = 230					
Large Rivers											
Intercept	16.32	6.13	2.66	0.0088	NA	Intercept	12.22	10.91	1.25	0.2671	NA
QHEI	0.3	0.06	5.44	0.0001	0.2018	Pb	–38.92	4.74	67.31	0.0001	0.389
TIN	–0.56	5.78	–0.10	0.9224	0.0001	QHEI	0.28	0.07	16.04	0.0002	0.1189
TP	–0.2	1.13	–0.18	0.8567	0.0003	Cu	28.93	9.71	8.87	0.0042	0.0597
n = 120						NH_3	–57.33	21.13	7.36	0.0086	0.0466
						n = 65					

in headwater ICI scores in the water quality model, being overshadowed by the strong effect of NH_3-N. In wadable streams, however, TP remained significant in the water quality model, suggesting it was not serving exclusively as a proxy.

Habitat generally explained between 20 and 30% of the proportion of variation in IBI or ICI scores across stream size and models, demonstrating the importance of habitat. By comparison, TP explained less variation in IBI or ICI scores, ranging from approximately 2–16% depending on stream size and model. Adjusting mean IBI or ICI scores using habitat as a covariate, demonstrated significant differences associated with different levels of nutrient concentrations (Figs 1 and 2). In headwater streams, sites having either TIN or TP concentrations falling below the 50th percentile ($N < 1.37$ mg L^{-1} and $P < 0.17$ mg L^{-1}) had significantly higher mean IBI scores than those with either TIN or TP concentrations exceeding the 50th percentile. The difference was most evident in wadable streams, where mean IBI scores were significantly lower at each successive category of increasing nutrient concentration, starting at a comparably low threshold (i.e. $N > 0.61$ mg L^{-1} and $P > 0.06$ mg L^{-1}). In contrast to the fish community, mean ICI scores did not differ in headwater streams, excluding sites with NH_3-

Table 3 Regression results for nutrient and habitat models (ICI = QHEI + TIN + TP + Interaction [N×P]) and for water quality models from stepwise selection (ICI = QHEI + TIN + TP + N×P + NH_3 + DO + COD + TSS + Cd + Cr + Cu + Pb + Zn)

Nutrient and Habitat Model						Water Quality Model – Stepwise Selection					
Variable	bx	SE	*t*	*P*	Type II R^2	Variable	bx	SE	*F*	*P*	Part. R^2
Headwaters											
Intercept	–7.31	5.33	–1.37	0.1727	NA	Intercept	9.22	4.69	3.86	0.0525	NA
QHEI	0.50	0.07	7.35	0.0001	0.2741	QHEI	0.41	0.07	34.71	0.0001	0.3461
TIN	–3.51	4.03	–0.87	0.3855	0.0053	NH_3	–49.85	10.13	24.20	0.0001	0.1423
TP	2.27	0.79	2.87	0.0048	0.0543	*n* = 89					
N×P	–15.93	5.29	–3.01	0.0031	0.0601						
n = 146											
Wadable streams											
Intercept	6.22	3.12	1.99	0.0468	NA	Intercept	16.15	6.32	6.52	0.0114	NA
QHEI	0.33	0.04	8.52	0.0001	0.1487	QHEI	0.29	0.05	31.55	0.0001	0.1675
TIN	–5.96	2.29	–2.61	0.0095	0.0161	NH_3	–23.83	6.12	15.17	0.0001	0.1163
TP	2.36	0.51	4.62	0.0001	0.0488	TP	1.38	0.67	4.27	0.0399	0.0529
N×P	–11.12	3.41	–3.30	0.0012	0.0250	DO	1.22	0.38	10.39	0.0015	0.0273
n = 418						Zn	–9.36	3.13	8.95	0.0031	0.0253
						n = 221					
Small rivers											
Intercept	16.82	4.16	4.05	0.0001	NA	Intercept	20.51	7.36	7.77	0.0059	NA
QHEI	0.34	0.04	8.00	0.0001	0.2077	QHEI	0.35	0.05	44.17	0.0001	0.2675
TIN	–3.15	3.48	–0.91	0.3652	0.0034	NH_3	–45.72	10.84	17.79	0.0001	0.0789
TP	–0.32	0.67	–0.48	0.6300	0.0010	TP	–2.80	0.75	14.00	0.0003	0.0470
N×P	–1.61	7.21	–0.22	0.8236	0.0002	Pb	–22.22	5.34	17.31	0.0001	0.0362
n = 247						TSS	9.30	3.10	9.03	0.0031	0.0299
						n = 168					
Large rivers											
Intercept	–9.26	12.93	–0.72	0.4762	NA	Intercept	–34.17	19.75	2.99	0.0911	NA
QHEI	0.58	0.12	4.63	0.0001	0.2113	NH_3	–124.10	47.81	6.74	0.013	0.3571
TIN	10.73	11.67	0.92	0.3604	0.0105	QHEI	0.77	0.15	24.77	0.0001	0.14
TP	–1.11	2.37	–0.47	0.6422	0.0027	Pb	–40.04	8.13	24.23	0.0001	0.1057
N×P	–20.29	18.61	–1.09	0.2789	0.0148	TSS	28.21	8.99	9.84	0.0032	0.0769
n = 83						*n* = 45					

N $\geqslant$ 1.0 mg L^{-1}. However, differences in mean ICI scores associated with increasing nutrient concentration were appreciable in wadable streams. A visual depiction summarizing the relationships between mean IBI or ICI scores and quartile ranges of TP and TIN (as a surrogate for other water quality variables) in wadable steams (Fig. 3) shows that the magnitudes of slopes associated with TP and TIN are roughly similar for the IBI. This suggests that nutrient enrichment and overall water quality, barring gross pollution, may have equal influence on fish communities in small streams. The relationship is more complex for the macroinvertebrate community, coinciding with the significant interaction term. Macroinvertebrates appear to be influenced most strongly by water quality parameters at high concentrations, represented by concentrations of TIN > 75th percentile. Differences were ill-defined in small rivers and absent in large rivers, reflecting the limited association with nutrients demonstrated by the regression functions. Mean IBI or ICI scores at sites having concentrations of NH_3-N $\geqslant$ 1.0 mg L^{-1} were usually significantly lower than all other categories across stream size.

Sensitive fishes, insectivores and top carnivores generally showed a negative relationship to increasing nutrient concentration in headwaters, wadable streams and small rivers (Table 4). The number of sensitive species showed the strongest differences corresponding to increased nutrient concentrations, with sites in category 1 and 2 grouping separately from those with

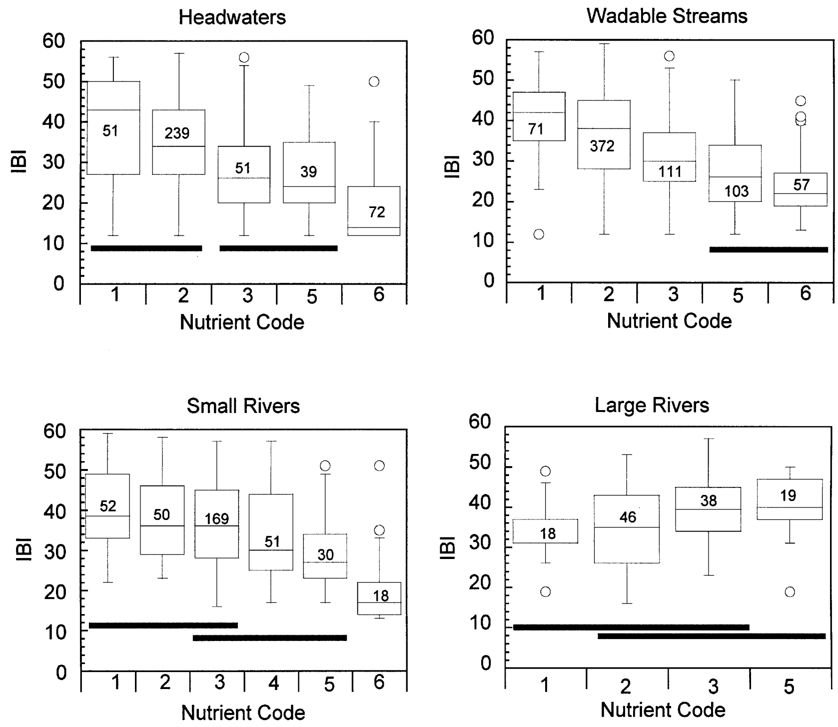

Fig. 1 Distributions of the Index of Biotic Integrity arranged by percentile values of total phosphorus and total inorganic nitrogen (see Methods for details on nutrient codes) for four stream size classes based on drainage area. Distributions with similar means are underlined. Number of sites are indicated inside individual boxes. Boxes enclose 50% of the data, whiskers encompass all values except those exceeding either the upper or lower quartiles by 1.5 times the interquartile distance.

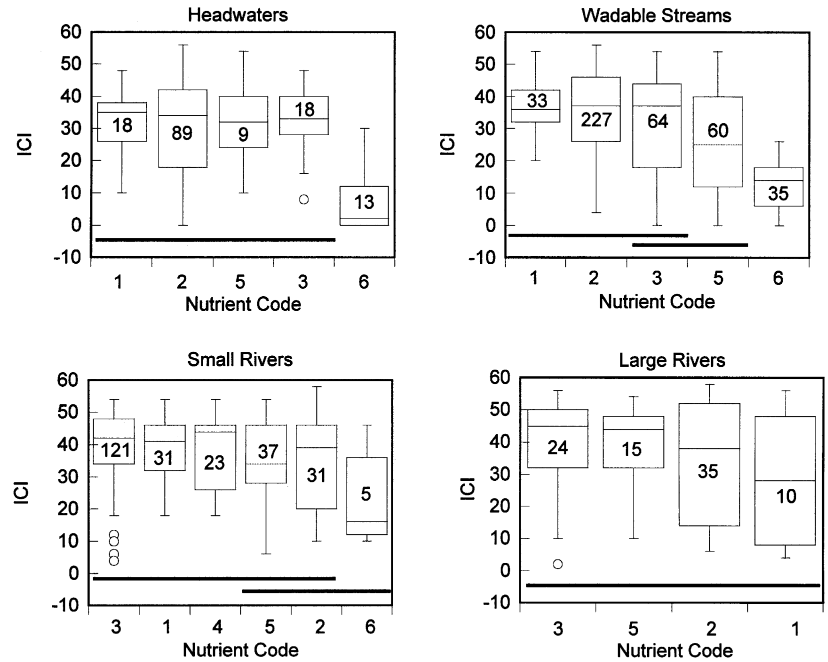

Fig. 2 Distributions of Invertebrate Community Index Scores arranged by percentile values of total phosphorus and total inorganic nitrogen (nutrient code) for four stream size classes based on drainage area. Distributions with similar means are underlined.

either TIN or TP exceeding the 50th percentile. The relative abundance of top carnivores decreased with increasing nutrient concentration in wadable streams and small rivers. Abundance of insectivores showed a negative relationship with increasing nutrient concentration in headwaters and wadable steams. Relative abundance of tolerant and omnivorous fishes increased significantly in relation to nutrient enrichment in headwaters, wadable streams and small rivers. A trend toward the highest relative abundance at

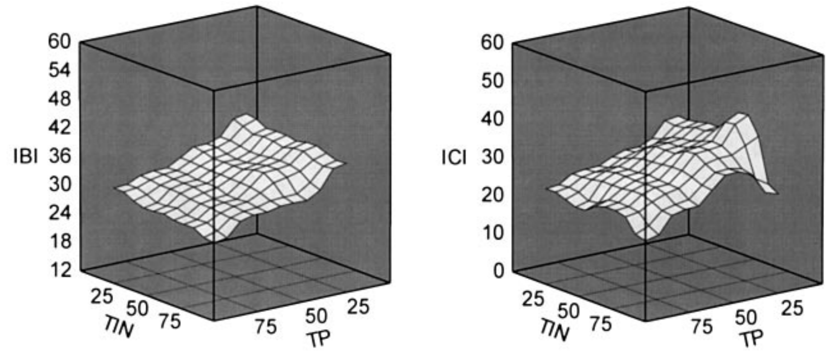

Fig. 3 Three-dimensional surface plots of mean IBI or ICI scores plotted by quartile ranges of total inorganic nitrogen and total phosphorus in wadable steams visually showing the relative magnitude of slopes given by the nutrient only regression model.

Table 4 Response of select fish community attributes (number of pollution sensitive species, per cent tolerant fishes, per cent as omnivores, per cent as insectivores, per cent as carnivores and relative abundance) to different levels of nutrient concentration expressed as Scheffé groupings (underlined) of nutrient categories (see coding scheme in Table 1) ordered by decreasing mean responses. Comparisons were made at the α = 0.5 level of significance

Community attribute	Headwaters	Wadable streams	Small rivers	Large rivers
Sensitive fishes	1 2 3 5 6	1 2 3 5 6	1 2 3 5 4 6	3 5 2 1
Tolerant fishes	6 5 3 2 1	5 6 3 2 1	4 6 5 3 2 1	2 1 5 3
Omnivores	6 3 5 2 1	6 5 3 2 1	6 4 5 3 2 1	3 5 2 1
Insectivores	1 2 3 5 6	1 2 3 5 6	2 3 1 5 4 6	5 3 1 2
Top carnivores	NA	1 2 3 5 6	1 2 5 3 4 6	5 3 2 1
Rel. weight (wading)	2 3 1 5 6	2 3 6 1 5	2 4 3 5 6 1	NA
Rel. weight (boating)	NA	3 2 5 1 6	5 2 3 4 1 6	5 3 2 1

intermediate levels of enrichment was evident, especially in wadable streams. Emulating results of the regression of IBI on TIN and TP, no relationship between community attributes and nutrients was noted in large rivers, except for top carnivores which were positively associated with higher nutrient levels.

Macroinvertebrate community indices were also influenced by nutrient concentration, but to a lesser degree than for fish (Table 5). In wadable streams, the number of EPT taxa and the relative abundance of Tanytarsini midges decreased relative to increasing nutrient concentration, while other dipterans and non-insects were positively associated with increasing nutrient concentration, reflecting the response of mean ICI scores. The relative abundance of mayflies was highest at TIN and TP concentrations falling between the 50th and 75th percentiles (Code 3) in wadable streams. Abundance of scrapers was higher in headwater streams at sites having either TP or TIN concentrations exceeding the 25th percentile (Fig. 4). In wadable streams, scraper abundance in nutrient category 2 was significantly higher than abundance at nutrient category 5 or 6. No trends were apparent in small and large rivers. The slight positive response in the macroinvertebrate community to intermediate levels of nutrient enrichment in wadable streams is

Table 5 Response of invertebrate community attributes (number of Ephemeroptera, Plecotoptera and Trichoptera taxa (EPT); % mayflies; % Tanytarsini; % other dipterans and non-insects; and abundance) to different nutrient concentration, expressed as Scheffé groupings (underlined) of categories (see coding scheme in Table 1) ordered by decreasing mean responses. Comparisons were made at the $\alpha = 0.5$ level of significance

Community attribute	Headwaters	Wadable streams	Small rivers	Large rivers
EPT taxa	1 2 3 5 6	1 2 3 5 6	1 3 2 4 5 6	3 5 2 1
Per cent mayflies	3 1 5 2 6	3 1 2 5 6	4 3 1 5 2 6	3 5 2 1
Per cent Tanytarsini	1 2 3 5 6	1 2 3 5 6	1 4 3 5 2 6	3 2 1 5
Other dipterans and non-insects	6 5 3 2 1	6 5 2 3 1	6 2 5 4 1 3	1 2 5 3
Abundance	6 2 3 5 1	3 2 6 5 1	3 6 5 2 1 4	3 1 5 2

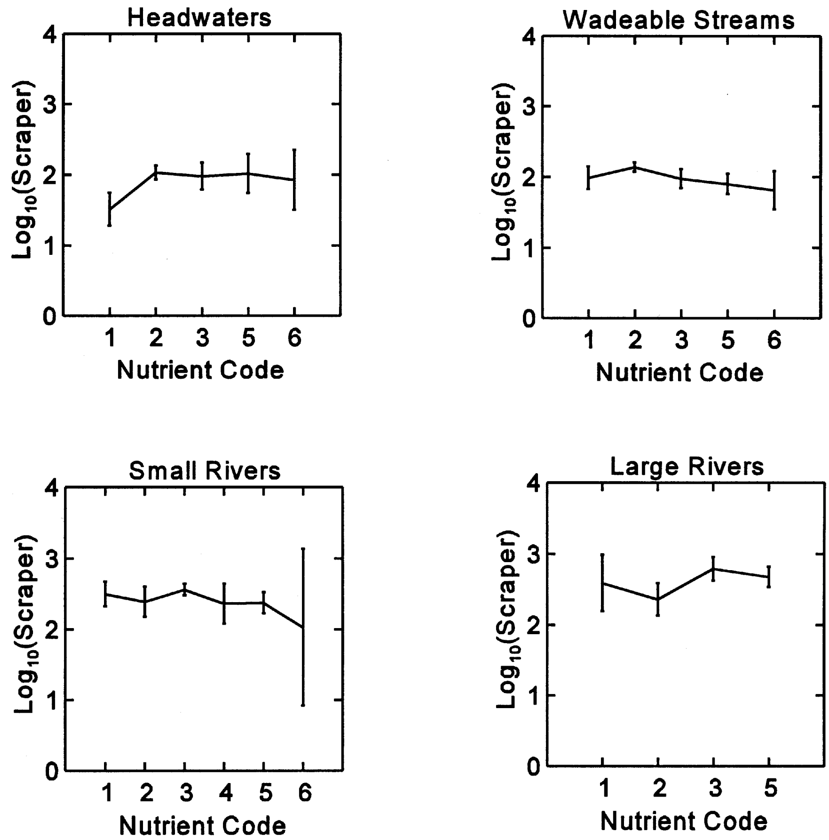

Fig. 4 Mean abundance of scrapers (Log_{10} Y ± 1 SE) plotted by nutrient categories (see Table 1) for four size classes of streams in Ohio.

evident in the plot of mean ICI scores by quartile distributions of TIN and TP, and reflects the significance of the interaction term in the nutrients-only regression model (Fig. 3). The effect of NH_3-N ≥ 1.0 mg L^{-1} (category 6) on macroinvertebrate community structure in headwaters and wadable streams was manifest in the increased relative abundance of other dipterans and non-insects, and decreased number or abundance of EPT taxa, mayflies and Tanytarsini midges relative to nearly all other nutrient categories.

Discussion

Our results show that biotic integrity of rivers and streams is negatively correlated with increasing nutrient concentration, especially phosphorus, and that differences in the indices are most defined in low order streams. A loss in the number of sensitive fish species, decreased relative abundance of top carnivores and insectivores, and an increasing proportion of tolerant or omnivorous fishes were correlated with nutrient enrichment. Collectively, these changes in fish community structure in headwaters resulted in a lowering of IBI scores by approximately five points when either TIN or TP concentrations exceeded the seventy-fifth percentile (3.61 mg L^{-1} and 0.31 mg L^{-1} for TIN and P, respectively). In wadable streams, IBI scores decreased by approximately four points with each successive categorical increase in nutrient concentration. Compared to other water quality parameters, TP and TIN did not explain a significant proportion of variation in IBI or ICI scores in small and large rivers. Existing reference conditions in small and large rivers, by which the IBI or ICI are calibrated, are highly enriched with respect to TP (e.g. median TP in the Eastern Cornbelt Plain of Ohio equals 0.12 and 0.40 mg l^{-1} for small and large rivers, respectively; compare with Table 1). Consequently, existing community structure may reflect the prevailing enriched conditions, and reduce sensitivity in the IBI or ICI to varying nutrient concentrations over the broad spatial scales investigated in this study. Also, the relationship is confounded in large Ohio rivers due to the disproportionate occurrence in nutrient categories 1 and 2 of samples from rivers having industrial pollution (Mahoning River), acid mine waste (Hocking River), or catchment scale habitat disturbances (Hocking and Maumee Rivers) that may override effects from nutrients.

Though the ICI was negatively correlated with increasing nutrient concentration, differences in macroinvertebrate community structure were less defined than in the fish community. Macroinvertebrates have been shown to respond positively to nutrient enrichment (Deegan & Peterson, 1992), and the relative abundance of mayflies, having a positive effect on ICI scores, was highest at intermediate levels of nutrient concentrations, especially in wadable streams in this study. Interestingly, the abundance of scrapers, the functional classification in the Ohio EPA database most closely matching grazers, was highest at sites in headwater streams having elevated nutrient levels (Fig. 4). The river continuum concept advanced by Vannote *et al*. (1980) suggests that grazer abundance should be low in headwater streams due to low autochthonous production. Assuming nutrient enrichment stimulated algal production in headwater streams, then the increase in grazers may be fitting a predictable response. Collectively then, although nutrient enrichment may have initiated changes in the macroinvertebrate community, negative responses in ICI metric scoring (e.g. percentage composition of other dipterans and non-insects) may have been offset by positive responses. Results of the water quality regression model also suggest that macroinvertebrate community structure may be more sensitive than are fish to water column chemistry.

Indirect effects through the food web may have been manifest in the loss of sensitive fish species, nearly all of which are classified as specialized insectivores (Ohio Environmental Protection Agency, 1987). Visually, an inverse relationship exists between the relative abundance of other dipterans and non-insects and numbers of EPT taxa, showing how macroinvertebrate community structure changes along a gradient of increasing environmental degradation (Table 3; Fig. 5). Loss of EPT taxa corresponds with a decreasing number of sensitive fishes (Fig. 5). Although the correspondence may reflect shared sensitivity to stressors, EPT taxa comprise a large proportion of the drift (Waters, 1972) on which specialized drift feeders (e.g. *Notropis photogenis*, Cope; *N. rubellus*, Agassiz) depend (Trautman, 1981; Etnier & Starnes, 1993). Similarly, an increased relative abundance of other dipterans and non-insects is associated with decreasing relative abundance of all insectivorous fishes and increasing relative abundance of omnivorous fishes. The overall trend toward decreasing macroinvertebrate diversity and increased abundance may favour omnivores over specialists.

Concentrations of NH_3-N $\geqslant$ 1.0 mg L^{-1} had a strong negative effect on IBI and ICI scores. Although metals and dissolved oxygen explained a significant portion of variation in IBI scores, NH_3-N explained substantially more variation. Whether the effect of NH_3-N concentrations $\geqslant$ 1.0 mg L^{-1} on IBI scores was due to chronic toxicity, nutrient enrichment or both, was not investigated; however, the clear separation of ICI scores in category 6, coupled with a loss of macroinvertebrate taxa and sensitive fishes relative to high nutrients alone (i.e. category 3 and 5), strongly suggests that chronic toxicity was involved. Although chlorine may be a confounding factor (Karr, Heidinger &

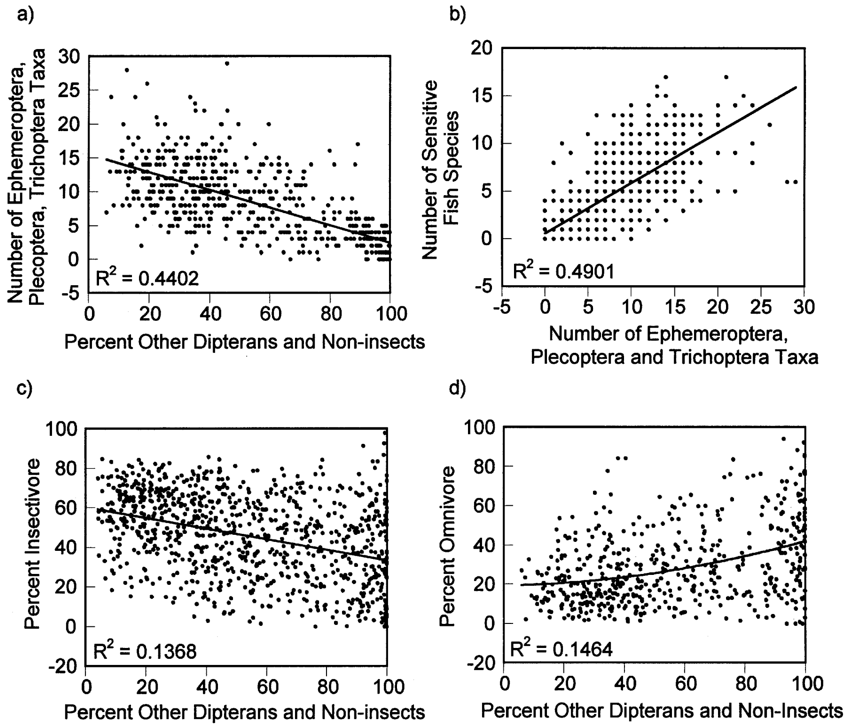

Fig. 5 Relationships between macroinvertebrate and fish community structure given as scatter plots of functional groups of varying tolerance to environmental disturbance: (a) number of EPT taxa and percentage composition other dipterans and non-insects; (b) number of sensitive fishes and EPT taxa; (c) percentage composition of insectivorous fishes to percentage other dipterans and non-insects and (d) percentage of omnivorous fishes to percentage of other dipterans and non-insects.

Helmer, 1985), a comparison of chlorine and NH_3-N effects on IBI scores from sixty-three sites near pollution control outfalls for which we had data, did not implicate chlorine (chlorine: $t = -0.45$, $P > 0.05$; ammonia: $t = -2.09$, $P = 0.04$). Given our limited data set with respect to chlorine, we are not suggesting its effects on aquatic communities are minimal; our combined results clearly show that ammonia does have a strong effect.

TP was important in explaining variation in IBI scores in headwaters and wadable streams and ICI scores in wadable streams. For regressions involving the IBI, the nearly identical slopes with respect to TP between the nutrient only and water quality regression models suggests that TP was not acting as a surrogate for the other variables compared. Moreover, separation between mean IBI scores at comparatively low nutrient concentrations, where other water quality parameters are not likely to be a factor, indicates a link between enriching effects of TP and biotic integrity. Whether differences in IBI scores were due directly to linked trophic responses, indirect effects (e.g. algal respiration and oxygen depletion) or both is unclear. Daytime dissolved oxygen concentrations are lowest and most variable at the least impacted reference sites with the highest TP concentrations (Fig. 6). As the DO data were collected during daylight, minimum values may

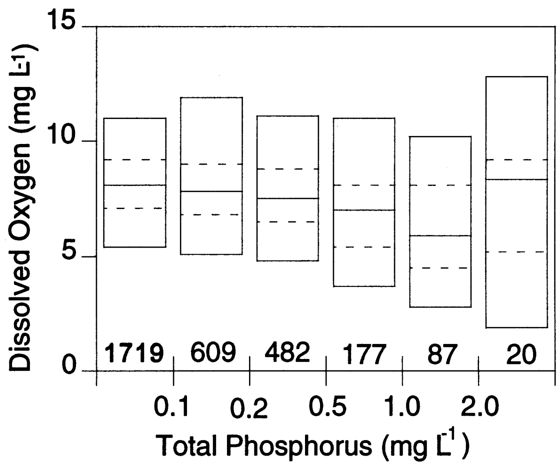

Fig. 6 Percentile distributions of dissolved oxygen for various ranges of total phosphorus for least impacted reference sites in the Ohio EPA database. Boxes enclose 90% of data points, lines inside boxes indicate quartiles.

be lower at night, implicating diurnal DO swings as a possible factor indirectly affecting biotic indices. Note, however, that DO values < 5.0 mg L^{-1} are typically associated with TP concentrations exceeding the 75th percentile defined in this study (see Table 1), and differences in IBI scores were detected at a lower TP concentration in wadable streams. The trend toward highest abundance of both fish and macroinvertebrates at intermediate nutrient concentrations might be

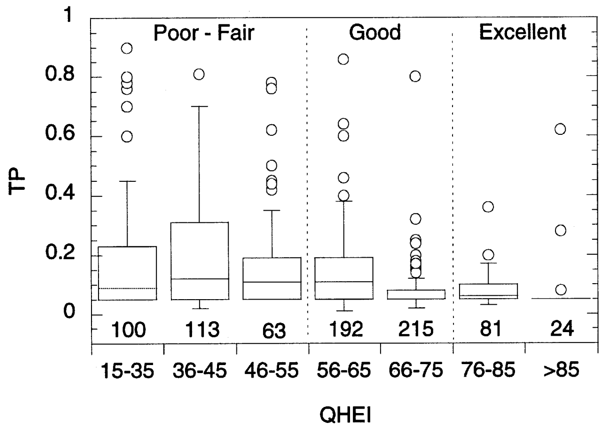

Fig. 7 Total phosphorus concentrations in relation to habitat quality for headwater streams in the intensively farmed Eastern Cornbelt plains of Ohio.

explained if nutrients stimulated production, and consequently fostered observed changes in fish and macroinvertebrate community structure. TIN did not explain a significant proportion of the variation in IBI or ICI scores in the water quality regression models, whereas concentrations of NH_3-N < 1.0 mg L^{-1} did, especially in headwaters and wadable streams. Because NH_3-N is more readily assimilated than oxidized forms by some algae (Wetzel, 1983), effects from nitrogen as a nutrient can not be ruled out. These results, on a broad scale, circumstantially support nutrient limitation (Pan & Lowe, 1995; Wu *et al.*, 1996) and phosphorus–chlorophyll relationships (Stockner & Shortreed, 1978; Bothwell, 1989; Van Nieuwenhuyse & Jones, 1996) documented for temperate streams, and the consequences of nutrient enrichment for higher trophic levels.

High TP concentrations across the range of stream sizes observed in this study reflects both phosphorus enrichment and degraded habitats. TP concentrations in Ohio streams are highest where habitat quality is most degraded (Fig. 7), and therefore caution against interpreting cause and effect between nutrients and biotic integrity. However, habitat quality was similar between nutrient categories 1 and 2 in wadable streams (based on distributions of QHEI scores; two sample *t*-test $t = 1.435$, $P = 0.152$) whereas mean IBI scores were different, suggesting that IBI differences were independent of habitat for that particular comparison. This provides further circumstantial evidence supporting a cause and effect relationship between nutrients and biotic integrity. Johnson *et al.* (1997) found that land use within the stream ecotone, defined as the 100 m stream buffer on each bank, explained more variance in summer TP concentrations than whole catchment land use in the Saginaw Bay drainage of Michigan, U.S.A., suggesting processes acting within the stream ecotone are important determinants of TP in streams. Streams with comparatively narrow wooded riparian buffers but intact physical habitat, have less measurable TP than those with degraded habitats in the intensively farmed Eastern Cornbelt region of Ohio (Fig. 7). This suggests that not only does adjacent land use influence nutrient export to the stream, but habitat quality within the stream may influence nutrient processing. Degraded stream channels with poorly developed riparian habitat exacerbate deleterious effects of residual nutrients via decreased riparian uptake, increased retention time due to siltation and wider channels, and by allowing full sunlight to reach the stream (Barling & Moore, 1994). Conversely, high quality habitats with mature, intact riparian zones may ameliorate potential adverse impacts of nutrients by terrestrial assimilation (with export later via leaf litter), by reducing sunlight and by reducing clay and silt loads to which nutrients are often adsorbed (Klotz, 1988).

Our results show that biotic integrity in low- to mid-order streams is negatively correlated with nutrients. Though nutrients may serve as surrogates for other variables or act indirectly in structuring biotic communities, some evidence for cause and effect was inferred because the influence of TP on the IBI appeared independent of habitat and other measured variables in wadable streams, at least up to the median TP concentration. Also, the abundance of fish and macroinvertebrates was generally highest at intermediate nutrient levels, and the abundance of grazers in headwater streams was highest at enriched sites, as one might predict from the river continuum concept (Vannote *et al.*, 1980). These results suggest control of toxins and oxygen demanding wastes to rivers may not be sufficient to protect aquatic life, and underscore the sensitivity of rivers to diffuse nutrient enrichment.

From a stream management perspective, both the sources and processing of nutrients into streams affect biotic integrity. Nutrient delivery to streams and stream habitat quality are influenced by the quality of riparian habitat (Doppelt *et al.*, 1993; Johnson *et al.*, 1997) and land use (Richards, Johnson & Host, 1996;

Allan, Erickson & Fay, 1997; Wang *et al.*, 1997). As both riparian vegetation and aquatic habitat quality affect how nutrients are assimilated, protecting existing high quality riparian buffers, or otherwise restoring them, is an obvious first step towards maintaining biotic integrity. Aspects of land use contributing nutrient loads, ranging from farming practices to urban stormwater runoff, must also be considered in catchment planning efforts to restore aquatic life.

Acknowledgments

Chris Yoder and the Ecological Assessment Unit staff provided support and helpful comments. Constructive and insightful reviews by three anonymous referees greatly improved this manuscript.

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(*Manuscript accepted 6 February 1998*)