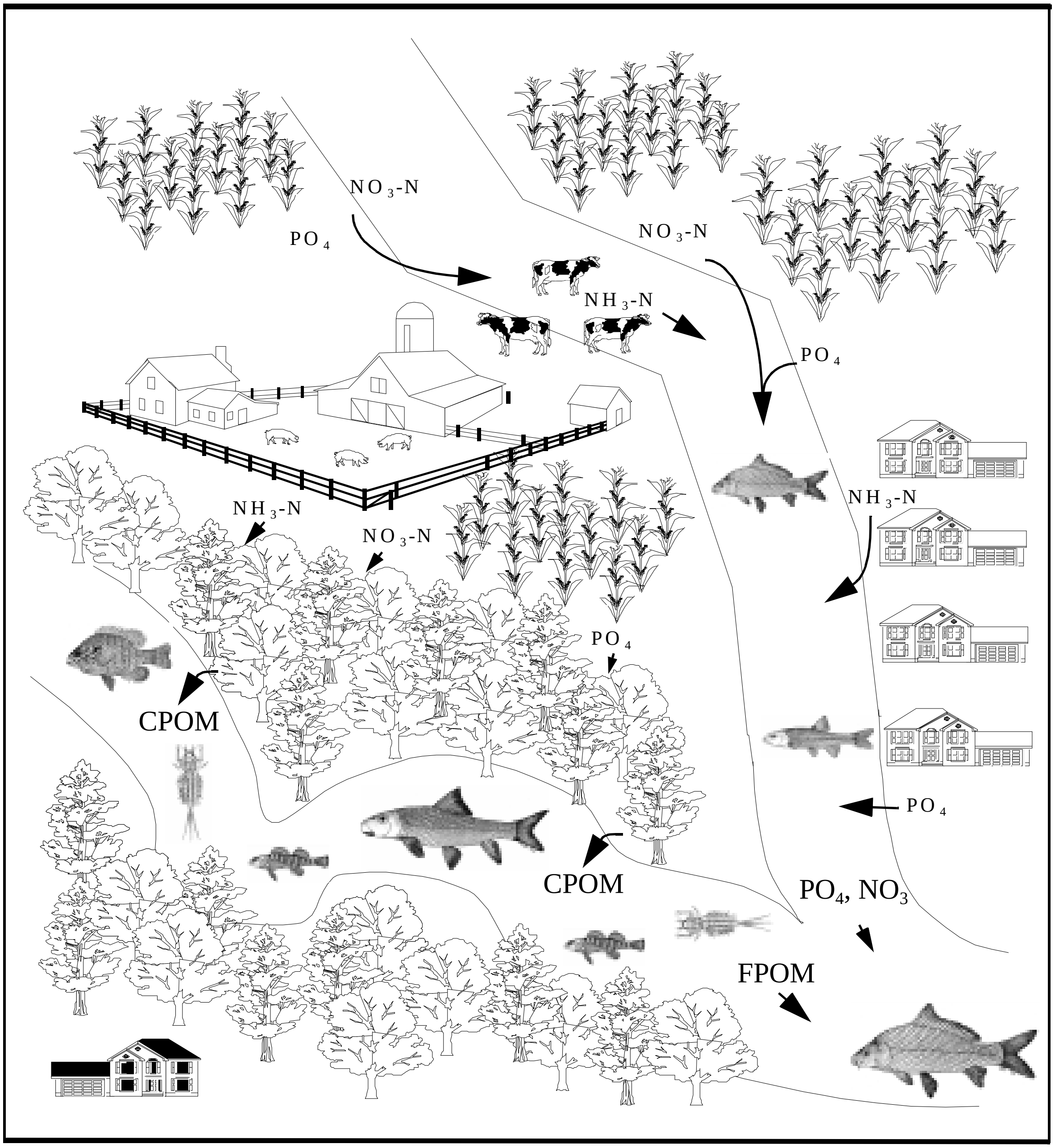
Association Between Nutrients, Habitat, and the Aquatic Biota in Ohio Rivers and Streams

Ohio EPA Technical Bulletin MAS/1999-1-1



Robert A. Taft, Govenor
Christopher Jones, Director
Ohio Environmental Protection Agency
P.O. Box 1049, Lazarus Government Center
122 S. Front Street, Columbus, Ohio 43216-1049

OhioEPA

Association Between Nutrients, Habitat, and the Aquatic Biota in Ohio Rivers and Streams[1]

Ohio EPA Technical Bulletin MAS/1999-1-1

January 7, 1999

Robert A. Taft, Govenor
Christopher Jones, Director
Ohio Environmental Protection Agency
P.O. Box 1049, Lazarus Government Center
122 S. Front Street, Columbus, Ohio 43216-1049

1. This is a technical bulletin and does not represent Ohio EPA policy.

ACKNOWLEDGEMENTS

Ed Rankin[2], Bob Miltner, Chris Yoder and Dennis Mishne authored this publication.
Please address inquiries to Ohio EPA, 1685 Westbelt Drive, Columbus, Ohio 43228.
This document is available electronically at http://chagrin.epa.state.oh.us.

This study could not have been accomplished without the extensive collection of biological and water quality samples collected by Ohio EPA Division of Surface Water staff over the 14 field seasons represented in this study. Ed Herricks provided helpful comments on an early version of this document. This study was funded by a Section 319 grant from the USEPA.

2. Author to whom correspondence should be addressed.

Table of Contents

List of Figures

Associations Between the Aquatic Biota, Habitat, and Nutrients in Ohio Rivers and Streams

SUMMARY

Nutrient chemistry, biological community performance, and habitat data from least impacted regional reference sites (REF) and a broader data set including sites (ALL) impacted by a variety of causes and sources were analyzed to determine the near-field (*i.e.,* localized) low-flow effects of nutrients and sediment on the aquatic assemblages of Ohio streams and rivers. Data were segregated by ecoregion and futher stratified by four ranges of stream and river size (headwater streams, 0-20 sq. mi.; wadeable streams, >20-200 sq. mi.; small rivers, >200-1,000 sq. mi.; and large rivers, >1,000 sq. mi.) for these analyses. The major conclusions of this study are:

• Headwater streams are important to the assimilation of nutrients and sediment in runoff in determining total maximum daily loads (TMDLs), and to the overall quality of downstream resources. Headwater streams compose 78% of the stream miles in Ohio that, in the aggregate, represent a significant source of assimilative capacity for the protection of downstream uses. The aggregate condition of headwater streams is correlated with the quality of water and aquatic life resources in larger streams, and reflects the integrity of the watershed as a whole.

• Wooded riparian buffers are a vital functional componet of the stream ecotone and are intstumental in the detention, removal and assimilation of nutrients from or by the water column. The riparian zone is essentially a component of instream habitat. It contributes food and nutrients in forms that desirable aquatic assemblages are adapted for, and contributes to the habitat heterogenaity by influencing channel morphology via large woody debris and bank stabilization. In short, riparian zones govern the quality of goods and services provided by riverine ecosystems by influencing the types of aquatic assemblages that can be sustained, water quality, and aesthetics

• The management of nonpoint sources of pollution and determining the assimilative capacity of a lotic system (*i.e*., TMDLs) needs to include more than dilution dynamics alone. Residual effects of nutrients and sediment are most manifest in measures of biological community performance (e.g., IBI or ICI) because of the ability of aquatic biota to integrate cumulative effects of multiple events. Measuring biological community performance also reduces the uncertainty regarding duration and exposure that are common to mass balance modeling approaches. The influence of the habitat and the biota on the ability of a watershed to assimilate nutrients and allied stressors (*e.g.*, silt, localized habitat modifications) must be considered in the development of management strategies to restore waters impared by nonpoint sources. Also, the recognition is needed that the functional extent of a stream or river goes beyond the wetted channel to include the flood plain so that more appropriate jurisdictional boundaries can be defined (*i.e.*, a better alternative to the ordinary highwater mark) that are relevant to the protection of aquatic life uses and the environment as a whole.

• Reference (REF) total phosphorus (TP) and nitrate-nitrogen (NO_3-N) concentrations differed between ecoregions with the highest background concentrations occurring in the Huron/Erie

Lake Plain (HELP) and Eastern Corn Belt Plains (ECBP) ecoregions, lowest in the Western Allegheny Plateau (WAP) ecoregion, and intermediate in the Erie-Ontario Lake Plane (EOLP) and Interior Plateau (IP) ecoregions.

- Reference (REF) TP and NO_3-N concentrations typically increased with stream size, especially so in large rivers. While nutrient concentrations are expected to increase in the larger mainstem rivers, the concentrations considered as "reference" are themselves indicative of enrichment that is largely the product of anthropogenic sources and activities.

- Degradation of biological communities (*i.e.*, biological integrity less than WWH criteria) was not observed until median nitrate-N exceeded 3-4 mg/l. This result, however, may be confounded by nitrate-N concentrations that remain elevated following high stream flows after flows returns to normal. A consequence of this is that distributions of low flow nitrate-N concentrations are highly skewed. Also, high nitrate-N concentrations are associated with WWTP discharges, and therefore may serve as a surrogate for other water quality variables that are correlated with nitrate-N. Furthermore, statistical relationships between nitrogen and biological communities in Ohio streams may be muted because nitrate-N concentrations at least impacted reference locations reflect highly enriched conditions when compared to other temperate North American streams and proposed trophic classifications (Dodd et al. 1998). Essentially nitrogen was usually present in concentrations saturating to algal growth, and therefore not limiting, especially given that elevated nitrate-N concentrations lag behind flow curves. Suggested total inorganic nitrogen criteria are given in Table 1.

- Biological community performance in headwaters and wadable streams was highest (*i.e.,* Index of Biotic Integrity [IBI] or Invertebrate Community Index [ICI] values 50-60) where TP concentrations in were lowest. Conversely, biological integrity was successively lower (*e.g.*, marginally good, fair, poor) with increasing TP concentrations. The association between increasing TP concentration and decreasing IBI or ICI scores was statistically significant. When TP was categorized by median concentrations, IBI scores associated with TP concentrations less than the median (0.17 and 0.12 mg/l in headwaters and wadeable streams, respectively; Miltner and Rankin 1998) were significantly higher than those associated with TP concentrations exceeding the median. The difference was most defined in wadeable streams where the mean of IBI scores associated with the lowest quartile of TP concentrations (<0.06 mg/l) was significantly higher than means from the other three quartiles (Miltner and Rankin 1998). The association between increasing TP concentration and decreasing ICI scores, though significant, was not as strong as that for the IBI. The lowest TP concentrations were also associated with the highest quality stream habitats (*i.e.,* Qualitative Habitat Evaluation Index [QHEI] scores >60-70). The correlation of low TP with high quality lotic habitat is thought to be the result of TP being sequestered by the well organized, diverse and trophically dynamic aquatic assemblages that are typically associated with high quality habitat. High quality habitat also results in lower downstream sediment delivery due respectively to the expulsion and filtering effects of better channel morphology and intact riparian buffers. See Table 2 below for suggested TP criteria.

- Habitat characteristics appeared to have some of the strongest effects on the aquatic biota and should be a major consideration in developing nonpoint source pollution abatement strategies where the objective is to restore and protect beneficial aquatic life uses. Sediment sensitive habitat features such as a lack of substrate and riffle embeddedness and a high degree of channel development (*i.e.*, riffle-pool-run sequences) and stability were positively correlated with IBI scores.

- Because habitat is a critical component stream function, *habitat data must be considered as an integral part of any attempt to restore aquatic life in a stream or river if such efforts are to succeed*. Implementation of Best Management Practices (BMPs) to reduce upland erosion without consideration of channel condition or other habitat limitations will not be sufficient to restore aquatic life uses such that WQS are attained, even though overall sediment and nutrient loadings may be reduced. Similarly, reductions in upland erosion rates may be insufficient if bank erosion and riparian interactions are not concurrently addressed. Habitat management efforts should focus on maintaining and restoring the riparian functions that are often lost when streams are channelized or riparian areas are otherwise encroached upon.

- Another consequence related to the importance of habitat is consideration of pollution control strategies in streams or watersheds where habitat has been irretrievably modified and attainment of the baseline Clean Water Act goals (*i.e.*, at least WWH) is precluded. In such situations habitat becomes the lowest common denominator and the controlling variable with respect to aquatic life use attainment, and can strongly influence water quality. Therefore, nutrient reduction strategies may be controlled by different criteria (*e.g.,* public water supply, phosphous in Lake Erie). However, given the importance of habitat in determining the availability of nutrients in the water column, meeting the goals of these non-aquatic life uses will likely be impeded without consideration of the critical role of riparian habitat in the context of the watershed unit.

- Where biological index scores do not exhibit a linear relationship along a gradient of nutrient concentration, they often display a threshold effect to high concentrations (espcially to nitrogen), and where a linear relationship does exist, exceptional biological communities and high nutrient concentrations may co-occur. Therefore, exceedences of the criteria listed in Tables 1 and 2 cannot be interpreted in a manner analogous to that commonly used for toxic substances. Because of this we suggest using a tiered or multicriteria approach, especially in light of the importance of habitat. In other words, a single exceedance should not necessarily trigger a violation of water quality standards. For the interpretation of chemistry results alone, how the central tendency or distribution of a series of samples compares to the central tendancy or other measure of the reference population is much more meaningful. Moreover, nutrient values should not be interpreted in a vacuum of biological information given that high values of both can co-occur. Instances where biological index scores meet the biological criteria but nutrient concentrations are high, implies that nutrients are not *locally* problematic. The question then becomes one of whether the nutrients are assimilated before causing a problem, and that argues for iterative sampling to address that possibility, and consideration of downstream uses. That question also argues for some measure of trophic state via the primary producers, either measures of chlorophyll *a* or composition of the periphyton community in repsonse to nutrient level, and monitoring for secondary effects of over-enrichment such as diel dissolved oxygen variations or the presence of cyanotoxins. As previously mentioned, measures of habitat quality and land-use information must be a tier of the criteria. For example, instream habitat may be good in an agricultural setting, yet extensive tile drainage can bypass the filtering or assimilative effect of the riparian zone. In short, the aspects of habitat and landuse that either facilitate or inhibit assimilation, or exacerbate impacts must be accounted for when assessing instream nutrient concentrations. Lastly, any approach must be iterative. That is, progress toward meeting nutrient goals for larger rivers and streams may first depend on meeting goals in upstream reaches (*i.e.*, headwaters). Restoration of headwaters may require a phased approach, and once restored, may change the reference condition of large rivers.

• Table 1. Median and seventy-fifth percentile nitrate+nitrite nitrogen concentrations by stream size and ecoregion for reference sites, oligo-mesotrophic and meso-eutrophic boundaries given by Dodd et al. (1998), and proposed statewide criteria for WWH, EWH and MWH streams. Values corresponding to the IBI range typical of the MWH use represent bets attainable attainable concentrations for MWH streams

			Ecoregional Criteria			Oligo-mesotrophic Meso-eutrophic		State-wide Criteria	
	HELP	IP	EOLP	WAP	ECBP	boundaries†	WWH	EWH	MWH*
Headwaters (drainage area < 20 mi^2)									
median	0.38	0.49	0.42	0.15	0.98	0.7			
75th %	2.26	1.18	1.00	0.34	2.24	1.5	1.0	0.5	1.0
20 - 29	1.22	3.15	0.56	0.21	0.86				
Wadable (drainage area \$ 20 mi^2 < 200 mi^2)									
median	0.16	0.24	0.43	0.22	0.84	0.7			
75th %	0.60	0.54	1.05	0.47	2.80	1.5	1.0	0.5	1.6
20 - 29	0.68	1.42	1.60	0.50	1.34				
Small Rivers (drainage area \$ 200 mi^2 < 1000 mi^2)									
median	1.88	0.43	1.00	0.64	1.65	0.7			
75th %	3.24	0.96	1.42	1.02	3.06	1.5	1.5	1.0	2.2
20 - 29	2.01	-	1.97	1.55	1.88				
Large Rivers (drainage area > 1000 mi^2)									
median	1.47	2.63	-	1.50	3.08	0.7			
75th %	2.76	2.93	-	2.20	4.14	1.5	2.0	1.5	2.4
20 - 29	1.73	-	-	2.60	3.98				

†Oligotrophic-mesotrophic and mesotrophic-eutrophic boundaries are given by Dodd et al. (1998) and were derived from data sets covering a wide range of stream sizes.

* MWH criteria are the statewide median concentrations from the ALL database for an IBI range of 20 - 29.

• Table 2. Median total phosphorus concentrations by IBI range (from the ALL data set), ANOVA results, and suggested criteria for the protection of aquatic life.

	Ecoregion Criteria						Statewide Criteria		
IBI Range[1]	HELP	IP	EOLP	WAP	ECBP	ALL[3]	WWH†	EWH†	MWH
Headwaters (drainage area < 20 mi^2)									
20 - 29	0.42	2.88	0.19	0.05	0.58	0.34			
40 - 49	-	0.13	0.05	0.05	0.07	0.06			
50 - 60	-	0.05	-	0.05	0.05	0.05			
ANOVA[2]	ns	ns	0.05	ns	0.05	0.05	0.08	0.05	0.34
Wadable (drainage area ≥ 20 mi^2 < 200 mi^2)									
20 - 29	0.33	0.50	0.25	0.07	0.22	0.28			
40 - 49	-	0.15	0.07	0.05	0.11	0.09			
50 - 60	-	0.07	0.05	0.05	0.08	0.07			
ANOVA	ns	ns	0.10	0.10	0.10	0.05; 0.10	0.10	0.05	0.28
Small Rivers (drainage area ≥ 200 mi^2 < 1000 mi^2)									
20 - 29	0.25	-	0.20	0.25	0.25	0.25			
40 - 49	-	0.33	0.12	0.08	0.16	0.18			
50 - 60	-	0.15	0.08	0.05	0.17	0.14			
ANOVA	ns	ns	0.10	0.10	ns	ns	0.17	0.10	0.25
Large Rivers (drainage area > 1000 mi^2)									
20 - 29	0.22	-	-	0.51	0.60	0.32			
40 - 49	-	0.35	-	0.18	0.41	0.34			
50 - 60	-	-	-	0.15	0.46	0.24			
ANOVA	ns	ns	-	ns	ns	ns	0.30	0.15*	0.32

[1]Median total phosphorus concentrations for the given IBI range are from Appendix Table 2.

[2]ANOVAs were run on three categories of total phosphorus concentrations, ≤ 0.05, 0.06 ≤ 0.10, and > 0.10, total phosphorus concentrations listed in ANOVA rows show concentrations where differences in IBI scores between categories were significant.

[3]For IBI ranges, ALL is the average of all ecoregions. Data were pooled acrossed ecoregions for the ALL ANOVAs, otherwise ANOVAs were stratified by ecoregion and drainage area.

† Values in the WWH and EWH columns represent suggested total phosphorus concentrations that are protective of aquatic life.

* TP concentration chosen to reflect N:P ratio ≥ 10.

[ns] ANOVAs for the stream size and ecoregion were not significant (P > 0.05)

INTRODUCTION

Nonpoint sources of pollution are among the most pervasive of impairments to aquatic life in Ohio (Ohio EPA 1994a) and include not only the introduction of pollutants from surface and subsurface runoff, but the physical manipulation of lotic ecosystems and watersheds. Direct and indirect effects of riparian and stream channel modifications on lotic ecosystems have been documented (Karr and Schlosser 1977, Karr *et al.* 1983, Rankin 1995). However, the deleterious effects on aquatic life from polluted runoff, especially the primary nutrients (nitrogen and phosphorus), and the interaction with habitat quality, is neither widely acknowledged nor generally understood by resource management and regulatory agencies. Only recently has the issue been addressed of how land use, physiographic relief, soil types, and lotic habitat *interact* to affect instream nutrient concentrations and, in turn, the quality of aquatic assemblages (Richards *et al.* 1996, Allan *et al.* 1997, Johnson *et al.* 1997). However this historic lack of understanding has been evident in the management of water resources in the U.S. (Karr 1995).

The objectives of this study, a subset of the Ohio EPA Load Allocation Project as a whole, are to: (1) document the background or "reference" concentration ranges of nutrient and other conventional parameters at regional reference sites under typical summer-fall low flow conditions; (2) document the effects of stream size, ecoregion, and habitat on the low flow concentrations of these parameters; (3) determine whether the relative performance of fish and macroinvertebrate community assemblages are correlated with nutrients and identify any significant covariates (*e.g.,* habitat variables); (4) develop analytical tools to better relate biological community performance data to water chemistry data; and, (5) suggest where, when, and under what circumstances the control of nutrients is most critical to the restoration and protection of Ohio's lotic ecosystems.

This study focuses largely on the effects and interactions of residual nutrient concentrations (RNC) and lotic habitat and how these are correlated with the relative health and well-being of resident aquatic communities as defined by the indices and metrics of the Ohio EPA biological criteria. Based on the background information presented here, a goal of this study is to develop a framework by which biocriteria and habitat information is routinely integrated into the load allocation and TMDL process. Reference statistics for chemical and physical parameters analyzed in this study (REF) are organized by ecoregion and stream and river size and are summarized in Appendix 1. Empirical relationships between expressions of biological community performance (*i.e.,* IBI and ICI) and key chemical and physical parameters (ALL) are summarized in Appendix 2.

Background

Effects of Nutrients on Aquatic Life

Nutrients, except under unusual circumstances, rarely approach concentrations in the ambient environment that are toxic to aquatic life. U.S. EPA (1976) concluded that “levels of nitrate nitrogen at or below 90 mg/l would not have [direct] adverse effects on warmwater fish." However, nutrients, while essential to the functioning of healthy aquatic ecosystems, can exert negative effects at much lower concentrations by altering trophic dynamics, increasing algal and macrophyte production (Sharpely *et al.* 1994), increasing turbidity (via increased phytoplanktonic algal production), decreasing average dissolved oxygen (D.O.) concentrations, and increasing fluctuations in diel D.O. and pH. Such changes are caused by excessive nutrient concentrations resulting in shifts in species composition away from functional assemblages of intolerant species, benthic insectivores and top carnivores (*e.g.,* darters, insectivorous minnows, redhorse, sunfish, and black basses) typical of high quality warmwater streams towards less desirable assemblages of tolerant species, niche generalists, omnivores, and detritivores (*e.g.,* creek chub, bluntnose minnow, white sucker, carp, green sunfish) typical of degraded warmwater streams (see Figures 3 and 4).

Scope and Purpose of This Study

Considerable efforts have been undertaken in Ohio and elsewhere to determine the significance and magnitude of *flow-dependent* nonpoint source nutrient loadings to Lake Erie (Baker 1985, 1988) and other large water bodies. Hence, much of the emphasis of reducing nutrient delivery to lotic systems via nonpoint sources has been driven by management objectives aimed primarily at reducing negative effects (*i.e.,* eutrophication) in lakes, bays, and estuaries (*e.g.*, Rohlich and O'Connor 1980 for the Great Lakes). In this situation the primary concerns are with the "far-field" effects of nutrient exports, specifically loadings that contribute to eutrophication. These impacts are generally controlled by the loadings that are delivered by short-term runoff events, and nonpoint source management initiatives have emphasized the reduction of nutrients in runoff. However, the residual impacts within the local lotic ecosystem (*i.e.,* "near-field" effects) are the combined result of the assimilative capacity of the stream or river and the residual concentrations and ecological effects that are "left behind" by the accumulation of these events. *In this context assimilative capacity encompasses more than the dilution dynamics of the receiving stream, but include all factors (i.e., habitat and biota) that affect this capacity.* The near-field effects of nutrients on the aquatic biota (including both flow-event inputs and low flow residual concentrations) resident to these lotic ecosystems are most evident in the measures of aquatic community performance (*e.g.,* IBI, ICI, and associated metrics and variables) which exhibit both long and short-term responses to stressors such as excessive nutrients. Relating RNCs (*i.e.,* the concentrations existing during normal summer-fall low flow periods) to measures of aquatic community performance reduces uncertainty regarding duration and exposure concerns that are common to mass balance modeling approaches. This is due to the ability of the aquatic biota to integrate the cumulative effects of multiple events and their comparative stability through time, and thus provide measurable endpoints for water quality goals.

This study focuses primarily on the near-field response of aquatic assemblages in streams and rivers to residual concentrations of nutrients measured during normal, summer-fall low flow periods, and how habitat can mediate assimilation and help to ameliorate nutrient effects. The aquatic biota of lotic ecosystems is frequently subjected to various short-term events including flow extremes, turbidity, sediment and increased levels of nutrients and other conventional substances during runoff events. Excepting catastrophic toxicity or extreme physical effects, individual short-term events have comparatively little influence on the character and makeup of an aquatic community. The cummulative effects of these events coupled with the more frequently occurring conditions present under normal summer-fall seasonal flows are much more critical to the eventual "product," or composition, structure and function of the aquatic community. The end product is essentially the biological integrity goal of the Clean Water Act (Section 101[a][2]) and is quantified by the measures comprising the biological criteria.

The available scientific information about nutrient spiraling in lotic ecosystems indicates that headwater streams strongly influence the elemental dynamics of higher order streams and rivers within a watershed through the cumulative cascading of near-field effects in a downstream direction. Thus, unlike studies that attempt to measure the total loading of nutrients and sediment in runoff and subsequent downstream export via high flow events, we are primarily concerned here with the cumulative consequences of "what is left behind" by runoff events and the subsequent cascading of ecological effects throughout a watershed. These consequences are also *de facto* measures of the effects of adjacent landuse practices. Thus, much as the capillaries of the human circulatory system are critical to the eventual functioning of the veins and arteries, the network of headwater streams within a watershed are critical to the functioning and quality of services provided by the larger order streams and mainstem rivers. That three-quarters of all streams in the U.S. are first or second order (Leopold *et al.* 1964) underscores the importance of the land-water interface and the function of headwaters in maintaining watershed integrity.

High-Flow vs. Low-Flow Measures of Nutrients

The concentration of nutrients (as a logarithmic function) in lotic ecosystems increases significantly with increased flow (Edwards 1973; Brooker and Johnson 1984 *c.f.* Lowrance and Leonard 1988). However, a precise predictive relationship does not exist because similar concentrations can occur at different flows (Lowrance and Leonard 1988). For example, a two-inch rainfall immediately following fertilizer application will likely result in different instream nutrient concentrations than the same amount of rain at the end of the growing season, although concentrations during each event will likely be elevated compared to the more frequently occurring low-flow concentrations. In addition, the assimilation and removal of nutrients by an intact and healthy riparian buffer (*i.e.,* composed of mature, woody vegetation) is significant (Fennesy and Cronk 1997; Lowrance *et al.* 1984; Peterjohn and Correll 1984), although the magnitude of nutrient uptake varies seasonally (Meyer *et al.* 1988). Other studies have demonstrated that tillage practices in an agricultural watershed (*i.e.,* conventional vs. no-till) can have substantial effects on the rate of nutrient delivery to streams (Chichester and Richardson 1992).

Flow weighted sampling of chemical constituents is required to accurately estimate total loadings of nutrients for the calculation of Total Maximum Daily Loads (TMDLs). Large runoff events, which deliver a high proportion of the annual loading of nutrients in a short time period (Baker 1985), are known to affect water quality in downstream (far-field), estuarine or lentic environments. However, direct evidence of negative, near-field effects of *elevated* concentrations of nutrients during these short-term events on resident aquatic assemblages is lacking. Given the low acute toxicity of elevated nutrients during such short-term events, it is the residual effects (*e.g.,* the elemental flood subsidies of Meyer *et al.* 1988) of nutrient loadings that are likely of most consequence to aquatic community performance. The cumulative effects of these events on trophic and energy dynamics of lotic ecosystems may be long lasting.

The retention of nutrients in a stream reach and nutrient fluxes are important in determining how nutrients affect aquatic assemblages. Lotic reaches that either export or assimilate nutrients into desired biomass quickly (*e.g.,* streams with high quality habitat and high gradient) may be less impacted by short-term loadings of nutrients. Meyer *et al.* (1988), in a review of elemental dynamics in streams, summarized a range of possible effects of runoff events:

> "In one sense, (nutrient) concentration at baseflow is what supports production, and hence

> slight seasonal changes in concentration may have a major effect. On the other hand, storms may serve as an element subsidy by mobilizing previously unavailable nutrient sources.... The distinction between a disturbance and a subsidy is unclear. Increased flows of low magnitude, but short recurrence interval may provide elemental inputs that subsidize the community between major events. Larger storms with longer recurrence intervals may be disturbances because they scour the community and remove much of the nutrient capital associated with benthic organic matter. Are there thresholds for spates above which the short- and long-term system productivity is decreased, but below which it is enhanced?"

These questions are important when examining differences among similar types of streams, but become even more important when watersheds have been exposed to extensive anthropogenic disturbances. Further research is needed to further quantify some of these relationships.

The Role of Physical Habitat

Physical habitat quality is a critical factor in determining if the effects of RNC will be mitigated or exacerbated. High quality lotic habitats with intact riparian zones and natural channel morphology may ameliorate the potentially adverse effects of RNC by assimilating excess nutrients directly into plant biomass (*e.g.*, trees and macrophytes), by sequestering nutrients into invertebrate and vertebrate biomass, by "deflecting" nutrients into the immediate riparian zone during overland (flood) flow events (see reviews by Malanson 1993; Barling and Moore 1994), and by reducing sunlight (a principal limiting factor in algal production) through shading. Also, high quality habitats minimize nutrient retention time in the water column during *low flows* because they tend to have high flow velocities in narrow low flow channels (*e.g.*, unbraided vs. braided riffles), and coarse substrates with little potential for adsorption. Additionally, a healthy community of aquatic organisms typical of high quality habitats process and utilize nutrients very efficiently (see *The Phosphorus Cycle in Lotic Ecosystems* and *Processing of Nutrients in Natural vs. Modified Lotic Ecosystems* below).

Conversely, degraded, poor quality lotic habitat with reduced or debilitated riparian zones (in terms of width and function) and simplified channel morphology generally exacerbate the deleterious effects of RNC by reducing the riparian uptake and conversion of nutrients, by increased retention time through increased sediment-water column interface via a wide channel and subsequent loss of low flow energy (*e.g.*, increased intermittency), retention of nutrients within the channel due to diminished filtering time during overland flow events, and by allowing full sunlight to stimulate nuisance growths of algae. These factors also interact to increase the retention of RNC in the most available dissolved forms, attached to fine sediments (especially clays and silts) and in planktonic and attached algae. Low gradient streams, because of longer nutrient and sediment retention times, are more susceptible to the effects of nutrients than otherwise similar high gradient streams. For example, "fresh" sediments in high gradient streams may create a buffer from high phosphorus concentrations by providing adsorption sites for phosphorus (Klotz 1988). In contrast, in low gradient streams with high sediment retention, such adsorption sites may be secured by existing phosphorus and the sediment will have little effect on ameliorating RNC enrichment.

Active connections between the riparian and floodplain habitats and the stream channel are also important in nutrient and sediment dynamics. Naiman *et al.* (1988) and Malanson (1993), in a review of the functions of riparian habitats, discussed the need to broaden the application of river continuum theories to include the effects of nutrient cycling both laterally within riparian areas, as well as longitudinally in a downstream direction, and to include the role of such habitats as sinks and sources of nutrients in lotic ecosystem management. These connections are reduced or altogether eliminated in streams that have been altered for flood control or agricultural drainage purposes. Therefore, these concepts need to be fully incorporated into the design and implementation of nonpoint source managment strategies.

Nutrient Cycling in Lotic Ecosystems

The processing of nutrients in lotic ecosystems is complex, variable, and affected by abiotic factors such as flow, gradient, groundwater quality and quantity, and channel morphology. In an attempt to illustrate the basics of this process, a brief summary of nitrogen and phosphorus cycling in lotic ecosystems follows. Understanding this is essential to comprehending the range of ecological effects resulting from anthropogenic impacts on these processes, and to effectively intervene in an attempt to minimize the negative effects. The major theoretical framework for these processes is the River Continuum Concept (Vannote *et al.* 1980; Minshall *et al.* 1983; Minshall *et al.* 1985; with recent work in this area summarized by Minshall 1988 and Malanson 1993) which emphasizes the importance of headwater streams to ecological function in the higher order downstream reaches. Natural, unmodified headwater streams generally *retain* and *process* coarse particulate organic matter (CPOM) and woody materials and *export* processed fine particulate organic matter (FPOM) to downstream reaches (Wallace and Ross 1982). In the eastern U.S., the nutrients contained in leaves from deciduous tree species and other components of CPOM, are processed through the lotic ecosystem in *many* different steps. The end product is FPOM *and* high quality biomass in the form of a high diversity of aquatic invertebrates and vertebrates representing balanced trophic relationships. As nearly 78% of Ohio's lotic surface waters are headwater streams (<20 sq. mi.), the aggregate importance of these waters to downstream uses is clear; they supply high quality biomass and nutrients in a form favoring high biological integrity in downstream reaches. In short, headwaters represent the primary interface between watersheds and human land uses, and are the initial entry points for energy and nutrients into lotic ecosystems. The form, manner, and rate at which nutrients are delivered to headwaters and eventually transported downstream profoundly affects the ecological integrity of the larger streams and rivers that harbor many of the direct use benefits for humans.

The Nitrogen Cycle in Lotic Ecosystems

Figures 1 and 2 summarize the cycling of nitrogen and phosphorus in lotic ecosystems typical of the Midwestern U.S. The following summary of the nitrogen cycle in streams is from Newberry (1992):

> "Inputs of nitrogen are from precipitation, atmospheric nitrogen fixation, groundwater, and surface water flow. Plant uptake provides temporary nitrogen storage. Outputs of nitrogen are the release of nitrogen gases, N_2O and N_2, to the atmosphere by denitrification; to streamwater; to groundwater; and removal by forestry and other harvesting operations.
>
> Nitrogen is stored primarily in three forms. The first two, nitrate (and nitrite) and ammonium, are inorganic and are available as plant nutrients. The third form is organic nitrogen, contained in live and decaying plant and animal material, and in microbial biomass. Organic

> nitrogen composes the bulk of nitrogen in the soil (Bowden 1987) and is not readily available as a plant nutrient. Through the microbially-mediated processes of mineralization and nitrification, however, quantities of organic nitrogen and ammonium are transformed into nitrate (Brady 1990). Nitrifying bacteria are distinct from denitrifying bacteria.... Because the ammonium ion, NH4+ is positive, it binds readily to the soil which has an overall negative charge. Ammonium is thus not as mobile as the negative nitrate ion, NO3-, that does not bind with the soil. Nitrate is very mobile and travels readily to groundwater."

Although data for ammonia-N and nitrite-N are summarized in the appendices, their effects are most frequently associated with point sources of (*e.g.,* WWTPs, livestock, or acute fertilizer impacts). This study focuses on nitrate and phosphorus; the negative enrichment effects being potentially widespread and related to erosion and diffuse runoff from urban and agricultural sources. The column labeled nutrient regulation in Figure 1 summarizes the potential points of intervention where nitrogen loadings could be affected by best management practices (BMPs).

Nitrogen Sources	Nitrogen Input Forms	Nitrogen Transformations	Nitrogen Regulation	Nitrogen Outputs
Precipitation	Nitrate	Denitrification	Fertilizer Application	Nitrogen Gases to Atmosphere
Groundwater	Nitrite		Loadings	Groundwater
Point Sources	Gaseous	Mineralization & Nitrification	Riparian Uptake	Surface Water
Runoff	Ammonia		Groundwater	Vegetation
Natural Decay (organic)	Organic		Overland Flow	Animal Biomass
			Stream Morphology	

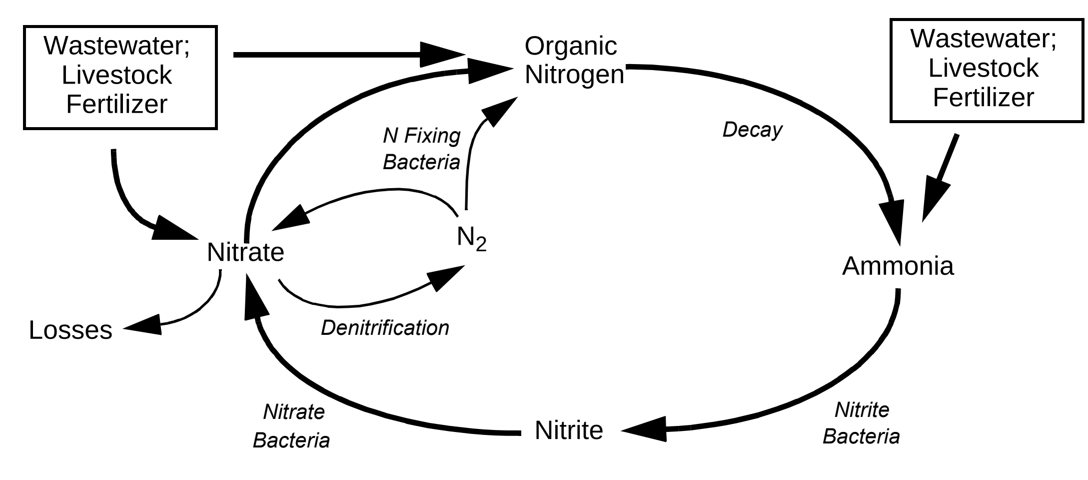


Figure 1. Summary diagram of the Nitrogen Cycle in streams and rivers outlining the key characteristics of nitrogen delivery, processing, and regulation in lotic systems. Modified from Odum (1971).

The Phosphorus Cycle in Lotic Ecosystems

The cycling of phosphorus within and between streams and the nearby land is summarized in Figure 2. In streams and rivers of the eastern U.S., phosphorus can be a limiting factor in algal and macrophyte growth, and has been observed with greater frequency than nitrogen limitation (Newbold *et al.* 1983; Sharpely *et al.* 1994). The dynamics of nutrient limitation in lotic environments is not as straight foward as that for lentic environments. Unlike pelagic lentic environments where phosphorus is often bound and tightly cycled within the biota, lotic environments are open and therefore continually receive phosphorus from upstream, groundwater, or runoff. Current also helps reduce limitation by reducing diffusion barriers. Finally, the interface between sediments, where inorganic phosphorus is frequently adsorbed, and water is obviously more immediate in lentic environments. Under natural conditions much of the phosphorus delivered to streams is bound in organic forms (*e.g.,* in leaves, woody debris, invertebrates, etc.) and is then transferred between and among the different trophic levels within the lotic ecosystem. The role of macroinvertebrates in this transformation process is very important:"....invertebrates may act as temporal mediators; their feeding activities result in a more constant supply of detritus to downstream communities by reducing the buildup of benthic detritus below levels subject to episodic transport during spates" (Ward 1989).

When anthropogenic sources of phosphorus are delivered to a stream the ratio of dissolved phosphorus immediately available to algae may be high relative to particulate forms of phosphorus (*e.g.,* attached to soil particles; Robinson *et al.* 1992). Total phosphorus (TP; the form measured in this study) consists of both dissolved phosphorus (DP), which is mostly orthophosphate, and particulate phosphorus, including both inorganic and organic forms, (PP; Sharpely *et al.* 1994). Runoff from conventional tillage is generally dominated by PP; however, the proportion of TP as DP increases where erosion is comparatively low such as with no-till fields or pasture (Sharpely *et al.* 1994). Streams with low gradients and a morphology that enhances deposition of sediments in the low flow channel (*e.g.,* channelized streams) may continually release dissolved phosphorus from sediments. In lakes with P-enriched sediments this process can result in eutrophication problems even after P reductions in the watershed have been acheived (Sharpely *et al.* 1994). Certain fish species (*e.g.*, gizzard shad in lakes, bluntnose minnow in streams) can actually "pump" nutrients from the sediments through feeding and excretion and affect nutrient cycling (Stein et al. 1995). Nutrient recycling occurs during downstream transport (Newbold *et al.* 1983) and is termed "nutrient spiraling."

Phophorus Sources	Phophorus Input Forms	Phophorus Transformations	Phophorus Regulation	Phophorus Sinks
Precipitation	Orthophosphate (Inorganic Phosphorus)	Phosphatizing Bacteria	Light	Groundwater
Groundwater			Fertilizers	Surface Water
Point Sources	Particulate Phosphorus (Attached Inorganic)	Anaerobic reduction of ferric hydroxides and release of inorganic P	Point Sources	Vegetation (Riparian & Aquatic Macrophytes)
Runoff			Riparian Uptake	
Natural Decay (organic)	Organic Phosphorus		Groundwater	Animal Biomass (Bones & Teeth)
			Overland Flow	
			Stream Morphology	

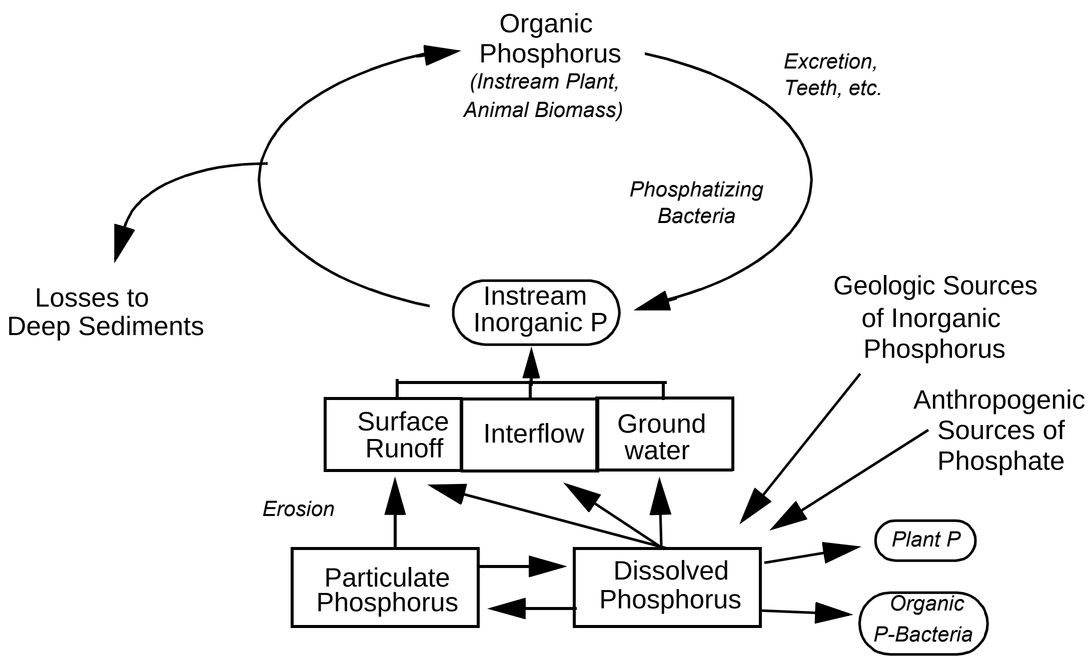


Figure 2. Summary diagram of the Phosphorus Cycle in streams and rivers outlining the key characteristics of phosphorus delivery, processing, and regulation in lotic ecosystems. Modified from Novotny and Chesters (1981).

Processing of Nutrients in Natural vs. Modified Lotic Ecosystems

In lotic ecosystems the distribution and cycling of nitrate and phosphorus among trophic groups, and between particulate organic matter or sediments and the biota, is affected by channel morphology and habitat features (*e.g.*, stable woody debris) that trap organic debris and impede its downstream movement during high flow events (Minshall *et al.* 1983, Raikow *et al.* 1995, see Figure 3). In natural streams organic debris dams aid transport of organic matter to the floodplain and foster the transformation of nutrients into *desirable* biomass. Working concomitantly with habitat, a diverse and high quality biological assemblage sequesters nutrients by processing and partioning them between a variety of species and trophic levels, and thereby acts to mute episodic downstream transport. In contrast, much of the dissolved and inorganic forms of nutrients delivered to, and cycled within, degraded stream ecosystems are readily available to algae, thus fostering a rapid transformation into *undesirable* biomass. Aquatic organisms can greatly affect the form and rate of export of organic matter and nutrients from headwaters to downstream reaches. For example differences between FPOM export in streams having intact macroinvertabrate populations and those where macroinvertebrates were experimentally removed were greater than the differences in FPOM export experienced between 50-year high and low flow events in a typical stream (Wallace *et al.* 1991). Increased algal biomass can result in wide fluctuations in D.O. (especially in open channels where full sunlight is available) and can disrupt and circumvent orderly pathways of energy flow through a lotic ecosystem. Such modifications in energy dynamics and energy flow can have significant effects on species composition in streams and rivers, favoring less desirable and tolerant species.

Figures 3 and 4 summarize and compare the predominant pathways and forms of nutrient export in headwater streams under natural (Fig. 3) and modified (Fig. 4) channel morphologies. The sizes of the arrows indicate the relative amount of nutrients exported to downstream reaches or between the adjacent riparian zone and the stream. Calculation of background expectations of nutrient parameters requires consideration of how nutrients spiral through lotic ecosystems and the mechanisms of nutrient processing performed by the biological components (Vannote *et al.* 1980; Minshall *et al.* 1983; Minshall *et al.* 1985). Under relatively unimpacted background conditions (Figure 3), headwater streams are generally heterotrophic systems, that is organic carbon production is supplied from outside the stream channel (*i.e.*, the riparian vegetation). In these streams the biological organisms feed on and process organic forms of nutrients (nitrogen, phosphorus) and physical and biological turnover is slow making the nutrient spirals longer (Minshall *et al.* 1983). In other words, there are many steps required to process raw organic matter into readily available inorganic nutrients (*e.g.,* phosphorus).

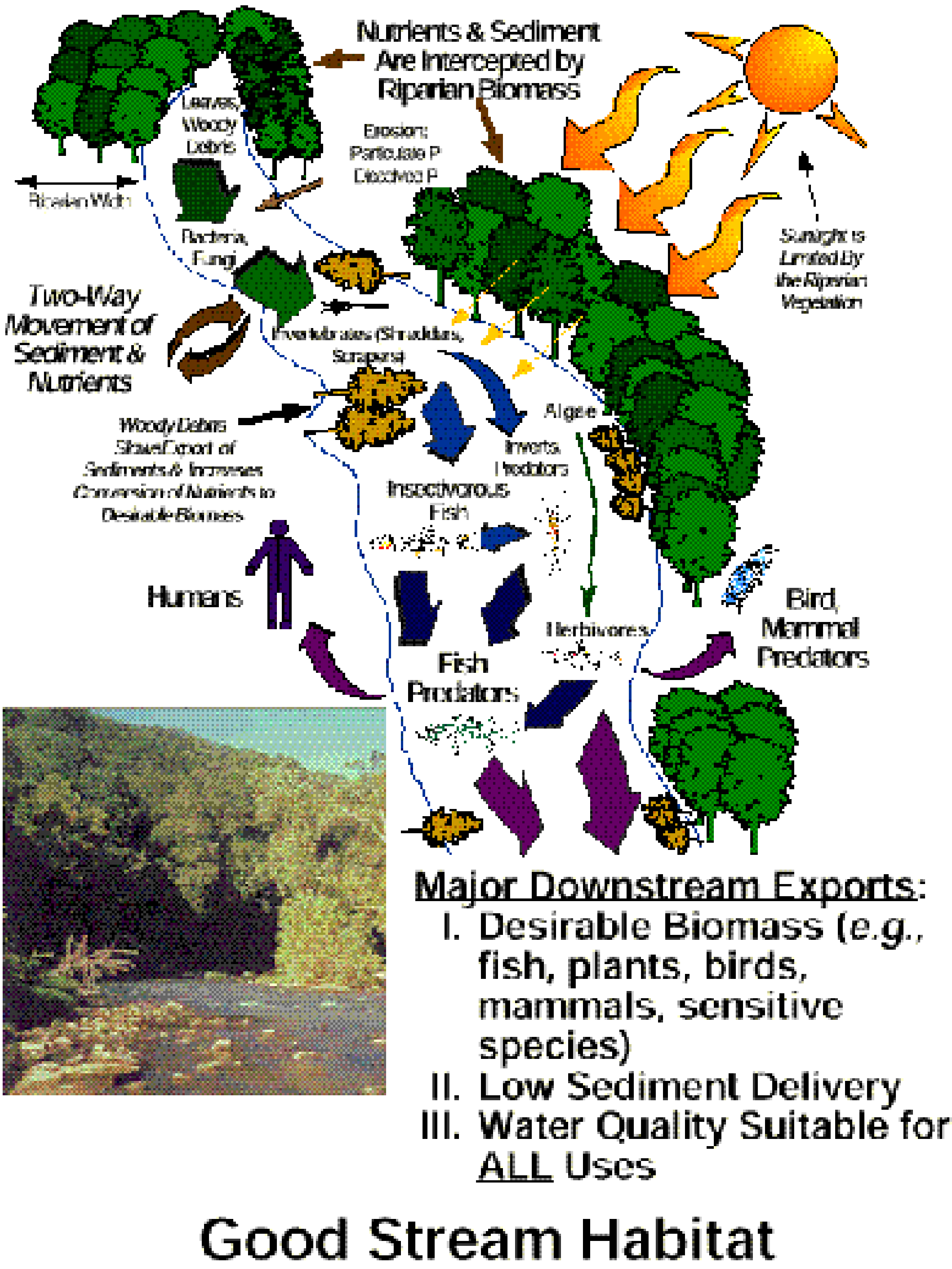


Figure 3. Illustration of nutrient and energy flow in a stream with natural and functional riparian structure.

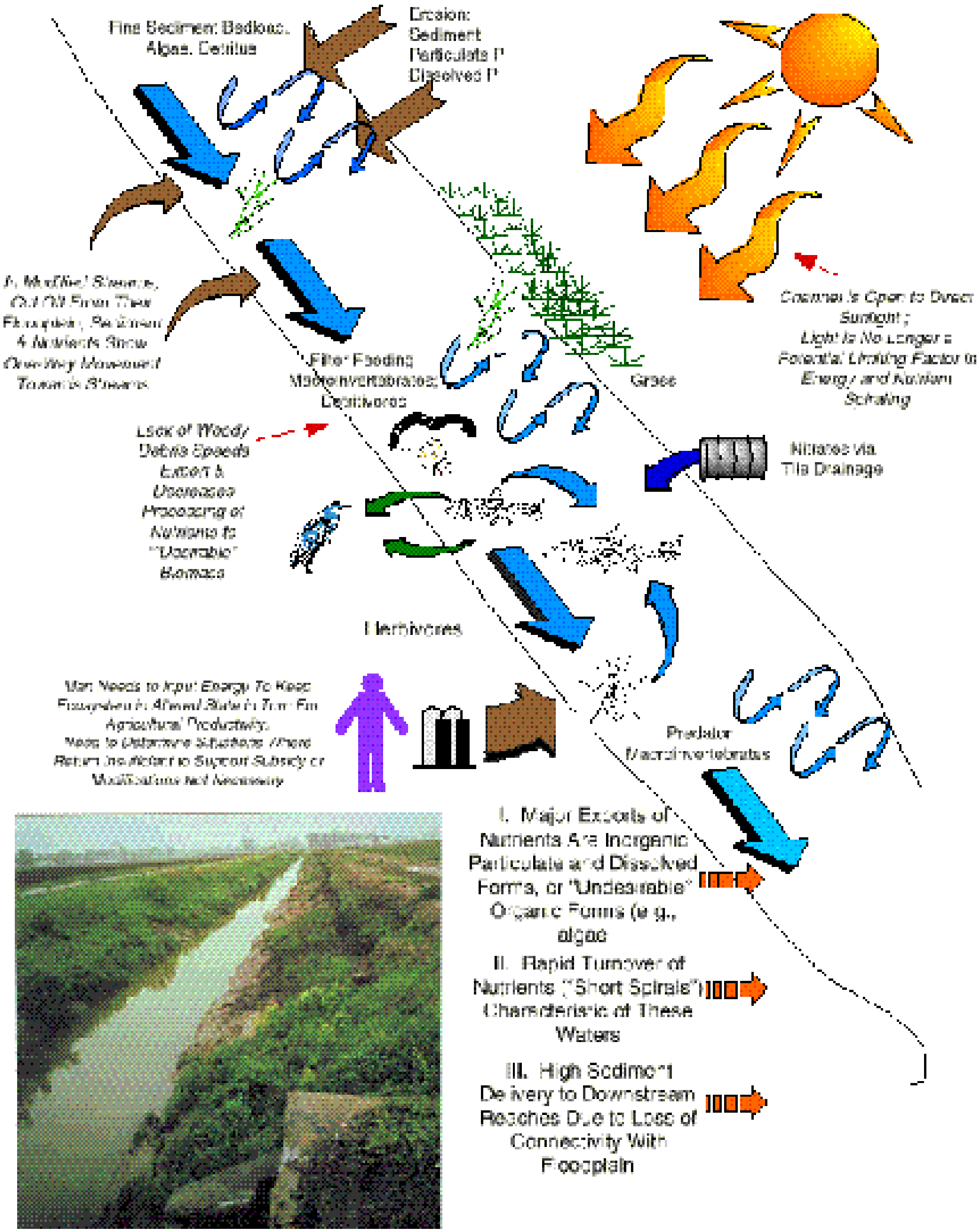


Figure 4. Illustration of nutrient and energy flow in a stream with extensively modified stream habitat and poor riparian structure.

Large rivers are more autotrophic with an increacing fraction of the organic carbon being fixed by primary producers within the streamchannel with increasing stream order. In these waters nutrient turnover is rapid (*i.e.*, short spirals), resulting in higher concentrations of readily available forms of nutrients. In Ohio a relationship with stream size was most evident in the Western Allegheny Plateau ecoregion where anthropogenic sources of nutrients are generally the lowest statewide (Figure 5). In headwater streams that have been either channelized, had riparian vegetation removed, or the habitat otherwise degraded, the nutrient processing mimics that of large rivers in having comparatively short spirals (rapid turnover) and high algal biomass (Figure 4). Modified streams usually support large populations of omnivores and detritivores (see Figure 4) which have been shown to further increase nutrient recycling in streams (Grimm 1988). Retention of inorganic nutrients is exacerbated further in streams with low gradients where a combination of excess sunlight, readily available nutrients, and slow flow velocity and volume result in degraded aquatic communities predominated by undesirable and highly tolerant species.

Stream gradient has been correlated with mean particle size such that streams with higher gradients have larger average diameter substrate particles than streams with low gradients (Leopold *et al.* 1964; Morisawa 1968). Because phosphorus is delivered to streams attached to fine particles (*i.e.*, particulate phosphorus or PP), streams with a high bedload of fine sediment also have the highest RNC of TP. The retention time for water and fine particles within the low flow channel of low gradient streams is longer than for higher gradient streams, resulting in an accumulation of TP. This is especially true for modified stream channels as was previously illustrated in Figure 4. This provides more time for the available phosphorus to be utilized in potentially undesirable ways such as the production of excess algal biomass, thus promoting tolerant and omnivorious organisms and circumventing assimilation among multiple species and trophic levels.

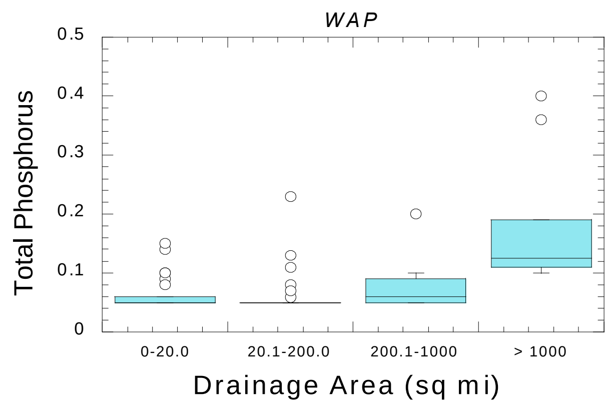


Figure 5. Total phosphorus (mg/l) by stream size in the Western Allegheny Plateau ecoregion measured at reference sites. Boxes enclose the 25th and 75th percentiles, outliers deviate more than two times the interquartile range from the median.

Malanson (1993) reviewed the literature regarding the role of riparian vegetation on the processes of nutrient spiraling in lotic ecosystems, especially the role of geomorphology in determining how these areas act as sources and sinks of nutrients. He argues for a broadening of the perspective of riparian habitats to include consideration of direct and indirect effects on water quality. This includes considerations of instream habitat quality which is a direct result of fluvial geomorphology and the quality and condition of the riparian zone. This concept was illustrated in Figure 3 and considers the bidirectional nature of nutrient movement into a stream from riparian areas and *away from the stream* and into riparian areas.

METHODS

Water Chemistry Data

The water column chemistry data used in our analyses were based on grab samples collected by Ohio EPA and was limited to data collected after 1980 to eliminate potential problems which might arise from the different detection limits that existed prior to that time. Only data from the summer-fall period defined as June 15 to October 15 was used to match the scope and objectives of this study. Values below detection limits were assigned one-half of the value when calculating statistics that are substantially influenced by below detection data values (*e.g.,* means, standard deviations). The detection limits of the parameters used in this analysis appear in Table 3. Field and laboratory QA/QC procedures for these and other parameters are detailed in the Ohio EPA Manual of Surveillance Methods and Quality Assurance Practices (Ohio EPA 1991a). The reference database (REF, see below) was edited to exclude data collected under high flows as noted by the field personnel collecting the sample or as determined from USGS gaging station data.

Table 3. Dectection limits of parameters (with acronyms) used in the analysis described in this report.

Parameter	Detection Limit
Ammonia (NH3-N)	0.05 mg/l
Nitrite (NO2-N)	0.02 mg/l
Total Kneldahl Nitrogen (TKN)	0.20 mg/l
Nitrate+Nitrite (NO3-N)	0.10 mg/l
Total Phosphorus (TP)	0.05 mg/l
Total Suspended Solids (TSS)	5.00 mg/l

Extremely high total suspended solids (TSS) values (>200 mg/l) or high total iron levels (>10,000 µg/l) were also used as an indicator of high flows (Figure 6) and field sheets from such samples were scanned for notes regarding elevated flow conditions. Any reference site (REF) samples noted as collected under elevated flows were eliminated from the database. Figure 7 is a histogram of total suspended solids (TSS) for individual high flow samples compared to means of low-flow samples. Although the average high flow TSS is one order of magnitude higher than the grand mean of the low flow data (355 vs 35 mg/l), the relationship is not entirely predictive (see correlation inset in Figure 6). The variability may be related to some reasons outlined earlier (*e.g.*, wide riparian zones may reduce solids runoff) or it may be an artifact of the high flow data collection not being collected along any specific point on the high flow curve. In any case, this database is insufficient to determine whether such a correlation exists. Future work needs to more carefully document the mechanism(s) of high flow nutrient associated impacts on the aquatic biota.

Biological Data

Fish or macroinvertebrate community data from 492 of the 1226 chemical REF sites (473 with fish data, 221 with macroinvertebrate data) were used in this study. All fish and macroinvertebrate data were collected with standardized methods as described by Ohio EPA (1989a). Multimetric biological indices including the Index of Biotic Integrity (IBI) for the fish assemblage and the Invertebrate Community Index (ICI) for the macroinvertebrate assemblage (Ohio EPA 1987a) were used in the analyses. The advantages of using such indices as indicators of water resource integrity and their ability to integrate and portray the effects of nonpoint sources (including hydromodification) are discussed by Karr (1991) and for application in Ohio by Ohio EPA (1987a), Yoder and Rankin (1995a,b), and DeShon (1995).

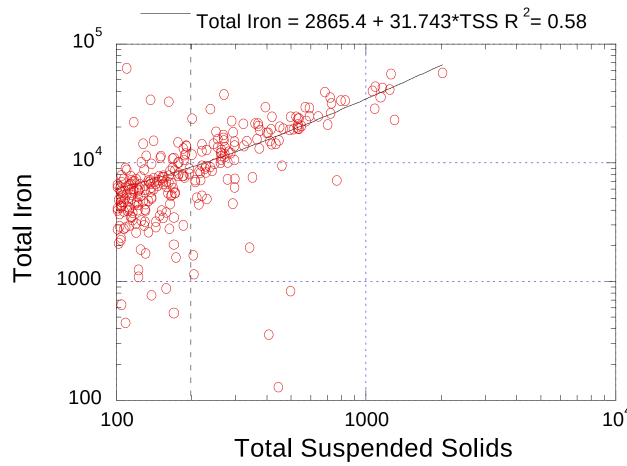


Figure 6. Total suspended solids (mg/l) vs total iron (ug/l) at reference sites. High values of these parameters were used as possible indicators of high flow samples.

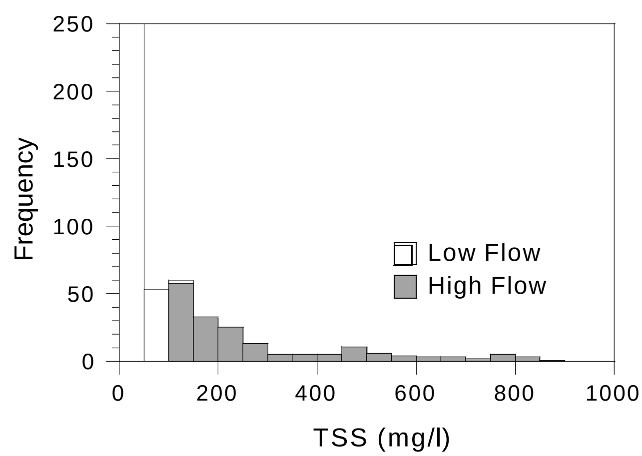


Figure 7. Frequency histogram of total suspended solids (TSS) values from high flow samples and average values from low flow samples from refernce sites in Ohio streams.

Reference Sites

Data from reference sites (REF) were used to define background expectations for the water chemistry parameters. REF sites encompass both the biological reference sites, which were used to generate the numerical biocriteria (BioC), and background water quality (BWQ) sites which were originally established to develop background (*i.e.,* upstream) chemical concentrations for use in the Ohio EPA wasteload allocation process. The BioC sites consisted of two subsets of reference sites: 1) least impacted sites intended to demonstrate the best attainable biological performance indicative of the Warmwater Habitat (WWH) and Exceptional Warmwater Habitat (EWH) use designations, and 2) physically modified sites (channelized, impounded, or non-acidic mine impacted) intended to establish the attainable biological performance expected for the Modified Warmwater Habitat (MWH) use designation. The BioC sites were selected throughout Ohio respective of stream and river size and ecoregion (Ohio EPA 1987).

Table 4. Codes assigned to the Load Allocation Reference Database (REF) based on types of impacts. Sites with X codes had been BWQ reference sites but were deleted from analyses becuase of point source related impacts or because they were located in a Lake or Reservior.

R	Reference, Least Impacted
M	Reference, Physically Modified
H	Suspected Physical Modifications
I	Impounded
A	Potential Mine Impacts
C	Urban Impacts
N	Potential Septic System Impacts
W	Wetland Influenced

XP - Excluded due to Point Sources
XR - Ecluded due to Lake or Reservior
XT - Excluded, Toxic Sediments
XD - Excluded, Landfill Impacts

The original BWQ sites were not necessarily selected following the criteria used to select the BioC sites, but were intended to reflect "upstream" water quality. The original intended purpose of the background water quality database was to provide an estimate of upstream water quality for conducting wasteload allocations for point sources. The BWQ sites were selected primarily based on being located upstream from point sources and were spatially stratified by hydrologic units. One result of this selection process is a strong bias toward small drainage areas for BWQ sites (Figure 8). Nearly 60% of the sites were from streams with a drainage area of 20 sq. mi. or less (BioC headwater site type) and less than 10% had a drainage area greater than or equal to 200 sq. mi (BioC wading site type). After the BioC and BWQ databases were combined each BWQ site was categorized as a true least impacted reference site or as impacted by habitat or nonpoint sources (similar to the modified BioC reference sites; see Table 4). The distribution of sites within these categories is summarized in Table 5. Preliminary analyses indicated that the site category was important for certain parameters (Figure 9). Therefore some analyses were limited to the least impacted reference sites database (code "R" in Table 4).

Because background nutrient concentrations vary with stream size, reference expectations were defined within the following stream and river size categories: headwater streams, 0-20 sq. mi.; wadeable streams, 20.1 - 200 sq. mi.; small rivers, 200.1 to 1000 sq. mi.; and large rivers, greater than 1000 sq. mi. These categories were selected based on: 1) observed patterns of nutrients with stream size, 2) theoretical expectations about nutrient concentrations in streams and rivers, and 3) observed changes in the biological communities with stream size. The entire REF water chemistry database consisted of more than 7500 grab samples (mean samples/site/year = 3.7).

Intensive Survey Sites

Besides the REF database, data from all other Ohio sites (ALL) collected after 1980 and limited to the June 15 through October 15 period where *both* water chemistry and biological data were available were also included in this study. The number of data points available at the ALL sites-varied by parameter because not all parameters were analyzed at each site or in each sample. The ALL database consisted of nearly 20,000 grab samples (mean samples/site/season = 7.9). This represents sites where biological data was also available for the same year. Much of the chemical data in the ALL database was collected at biosurvey or fixed monitoring sites and under low to normal summer flow conditions. However, a few samples were collected at higher flows and reflect the influence of runoff events. Frequency analyses (*e.g.,* box-and-whisker plots) were used to analyze the ALL database to reduce or eliminate the influence of the small proportion of any high flow samples on the conclusions. The ALL database (besides the REF database) provided the necessary resolution (these include both poor water quality and high water quality conditions) to determine the range of nutrient concentrations associated negative effects in the aquatic community as measured by the IBI and ICI

.

Analytical Tools

Because the design of this study is both exploratory and hypotheses testing, exploratory tools such as box-and-whisker plots and other visual methods (*e.g.,* scatter plots, gradient maps of nutrient concentrations) and multivariate techniques (*e.g.,* Principal Components Analysis) were used to visualize regional patterns in nutrient concentration and relationships with biological performance indicators. Environmental data frequently exhibit a "wedge" distribution of data points between two parameters, with the upper-edge representing a threshold beyond which cooccurrance of the two parameters is unlikely. For example, plots of species richness versus stream size or drainage area exhibit this pattern (Karr 1981, Fausch *et al.* 1984). Terrell *et al.* (1996) examined similar wedge-shaped patterns of variation in habitat and fish standing stock relationships. A line fit by eye through the upper 5% of these points along the angle of the upper surface of the wedge represents the maximum number of species expected for a given stream size. Lines drawn through the upper 5% of plots of a biological index versus the concentration of a water chemistry parameter is similarly interpreted as the maximum biological index values normally expected to coincide with a given chemical concentration. If a chemical parameter exceeds such a value, there is a strong likelihood the aquatic community would be unable to achieve that level of performance (*i.e.*, at least 95% of all observed index values were associated with values below this concentration). A large database that represents the range of expected anthropogenic impacts is necessary to develop these relationships - the ALL sites database fulfilled this need. Parameters that have strong effects on aquatic organisms will likely show a strong relationship, whereas parameters that have only weak or diffuse effects, or effects that may act indirectly or variably depending onother factors, will result in less distinct threshold responses.

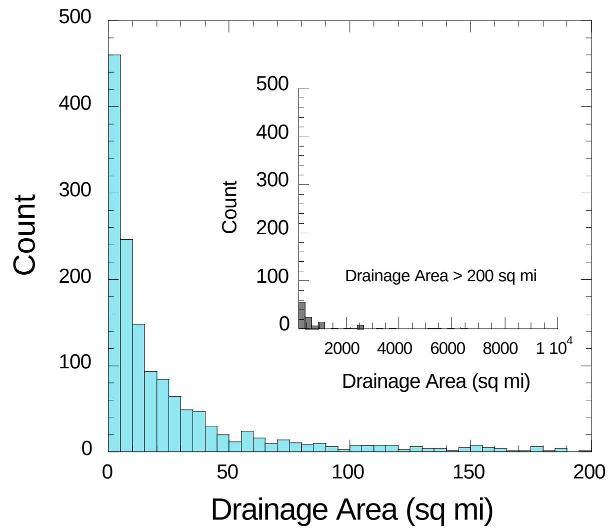


Figure 8. Drainage are distribution of reference sites used in this paper. Note bias towards small streams.

To simplify calculation of these thresholds, medians (50th percentile), upper quartile (75th), 90th, and 95th percentiles were calculated for each chemical parameter within discrete ranges of the IBI (12-19, 20-29, 30-39, 40-49 [WWH], and 50-60 [EWH]) and ICI (0-9, 10-19, 20-29, 30-39, 40-49 [WWH], and 50-60 [EWH]). These ranges approximate narrative ranges of very poor, poor, fair, good, and exceptional quality. Summaries of these statistics were stratified by ecoregion and stream size and are summarized in Appendix Table 2.

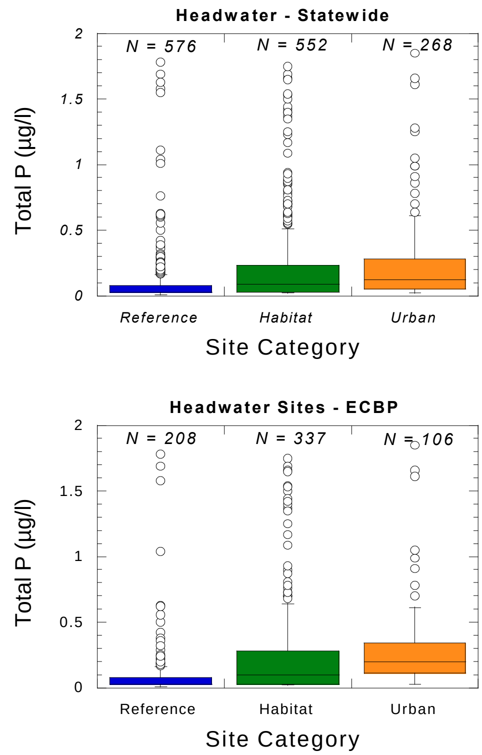


Figure 9. Total phosphorus (mg/l) concentrations by reference site category for headwater streams (drainage area #20mi2), illustrated with box and whisker plots for the ECBP ecoregion (bottom panel) and statewide (top panel).

Regional patterns in nutrient concentration were also examined using Principal Components Analysis (PCA) in a manner similar to that performed by Whittier *et al.* (1987) and Larsen *et al.* (1986) on a smaller Ohio dataset. This multivariate technique was used to collapse multiple nutrient or ionic strength variables into one or more variables that account for most of the variability within the original dataset. A log (x+1) transformation was performed on each variable prior to PCA analyses to achieve normality in the data (Gallant *et al.* 1989). These components were then plotted against each other with points coded by ecoregion to illustrate regional patterns in water chemistry.

RESULTS AND DISCUSSION

Relationship Between Nutrients and Ecoregion

Calculation of instream background or reference concentrations of nutrient parameters requires consideration of how ecoregion attributes affect these concentrations. Most of the analyses produced in this study show that there is a distinct gradient of nutrient concentrations among ecoregions with the Huron/Erie Lake Plain (HELP) ecoregion having the highest levels and the Western Allegheny Plateau (WAP) ecoregion the lowest. This is illustrated by the distribution of TP values by ecoregion in headwater streams (Figure 10), a pattern that is typical for nutrient parameters in Ohio. The Eastern Corn Belt Plains (ECBP) ecoregion generally was most similar to the HELP ecoregion and the Interior Plateau (IP) and Erie/Ontario Lake Plain (EOLP) ecoregions were intermediate between the HELP/ECBP and WAP ecoregions. Intra-regional (within ecoregion) variation in TP concentrations was also evident (illustrated for headwater streams in Figure 11) and appears to correspond to the degree of anthropogenic influences.

The ecoregional differences in nutrient content in REF sites were also illustrated through Principal Component Analyses (PCA). Similar to the analyses of Whittier *et al.* (1987) and Larsen *et al.* (1986), PCA ordinations were derived for nutrients (nitrate-N, TKN, and TP) and, for ionic strength parameters (alkalinity and conductivity) and then for all five variables combined (Table 6). In Figure 12, Panels A-C, the first principal component of the nutrient richness parameters represents the x-axis and the first principal component of the ionic strength parameters represents the y-axis; each point was coded by ecoregion. The HELP ecoregion data is illustrated on both figures, but the other ecoregions are plotted in pairs to increase data comprehension when data points overlap. The patterns here are similar to those found by Whittier *et al. (1987)* and Larsen *et al.* (1988) for a smaller Ohio dataset of wadeable streams.

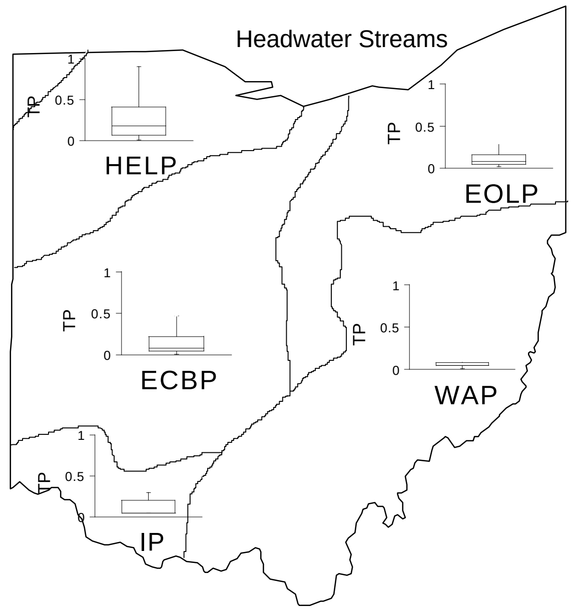


Figure 10. Distributions of total phosphorus (mg/l) concentrations for headwater (ALL) streams (drainage area < 20 mi²) by ecoregion. Box plots are plotted to the same scale.

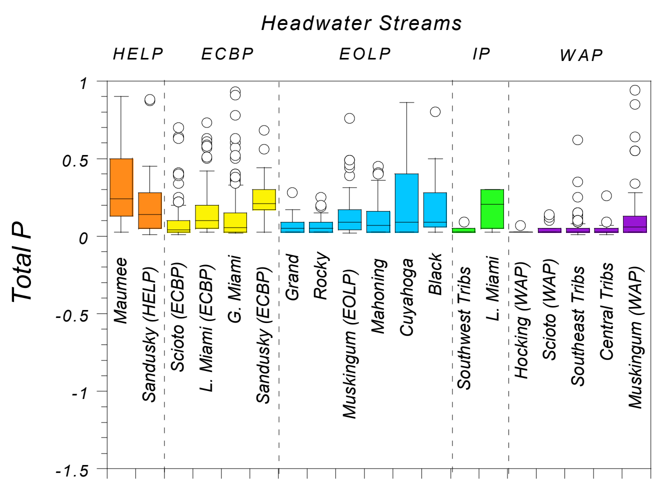


Figure 11. Distributions of total phosphorus (mg/l) concentrations for headwater streams (drainage area < 20mi²) within selected basins of the five ecoregions in Ohio.

The WAP sites were clustered by their similarity in having low nutrient richness, but a wide range of ionic strength results (Panel A). The IP ecoregion was characterized by both low nutrient enrichment and ionic strength (Panel B). The EOLP ecoregion had higher nutrient richness and ionic strength, but with moderate intra-regional variability (Panel A). The HELP ecoregion was characterized by both high nutrient richness and ionic strength at all sites (Panel B). Some extreme values on these plots represent habitat modified sites added to the REF site database to improve the analytical resolution of the PCA (Panel C). Gallant *et al.* (1989) explained that it was useful to perform PCA for nutrient and ionic strength parameters separately because the nutrient concentrations observed were largely of human origin, whereas the ionic strength parameters are largely derived from the natural properties of watersheds (*e.g.,* soils, geological features). One overall PCA, however, with both ionic strength *and* nutrient parameters included, showed similar patterns when the first and second principal components were plotted because the first component reflects nutrients and the second reflects ionic strength (Table 6; Figure 12, Panel D).

Table 5. Distribution of REF sites by suspected or known impact types. See Table 4 for impact type descritptions.

	A	C	H	I	M	N	R	W
A	50 (54)				4			
C		112 (139)			27			
H			201 (201)					
I				2 (12)	10			
M					34 (75)			
N						7 (7)		
R							461 (461)	
W								6 (6)

Associations Between Nutrient Concentrations and Aquatic Community Performance

At ALL sites there is typically an inverse relationship between nutrient concentration and biological index values (Appendices 1 and 2). Specific associations, patterns, and relationships for each parameter are discussed below.

Total Phosphorus

In streams and rivers of the eastern U.S., phosphorus can be a limiting factor in algal and macrophyte growth, and has been observed with greater frequency than nitrogen limitation (Newbold et al. 1983; Sharpely et al. 1994). Median TP concentrations at REF sites (i.e., background concentrations) were typically less than 0.10 mg/l except in the HELP ecoregion and in large rivers where median concentrations varied between 0.15 to 0.40 mg/l (Appendix 1, Figure 13). When ALL sites except large rivers are considered, ranges of median TP at sites with IBI or ICI values indicative of exceptional performance (values ranging from 50-60) were similar to that found at REF sites where concentrations were generally less than 0.10 mg/l (Appendix 2). TP concentrations at ALL sites with IBI or ICI values indicative of good performance (values ranging from 40-49) were generally higher with median TP values of between 0.10-0.20 mg/l. The exception was in the WAP ecoregion where TP concentrations were similar to sites with exceptional IBI or ICI scores (50-60; Appendix 2).

Table 6. Principal Components Analysis for reference sites < 200 mi2 drainage area (headwater and wading sites), including habitat modified sites, for three nutrient richness (nitrate-N, TKN, Total P) and two ionic strength parameters (conductivity, alkalinity).

PCA for nutrient richness measures: (N= 234)

Eigenvalues:	Magnitude	Variance Prop.
Value 1	1.871	0.624
Value 2	0.863	0.288

Unrotated Factors	Factor 1	Factor 2
Log (nitrate)	0.531	0.844
Log (TKN)	0.867	-0.359
Log (TP)	0.915	-0.149

PCA for ionic strength measures: (N= 234)

Eigenvalues:	Magnitude	Variance Prop.
Value 1	1.624	0.812
Value 2	0.376	0.188

Unrotated Factors	Factor 1	Factor 2
Log (alkalinity)	0.901	-0.434
Log (conductivity)	0.901	0.434

PCA for ionic strength and nutrient richness measures together:

Eigenvalues:	Magnitude	Variance Prop.
Value 1	1.624	0.812
Value 2	0.376	0.188

Unrotated Factors	Factor 1	Factor 2
Log (alkalinity)	0.901	-0.434
Log (conductivity)	0.901	0.434

.

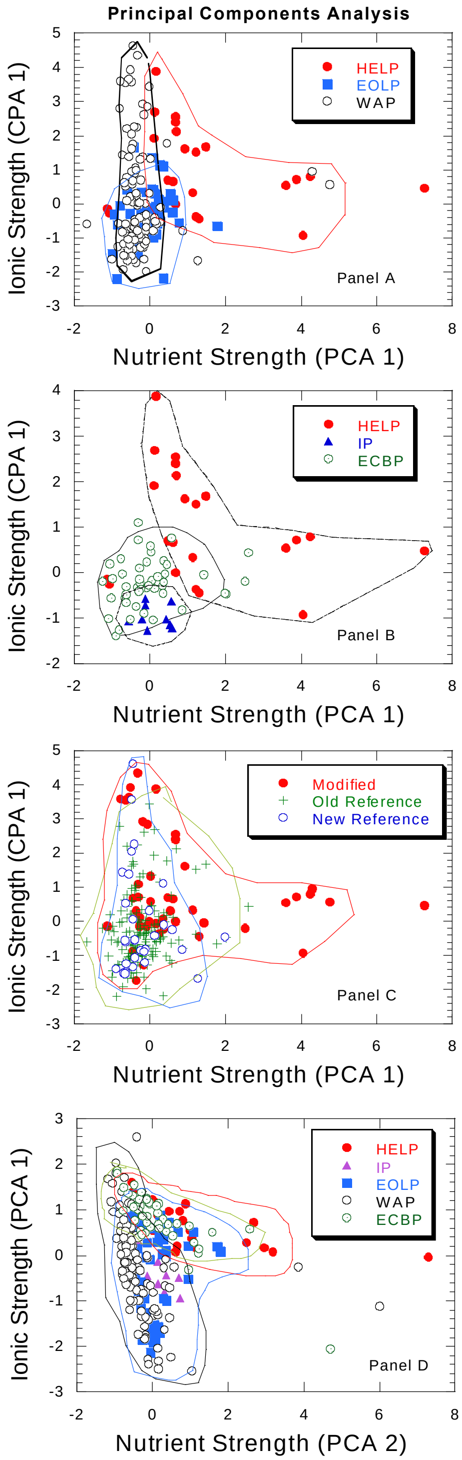


Figure 12. Plots of principal components for ionic strength on nutrient strength for reference data (including habitat modified sites) in Ohio by ecoregion (Panels A and B), and Reference Type (Panel C). Panel D is the

For most ecoregions and stream sizes the higher median TP values correspond to lower IBI or ICI values (as illustrated by headwater and wadeable streams in the ECBP ecoregion; Figure 14, ALL data). IBI and ICI values less than the WWH aquatic life use criteria were associated with substantially higher TP values. The range of observed median TP concentrations is especially narrow for the EWH range of the IBI and ICI suggesting that these communities are especially sensitive to increased concentrations of TP or associated factor(s). Stream substrate quality is also strongly associated with exceptional biological communities in Ohio (Rankin 1989, 1995; see habitat section of this paper) and the dynamics involved may suggest some possible interactions with TP. For example, silt and muck substrates (associated with low IBI scores) not only have negative effects and provide poor habitat for large macroinvertebrates and fish (Rabeni and Smale 1995), but can provide a significant supply of dissolved phosphorus for nuisance algal growth (Sharpely *et al.* 1994). The lower TP concentrations associated with the highest IBI and ICI values suggests that TP or some covariate (*e.g.*, sediment, other wastewater constituents) directly influence biological community performance. Miltner and Rankin (1998) showed that when IBI and ICI scores from wadable or headwater streams were regressed against TP, the regression coefficient for TP was not influenced by the introduction of covariates (e.g., COD, TSS, NH_3-N, DO, metals), demonstrating that the relationship with TP was independant of the covariates tested. However in larger stream size classes, TP did not explain a significant proportion of the variation in IBI or ICI scores when covariates were included in the regression models. The lack of association between TP and biotic index scores in larger streams and rivers may be because TP is present in concentrations saturating to algal growth (*i.e.*, not limiting), given the high background concentrations (> 0.1 mg/l, Appendix Table 1; Figure 13). In other words, larger streams in Ohio are enriched with respect to phosphorus.

A comparison of TP with a measure of habitat quality, the Qualitative Habitat Evaluation Index (QHEI) shows that TP concentrations are typically lowest at locations with high quality habitat (*i.e.,* higher QHEI scores; Figure 15). Habitat quality, and consequently biological community performance, is influenced by riparian quality on two scales - locally at the site and by the cumulative condition of the riparian zone throughout a watershed. Wooded riparian zones are effective in reducing TP, sediment loadings, and nitrates to streams and rivers (Fennessy and Cronk 1997). The low TP values at sites with high quality habitat are most likely related in part to the detention of sediment (filtering by riparian areas). This in turn prevents TP from being delivered since much of the TP in streams is delivered adsorbed onto sediment particles (Baker 1985). Maximum phosphorus retention is attained by mixed herbaceous and wooded riparian buffers (Fennessy and Cronk 1997).

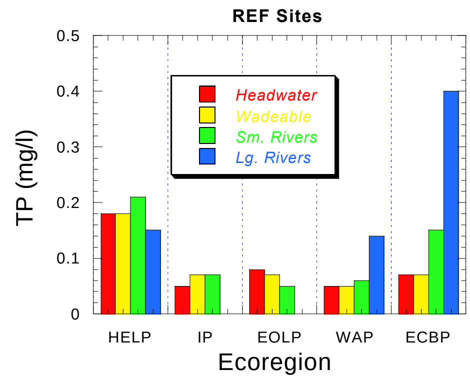


Figure 13. Median background concentrations of total phosphorus (mg/l) at reference (REF) sites by ecoregion and stream size.

In small streams with a dense, closed canopy, sunlight becomes the factor limiting to algal growth (Cummins 1974). In other streams algal production can be limited by nutrients, temperature, spates (high flow events), and grazing (Benke *et al.* 1988). As illustrated in Figure 3 the delivery of phosphorus to streams, the rate at which phosphorus is processed instream, and the amount of phosphorus which is readily available in the water column are affected by the condition of the riparian zone and instream habitat quality. In channelized streams where habitat quality is poor (low QHEIs), riparian zones are degraded, and a tree canopy is absent, nutrient spirals are short and phosphorus turnover is high. In this situationphosphorus is readily available in the water column because it enters the stream attached to suspended sediments and the rapid release from soft bottom sediments. In natural streams where habitat quality is good or exceptional (high QHEIs), riparian zones are intact, and a tree canopy is usually present, nutrient spirals are longer and much of the phosphorus is cycled within a diverse community (plants, fish, macroin-vertebrates, detritus, etc.) characteristic of these aquatic environments. The release of phosphorus back into the ecosystem is comparatively slow and proceeds along "orderly" pathways that do not induce nuisance conditions such as the proliferation of tolerant species and undesirable algal biomass.

Low gradient streams (*e.g.,* HELP ecoregion streams), especially headwater and wadeable streams, had higher TP than high gradient streams (Figure 16). This is likely due to the increased deposition and longer retention (*i.e.,* longer lasting "flood subsidies") of fine materials (*e.g.,* clayey silts) in low gradient streams and the recycling of phosphorus from these sediments. The aquatic biota present in streams with higher quality habitat typically are more efficient at processing the coarse organic material (CPOM) that enters the stream (*e.g.,* net-spinning caddisflies, shredders). TP is more efficiently converted into desirable aquatic biomass (macroinvertebrates, fish, macrophytes) which impedesthe entry of readily available P into the water column. Certain macroinvertebrate taxa are adapted to capturing particles of organic matter within specific size ranges and their abundance and distribution may be related to what is delivered from upstream reaches (Wallace *et al.* 1991).

The processing of phosphorus into desirable plant, invertebrate, or fish biomass as opposed to that which stimulates excessive algal production, can also affect dissolved oxygen (D.O.) and pH levels. Sites with excessively high algal production can exhibit wide variations in diel D.O.levels with high daytime readings and low night-time values; pH shows a similar pattern. An association between the distribution of sites with high TP concentrations and the probability of observing D.O. levels below 5 mg/l was evident in the REF data base (Figure 17). In samples with TP concentrations less than 0.10 mg/l (associated with exceptional ICI and IBI values) D.O. levels were rarely less than 5 mg/l. When TP was greater than 2 mg/l more than 25% of the D.O. values were less than 5 mg/l. The D.O. database consisted mostly of daytime readings which means that the true minimum values were likely lower.

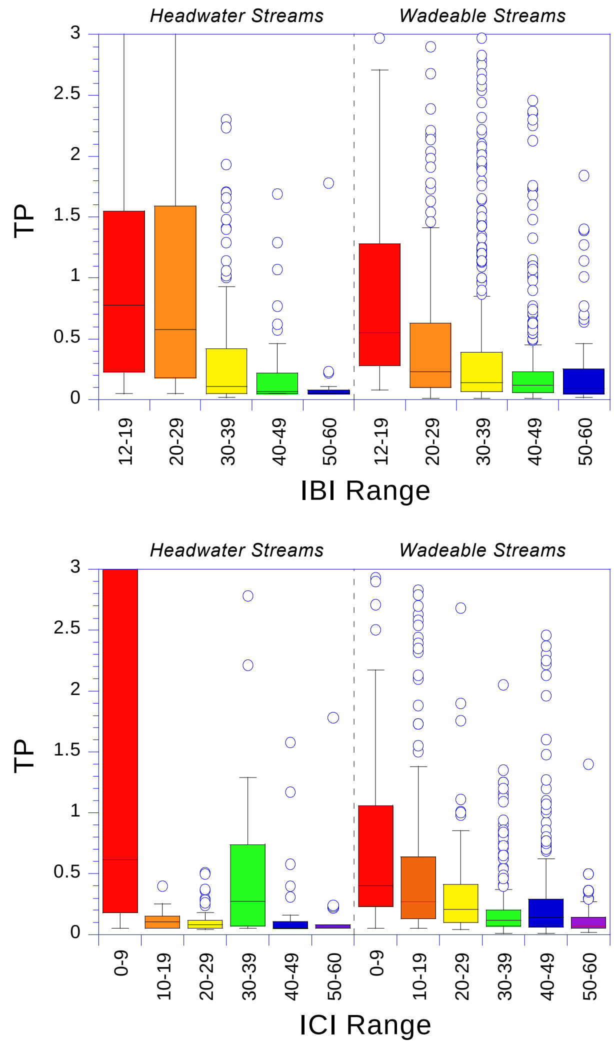
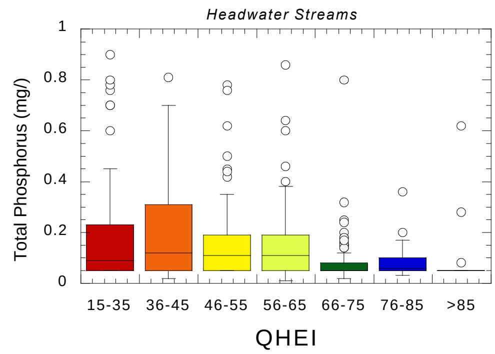


Figure 14. Background concentrations of TP (mg/l) at ALL sites by IBI range (top) and ICI range (bottom) for headwater and wadeable streams in the ECBP ecoregion.

Phosphorus dynamics in rivers and streams are clearly influenced by the quality of the instream habitat and riparian zone. While most nonpoint source abatement strategies emphasize the reduction of phosphorous through the reduction of sediment impacts from upland runoff, the critical role that instream habitat and riparian zone quality also play needs to be recognized. Riparian and instream habitat protection and enhancement efforts should greatly aid in reducing TP loads and in restoring a longer, more natural, nutrient spiral by providing biological and physical sinks for TP. This is accomplished by minimizing the amount of sediment bound phosphorus in the low flow channel through uptake and sequestering of phosphorus in woody biomass (see Malanson 1993), by providing phosphorus in the form of coarse organic material (*e.g.,* leaf litter, detritus), and by reducing sunlight (especially in headwater and wadeable streams). The key to deriving and maintaining high quality benefits provided by intact aquatic ecosystems is to preserve the attributes that foster long nutrient spirals.

Nitrate-Nitrogen (NO_3-N)

Unlike TP, NO_3-N is less frequently the limiting nutrient in algal and macrophyte growth in the lotic ecosystems of in the Eastern US (Newbold 1992). Background concentrations (medians) at REF sites illustrate ecoregional and stream size differences in NO_3-N levels (Figure 19). The effects of increased NO_3-N levels on near-field aquatic community performance under low flows are less clear than for TP. Although TP values were lowest at both REF and ALL sites with high IBI and ICI values, associations between NO_3-N concentrations and the IBI and ICI were more variable (Figure 20). Scatter plots of median NO_3-N values versus IBI and ICI results for each stream size and coded by ecoregion (ALL data; Appendix 4) also illustrate this variability. Only the highest median NO_3-N values (>3-4 mg/l) have a relationship with the IBI and ICI (ALL data) and consistently only in headwater streams (IBI only) and small rivers (IBI and ICI). In large rivers median NO_3-N seldom exceeds >4 mg/l and no relationships with the biological indices below this concentration were apparent. .



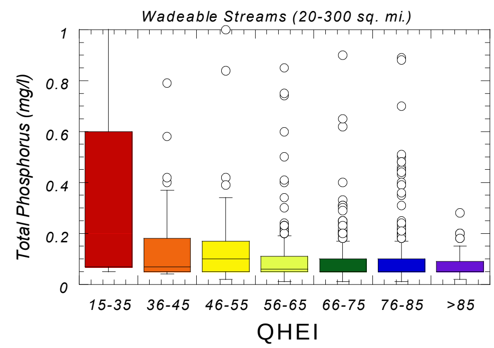


Figure 15. Background concentrations of total phosphorus (mg/l) at ALL sites by QHEI range for headwater streams (top) and wadeable streams (bottom) in the ECBP ecoregion.

Drawng a relationship between nitrogen and the IBI or ICI statewide (ALL data) is confounded by several factors. In the ECBP ecoregion, high IBI or ICI scores occur with high median NO_3-N concentrations (Appendix 4). This was especially apparent for the wadeable streams where virtually the only sites with IBI or ICI values above 40 and median NO_3-N concentrations above 2.5 mg/l occurred in this ecoregion. Excepting the HELP ecoregion, the background concentrations of NO_3-N in the ECBP are 2-4 times higher than the other ecoregions for a given stream size category due to the prevelance of tile drainage.

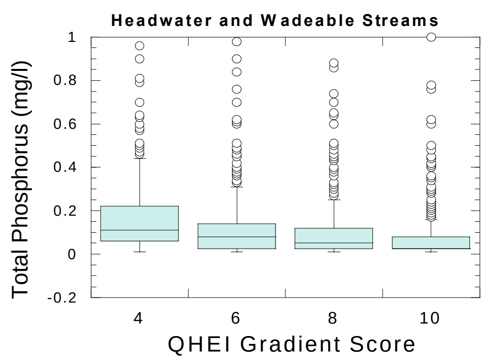


Figure 16. Total phosphorus by QHEI stream gradient score for headwater and wadeable streams.

Habitat in most of the ECBP streams where high IBI values were observed ranged from good to excellent, thus NO_3-N, at the concentrations observed in the ECBP ecoregion (<3-4mg/l), did not appear to negatively affect the biota at such sites. Neither was NO_3-N as strongly correlated with habitat quality as TP, especially in wadeable streams (Figure 21). This is likely a result of the ECBP having a combination of intensive agricultural tile drainage (*i.e.*, high potential for NO_3-N delivery) with many streams having intact habitat (high QHEI scores). By contrast, other ecoregions have either intensive agriculture and severely degraded habitat (HELP ecoregion), or less intensive agriculture overall (EOLP, WAP, and IP ecoregions). Consequently, high NO_3-N concentrations in other ecoregions are often associated with other impacts (*e.g.,* low D.O, high ammonia-N concentrations from wastewater treatment plants or livestock waste, and other WWTP effluent constituents) that tend to lower IBI and ICI scores. Similarly for large rivers, low nitrate concentrations are disproportionately associated with rivers having industrial pollution (Mahoning River), acid mine waste (Hocking River), or catchment scale habitat disturbances (Hocking and Maumee Rivers) that may overshadow effects from nutrients. Furthermore, existing reference conditions in small and large rivers are highly enriched with respect TP and NO_3-N, especially in the ECBP (see preceeding discussion on phosphorus). Consequently, existing community structure in small to large rivers may reflect the prevailing enriched conditions.

Finally, estimates of low flow NO_3-N are subject to variation due to the duration over which concentrations may remain elevated following a storm event, especially compared to the pattern for TP (Figure 22; after Baker 1985). TP concentrations peak prior to the peak of the hydrograph and decline sharply afterwards (Figure 22, upper). NO_3-N does not reach the peak concentration until after the peak of the hydrograph and remains elevated for a substantial period after the flow returns to lower levels (Figure 22, lower). A sample collected before a storm event compared to a sample collected after the flow hydrograph has returned to previous conditions could result in different NO_3-N concentrations for the same flow. Our measures of low flow RNC for NO_3-N may therefore be biased upwards as the effect of a storm event lingers well past the time that the flow hydrograph has returned to normal. This phenomenon may in part explain why NO_3-N and TP were not strongly correlated (Table 7). Consequently, the analyses here may not reliably estimate the range of effects of RNC for NO_3-N on aquatic life.

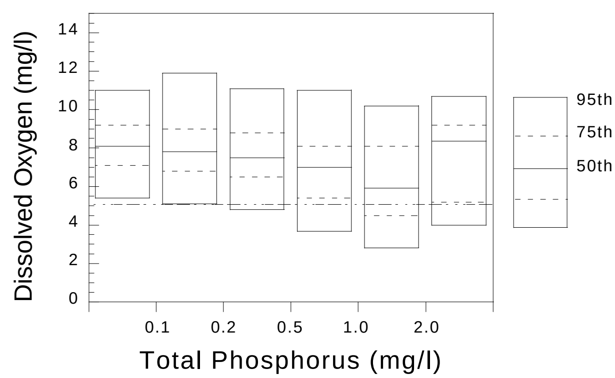


Figure 17. Distributions of daytime dissolved oxygen concentrations sampled at low-flow plotted by total phosphorus concentrations form the REF data set. The horizontal line denotes the 5.0 mg/l dissolved oxygen Water Quality Standard.

Table 7. Correlation matrix for nitrate, TKN, TP, alkalinity and conductivity from 234 REF headwater and wadeable stream sites in Ohio. Variables were log transformed.

	TKN	Nitrate-N	TP	Alkalinity	Conductivity
TKN	1.00	0.19	0.72	0.17	0.14
Nitrate-N		1.00	0.33	0.23	0.07
TP			1.00	0.15	0.15
Alkalinity				1.00	0.45

Another consequence of this lag effect is that nitrogen is less frequently the nutrient limiting to algal growth. Therefore, additions of phosphorus should have larger consquences. Cummulative frequency distributions of IBI ranges by nitrate-N and TP for headwater streams in the ECBP illustrate the effect dramatically. IBI ranges (*e.g.*, exceptional vs good) separate rapily with increasing TP concentration, whereas the separation with respect to nitrate-N does not occur until concentrations exceed 3 mg/l (Figure 18, upper panel). Showing the relationships with scatter plots (Figure 18, lower panel) illustrates linear relationship with phosphorus and the threshold effect for nitrogen.

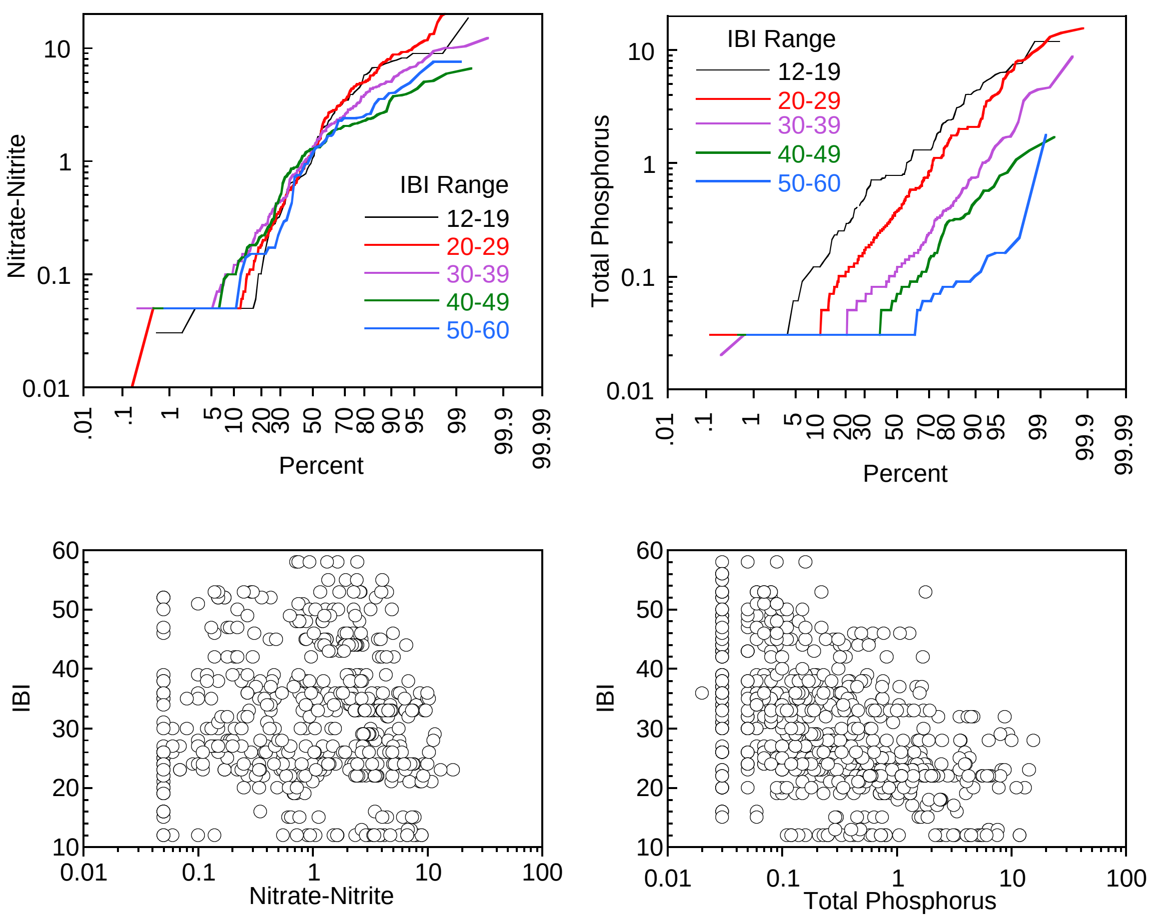


Figure 18. Cumulative frequency distributions by of IBI scores by IBI range and nitrate-N (upper left) and total phosphorus (upper right). Lower panel, scatter plots of IBI scores by nitrate-N and total phosphorus. All plots are for headwater streams in the ECBP ecoregion.

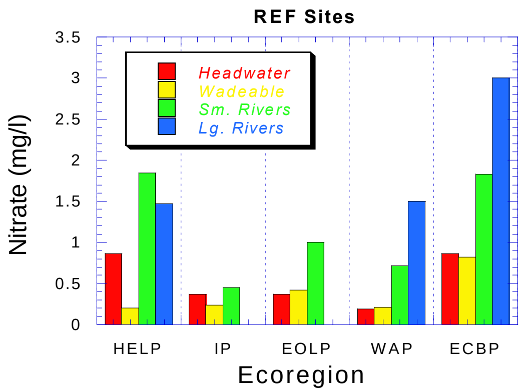


Figure 19. Median background concentratoins of nitrate (mg/l) at REF sites by ecoregion and stream size.

Unlike TP, D.O. concentrations show little correlation with high nitrate-N concentrations at low flows (Figure 23). Low D.O. concentrations associated with nitrogenous substances are frequently more related to the process of nitrification (conversion of ammonia to nitrite, then nitrite to nitrate) which can be a major source of biochemical oxygen demand (Kaushik *et al.* 1981). Nitrate loss in streams occurs through microbial denitrification and increases in the presence of organic biomass, such as leaf litter (Kaushik *et al.* 1981). An intact riparian zone, which can deliver a continual supply of carbon as leaves and woody debris, may reduce downstream delivery of nitrates to sensitive waterbodies. This occurs through the sequestering of nitrogenous substances within biomass similar to the process previously described for TP.

High loadings of nitrate-N can adversely affect lakes, reservoirs and estuaries, typically through compositional changes in phytoplankton (*i.e.*, toward dinoflagellates: *Pfisteria* blooms in the Southeastern US). However, the near-field effects of nitrate-N on periphyton communities and higher aquatic communities in Ohio rivers and streams, over the range concentrations observed in this study, are unknown, under-reported, or equivocal.

Relationship Between Habitat and Aquatic Community Performance Indicators
Habitat quality is a principal determinant of aquatic community performance in Ohio streams and rivers (Rankin 1989; 1995) and elsewhere (Karr and Schlosser 1977; Gorman and Karr 1978). However, habitat quality is not specifically addressed by conventional approaches designed to reduce the effects of nonpoint sources of pollution on aquatic life. The U.S. EPA TMDL guidelines (U.S. EPA 1991) state:

> "The TMDL establishes the allowable loadings or other quantifiable parameters for a waterbody and thereby provides the basis for States to establish water quality-based controls. These controls should provide the pollution reduction necessary for a waterbody to meet water quality standards."

The guidance further states:

> "EPA recognizes that it is appropriate to use the TMDL process to establish control measures for quantifiable non-chemical parameters that are preventing the attainment of water quality standards. Control measures, in this case, would be developed and implemented to meet a TMDL that addresses these parameters in a manner similar to chemical loads. As methods are developed to address these problems, EPA and the States will incorporate them into the TMDL process."

The Ohio EPA habitat quality database contains the “quantifiable non-chemical” habitat parameters that could be used in the TMDL process as suggested by U.S. EPA. The introduction of habitat measures into the TMDL process may require some "new" thinking, however, since habitat stressors are not necessarily analogous to the concept of “pollutant loads” (sediment loadings possibly being an exception where the source of these sediments is identified). Often, the stressor agents are direct manipulations of the physical habitat via riparian modification, dredging, and channelization. Because direct measures of habitat quality (physical measures and biological indices) exist, monitoring data can be used to determine success or failure of implementation strategies. In addition, the BMPs necessary to restore habitat quality are well known and include the protection and enhancement of natural features and processes. The specific measures needed to restore habitat quality and whether such activities are cost effective or acceptable will vary regionally. Some of this regional variation is related to soils and stream geomorphology. Rosgen (1994) provides a list of stream types, as classified by his stream classification system, with ratings on their sensitivity to disturbance, recovery potential, and streambank erosion potential that may prove useful in lotic habitat restoration efforts as part of the TMDL process.

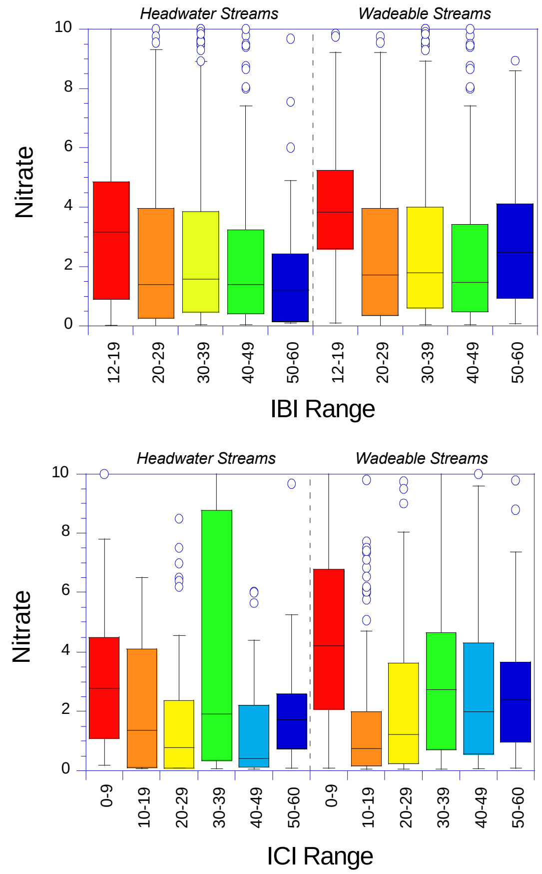


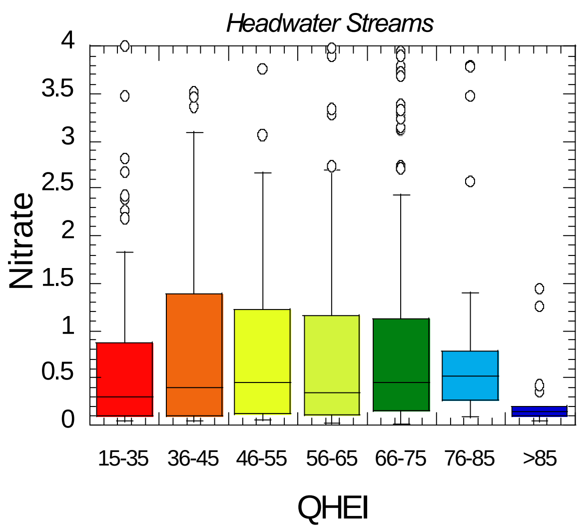
Figure 20. Background concentrations of Nitrate (mg/l) at ALL sites by IBI range (top) and ICI range (bottom) for headwater and wadeable streams in the ECBP ecoregion.

This section describes the key habitat parameters that can be used to establish goals for evaluating the success of habitat restoration activities and BMP implementation. As is discussed elsewhere, habitat must be considered in concert with the more traditional nonpoint source associated chemical parameters when evaluating strategies for restoring and protecting watersheds where a goal is the attainment of the biological criteria and conditions specified by the Ohio Water Quality Standards.

Relationship of Critical Habitat Parameters to Aquatic Community Performance

Ohio EPA uses the Qualitative Habitat Evaluation Index (QHEI; Rankin 1989, 1995) to assess the physical habitat quality of streams and rivers. This index measures the important components of lotic macrohabitat that are essential to sustaining high value aquatic communities. The major categories of macrohabitat include substrate quality, instream cover (physical structure), stream channel morphology and condition, riparian quality and bank erosion, pool and run-riffle quality, and gradient. Comparisons between the QHEI and the IBI resulted in a list of critical habitat components associated with the occurrence of IBI scores corresponding with the Warmwater Habitat or Exceptional Warmwater Habitat biocriteria ("warmwater attributes") and a list of components that are associated with degraded communities (high and moderate influence "modified attributes"; Rankin 1989; 1995). These modified attributes were further divided into "high" influence or "moderate" influence attributes based on the statistical strength of the relationships. By examining a combined database of least impacted reference sites and physically modified reference sites essentially free from point source associated chemical impacts, a relationship was developed between the IBI and the accrual of modified habitat attributes (Figure 23).

Using an IBI of 40 as a representative WWH biocriteria value, the accumulation of modified attributes corresponds to a decrease in the proportion of sites meeting this IBI threshold (Figure 24, upper). An accumulation of four modified attributes corresponds to fewer than 50% of sites achieving an IBI of 40. An accumulation of six modified attributes results in fewer than 25 percent of the sites achieving the threshold IBI value and with seven or more the occurrence is rare. Greater than 6 modified attributes virtually eliminates the probability of achieving an IBI of 40 (Rankin 1989; 1995). High influence modified attributes are particularly detrimental given that the presence of one is likely to result in impairment, and two will likely preclude a site from achieving an IBI of 40 (Figure 24, lower).



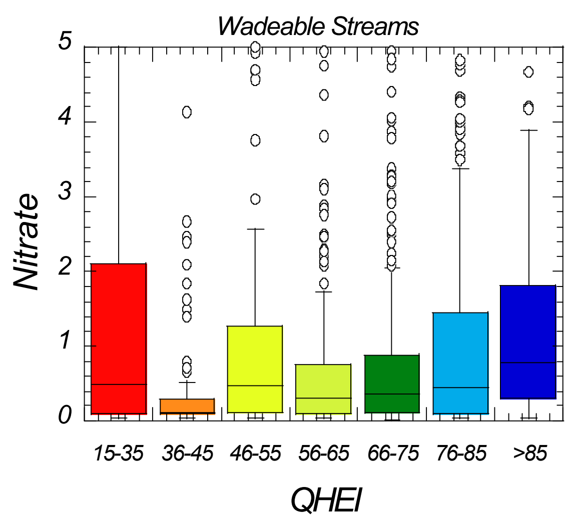


Figure 21. Background concentrations of nitrate-N (mg/l) at ALL sites by QHEI range for headwater (left) and wadeable (right) streams in the ECBP ecoregion.

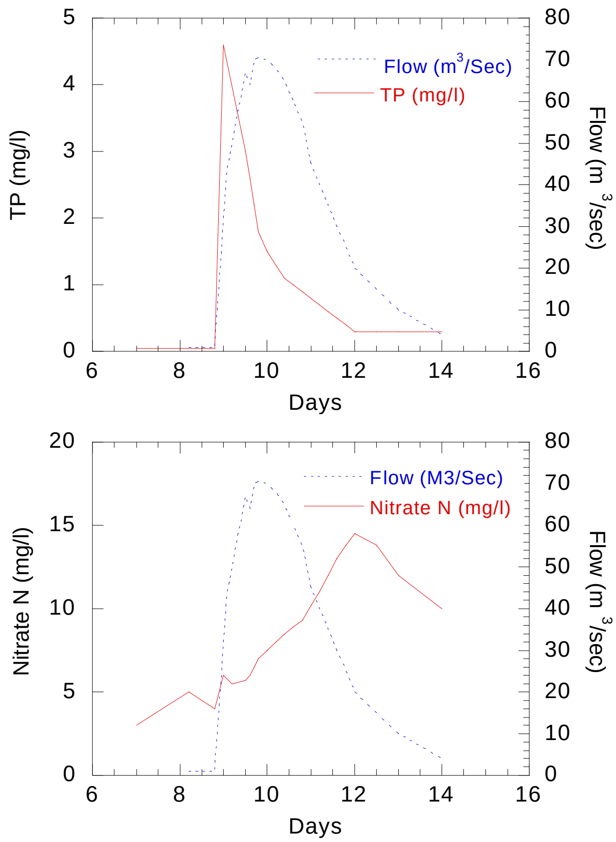


Figure 22. Typical concentration pattern for TP (upper) and nitrate-N (lower) in relation to stream flow surrounding a storm event (modified from Baker 1985).

Some modified attributes (*e.g.*, recent channelization, silt/muck substrates, no/sparse cover) are particularly strongly associated with low IBIs ("high influence modified attributes") probably because they are associated with other negative attributes. The accumulation of even one of these attributes is associated with lower IBI scores, and the presence of two of these attributes makes a site unlikely to achieve the WWH biocriterion (Figure 24). As a "rule of thumb", goals for BMPs and habitat restoration should strive to reduce the number of modified attributes (moderate influence) to four or fewer, reduce the number of modified to warmwater attributes to less than 6 and preferably less than 4, and eliminate all high influence modified attributes. These measures are needed to provide the habitat capability to have a reasonable probability of attaining the WWH biocriteria (see Table 7).

Habitat Restoration/Protection, Landscape Attributes, and Headwater Streams

Landscape scale is an important concept in habitat evaluation, protection, and restoration. Restoration efforts should focus on a reach scale rather than on single sites because the influence of habitat accrues at a watershed or subwatershed level. Previous analyses by Rankin (1989, 1995) have demonstrated this phenomenon. For example, where a stream reach or subbasin is largely unmodified, including the headwaters, isolated sections of modified habitat may exhibit biological community performance superior to what might be “expected” based on a linear correlation of site-specific habitat conditions with site-specific biological performance. As discussed by Pulliam (1988), a preponderance of high quality habitat reaches acts as a *source* of species and individuals. These sites with high biological integrity can ameliorate the negative influence of locally degraded habitat and result in higher than "expected" IBI scores. However, as modified attributes accumulate *throughout* a watershed, the modified sections begin acting as sinks (*sensu* Pulliam), extirpating sensitive individuals and species, and lowering biological integrity even at the remaining sites with higher quality habitat. Schlosser (1995) provides four possible large-scale spatial mechanisms for such patterns observed in fish populations.

The average IBI scores at *reference sites* within the 93 subbasins delineated by Ohio EPA are significantly correlated with the average QHEI scores at *all sites* within the subbasins (Rankin 1995). The relationship is consistent with Pulliam's (1988) concept of sources and sinks. The concepts previously summarized in Figures 3 and 4 illustrate the mechanisms responsible for habitat associated impairment of biological community performance. These relationships emphasize the need to protect and manage habitat on a watershed scale. Well-intentioned attempts to protect and restore habitat on a piecemeal basis (*i.e.,* small, isolated preserves) may be insufficient to restore and maintain the biological integrity and attain the provisions of the Ohio WQS for lotic ecosystems. This is yet another reason for including habitat considerations in the TMDL process.

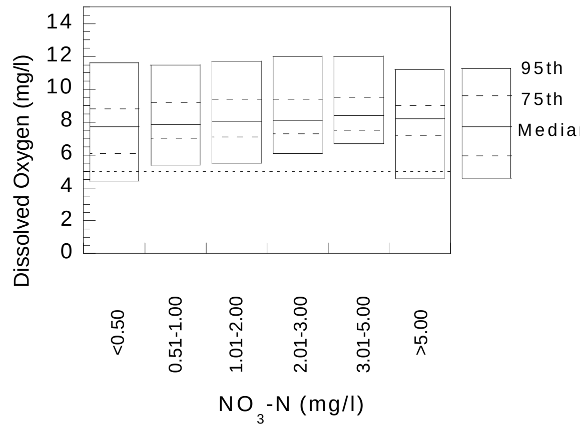


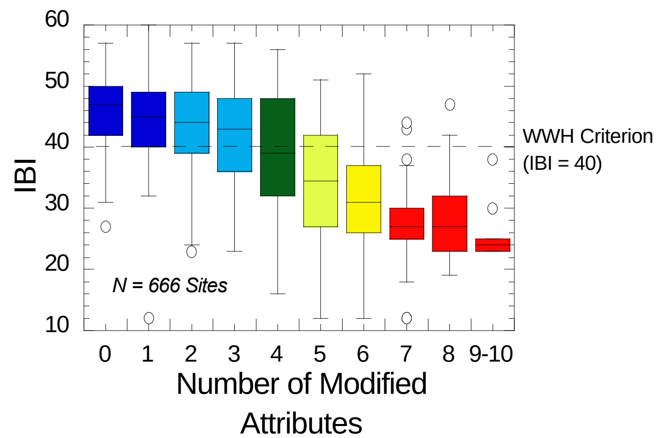
Figure 23. Background concentrations of nitrate-N (mg/l) vs daytime dissolved oxygen (mg/l) from low-flow REF samples. The horizontal line represents the 5 mg/l dissolved oxygen Water Quality Standard.

The concept of watershed scale habitat influences on biological integrity has even greater consequences when the role of headwater streams is considered. The prevailing notion that headwater streams are of little or no value or importance to overall watershed function has, and will continue to have, serious negative consequences for downstream water bodies (*i.e.*, larger streams and rivers, lakes, estuaries, etc.). While headwater streams are proportionally smaller in terms of physical size and volume, their sheer numbers imply importance in cumulative terms - more than 75% of Ohio streams digitized in the U.S. EPA RF3 database are headwater streams. Furthermore, headwater streams are the *primary interface* between the landscape and aquatic ecosystems. One study reported that bank and bed erosion can approximate two tons of sediment *per meter of stream* where protective buffers are lacking or debilitated (Rabeni and Smale 1995). Current ecological theory of nutrient exchange between the landscape and headwater streams emphasizes the critical role of riparian zones in "regulating these exchanges" (Schlosser 1995). Rabeni and Smale (1995) point out that the quality of a riparian zone is equally as important as the width because the development of rills and gullies can create direct paths that "short-circuit" the nutrient and sediment interception function of riparian zones. For larger streams and rivers, a neglect of headwater stream habitats and riparian zones creates similar problems where sediment delivery from these small streams causes headwater streams to act as point sources of sediment and nutrients. As Rabeni and Smale (1995) point out: "Continuity is the key: even the finest bucket is useless if a hole is poked through the bottom." These facts and evidence support approaches to headwater stream protection that are focused on the improved management of riparian zones in attempts to reduce sediment and nutrient delivery (*i.e.*, encouraging sediment and nutrient interception, processing, and storage within the riparian areas of headwater streams). An analogy is the function of capillaries within the human circulatory system - their individual size is small, but their collective function is critical to the health of the overall body and its components. In a similar fashion, headwater streams are the "capillaries" of the watershed providing, in the aggregate, essential functions such as the regulation of energy (*i.e.*, nutrient dynamics) and sediment. High value biomass is the end product of watersheds with healthy and largely intact headwaters while low value biomass becomes the predominant output when energy and sediment pathways become short circuited by riparian degradation.

Substrate Metric

The substrate metric of the QHEI measures the predominant substrate types, the number of substrate types present, substrate origin(s), substrate embeddedness, and degree of silt cover. As the QHEI substrate score increases, the likelihood of achieving an IBI consistent with WWH or EWH expectations increases (Figure 25). To ensure a reasonable probability of achieving the IBI threshold, substrate scores should be ≥13-14 to protect or restore the Warmwater Habitat use. For protection or restoration of the Exceptional Warmwater Habitat use, substrate scores should be ≥15-16 on average (Figure 24).

Substrate Embeddedness Another measure of substrate condition is the extent of embeddedness (Platts et al. 1983). As the extent of embeddedness increases, the likelihood of attaining the threshold IBI decreases (Figure 26). Thus substrate embeddedness is another useful habitat criterion for measuring the efficacy of BMPs. Depending on the perceived limitations to aquatic community performance in the stream or watershed of interest, various QHEI metrics, such as substrate condition, can be used as measures of the success of nonpoint source abatement strategies, along with a direct evaluation of the biota. Certain situations may require a more sensitive assessment tool that can detect more subtle habitat shifts in response to positive or negative landuse changes (i.e., before they might be detected by the QHEI).



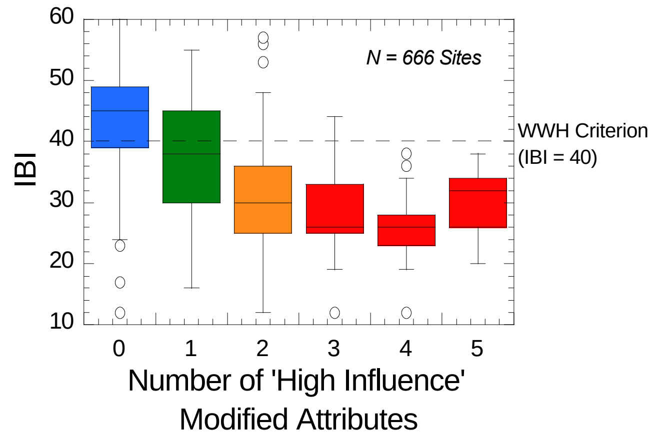


Figure 24. Number of all modified habitat attributes (upper) and high-influence modified habitat attributes (lower) versus the IBI for Ohio streams. Data are from a combination of least impacted and physically modified reference sites generally free from point source impacts.

One tool that may prove useful is the Riffle Stability Index (RSI; Kappesser 1994). This is a procedure to evaluate stream reach and watershed equilibrium in relation to erosion and sediment export in a watershed area. In a stream under equilibrium, the amount of sediment leaving a stream is about the same as the amount of sediment entering. When more sediment enters than is exported, aggradation of riffles and pools occurs and median particle size of the streambed decreases (Kappesser 1994). When export is greater than import, degradation occurs and median particle size of the streambed increases. The RSI compares the median particle size that a stream can carry (measured by the largest particles on a fresh point bar) with the cumulative distribution of particle size within a riffle. The greater the percent of riffle substrates that are finer than the "maximum mobile particle size" (determined from the point bar) the less stable (more aggraded) the riffle habitat. The Ohio EPA reference sites could be used to determine expected RSI values by ecoregion and provide a more sensitive tool to monitor the efficacy of nonpoint source management and abatement strategies than QHEI alone.

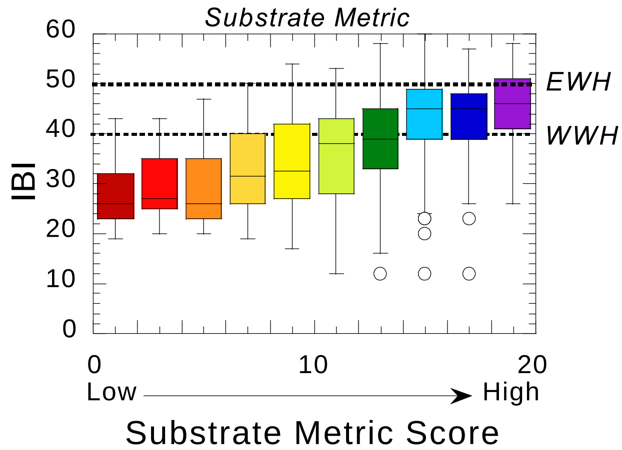


Figure 25. Substrate metric score plotted versus the IBI for Ohio streams. Data are from a combination of minimally impacted and physically modified reference sites generally free from point sources.

Channel Condition

Another useful metric of the QHEI is channel condition. This metric evaluates stream sinuosity (which is reduced when streams are modified), pool-riffle development, extent of channel modifications, and stream channel stability (Rankin 1989) . The pattern summarized in the boxplot illustrated in Figure 27 is similar to the other metrics just discussed. The lower the channel metric score (i.e., the more modification present, the greater the loss of sinuosity and pool-riffle development, and the greater the channel instability) the less likely a site is to achieve the threshold IBI of 40. Depending on the goal (i.e., WWH or EWH), a metric score of >14 (WWH goal) or >17 (EWH goal) may be specified for the channel metric as a target for restoration activities.

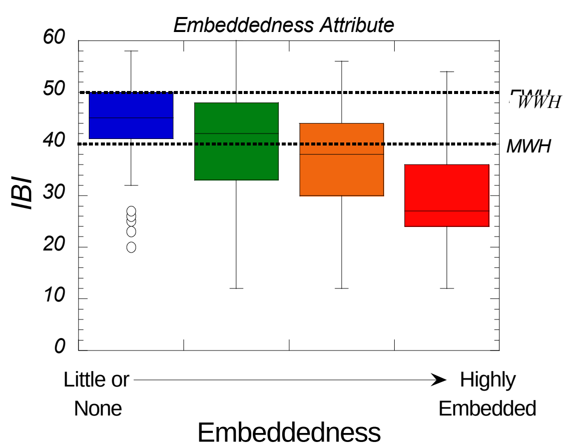


Figure 26. Substrate embeddedness score plotted versus the IBI for Ohio streams. Data are from a combinatin of minimally impacted and physically modified reference sites generally free from point source impacts.

IBI responds strongly to habitat quality, and so follows the ability for a stream or river to attain a specified aquatic life use designation. An obvious consequence of this conclusion is that *habitat data must be considered as an integral part of any attempt to restore aquatic life in a stream or river if such efforts are to succeed*. The selection of BMPs to reduce sedimentation and siltation to streams should consider: 1) whether the channel condition of the stream is natural or modified, and 2) whether the major source of sediment is from upland sources or bank erosion. Implementation of BMPs to reduce upland erosion without consideration of channel condition or other habitat limitations may not be sufficient to restore waterbodies such that WQS are attained, even though overall sediment loadings may be reduced. Similarly, while reductions in upland erosion rates may be beneficial, this *alone* will be insufficient if bank erosion and riparian interactions are not addressed concurrently. Recent work (Cooper *et al.* 1987; Phillips 1989) has illustrated the importance of riparian and other wetlands in the *storage* of fluvial sediments, to the extent that "wetland sediment storage accounts for a significant portion of the total erosion within a drainage basin in a given year... (Phillips 1989)." Habitat protection efforts should focus on protection of this essential riparian function that is often lost when streams are channelized or riparian areas are encroached upon.

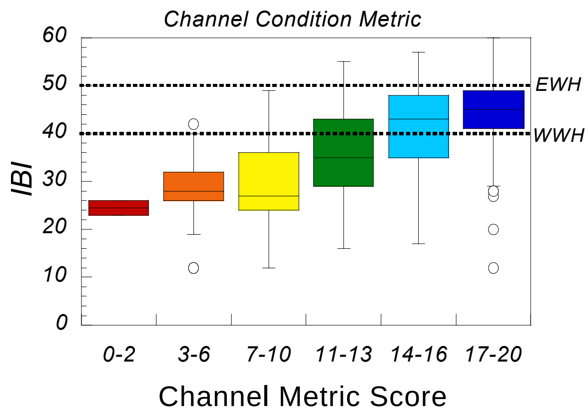


Figure 27. Channel metric score plotted versus the IBI for Ohio streams. Data are from a combination of minimally impacted and physically modified reference sites generally free from point source impacts.

Another consequence related to the importance of habitat is the design and effectiveness of nonpoint source pollution control strategies in streams or watersheds where habitat has been irretrievably modified and attainment of the baseline Clean Water Act goals (*i.e.*, at least WWH) is precluded. In such situations habitat is the controlling factor with respect to aquatic life use attainment. Nutrient reduction strategies for such waters will need to target a different set of criteria. Under these circumstances it may not be productive to invest heavily in nonpoint source pollution abatement, even where other uses (*e.g.,* public water supply) or downstream resources (*e.g.,* phosphorus in Lake Erie) are at issue. Given the importance of habitat in determining the fate and availability of nutrients in the water column, meeting Clean Water Act goals will likely be frustrated without consideration of the critical role of riparian and instream habitat.

Nutrients and "DELT" Fish Anomalies

An increased incidence of "DELT" anomalies (*i.e.*, the percentage of fish with deformities, eroded fins, lesions, or tumors; Ohio EPA 1989) on fish is an indication of chronic (sublethal) stress. Elevated DELT levels in Ohio streams and rivers typically occur in association with marginal or variable D.O. concentrations and increased chemical stress. A moderately elevated level of anomalies (less than 5-10%) accompanied by IBI and MIwb values indicative of good, very good, and exceptional aquatic community performance has been observed in Ohio rivers where summer-fall low flows are dominated by municipal sewage effluent. High levels of anomalies (generally >10-20%) in combination with IBI and MIwb scores indicative of poor and very poor aquatic community performance indicate a more serious problem with toxics (Yoder and Rankin 1995b). In a recent survey (1993) of the Little Miami River basin there was a significant association between the level of DELT anomalies and TP concentrations (Ohio EPA 1995). A key question is whether this relationship is causal, either directly or indirectly (*e.g.*, diurnal D.O. fluctuations through encanced algal productivity, phytotoxins from blue-green algae, dinoflagellates), or if TP is simply coincidental with one or more factors driving the relationship.

To examine the association between TP and DELT anomalies we used sites where the primary and secondary impacts to the biota had been characterized (Yoder and Rankin 1995b) and where data on DELT anomalies and TP were available. Impact types were characterized as largely of point source origin, nonpoint source origin (NPS) or, as indicated by the secondary impact types, a combination of point source and NPS. Because the Little Miami River is designated as EWH, the database was analyzed separately for EWH and for all stream uses combined. The rationale for analyzing EWH separately is that the higher proportion of sensitive fish species characteristic of EWH streams make them more susceptible to the impacts which cause DELT anomalies than species that typically predominate in WWH streams and rivers.

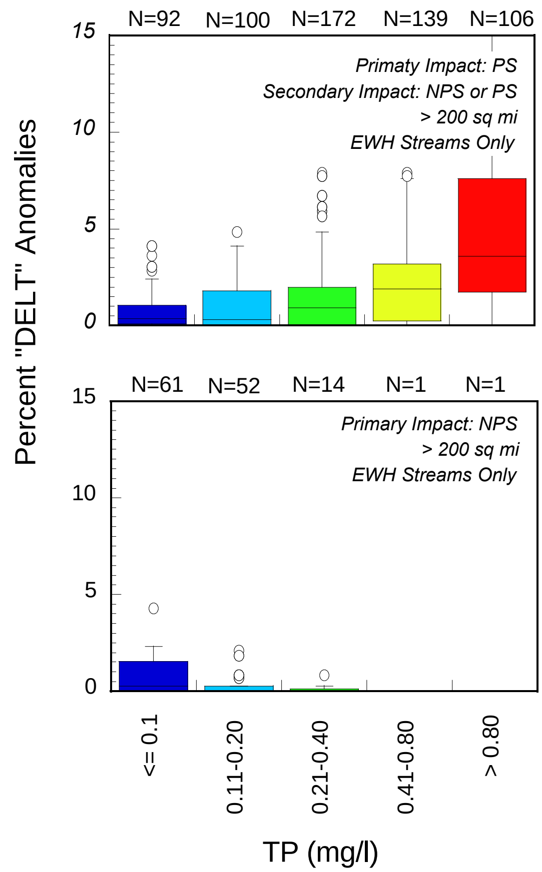


Figure 28. Distributions of percent DELT anomalies versus TP for primarily PS impacted sites (top panel) and primarily NPS impacted sites (bottom panel) in EWH designated streams.

.In EWH rivers (> 200 sq. mi.) dominated by point source impacts, DELT anomalies increased with increasing concentrations of TP (Figure 28, upper). EWH rivers primarily affected by NPS impacts revealed no apparent trend; however, sample sizes for TP > 0.4 mg/l were very small (Figure 28, lower). When the sample size was broadened to include the other aquatic life uses (WWH) the association between DELTs and TP was evident for both point and nonpoint sources (Figure 29, lower). NPS associated impacts showed a similar pattern to point sources, but with a lower incidence of DELT anomalies and fewer data observations in the highest TP categories. The 0.21-0.80 mg/l TP boxes in Figure 29 (upper) are primarily from rivers in the HELP ecoregion that have also been extensively channelized (*e.g.,* L. Auglaize River) or have had their headwater stream network severely modified (*e.g.,* St. Marys River, Tiffin River). The severity of the channel modifications in combination with nutrient enrichment may mimic, in part, the nutrient stresses associated with municipal wastewater treatment plant discharges. Percentile plots of daytime D.O. data from these sites shows that D.O. levels inversely correspond to changes in TP (Figure 30). Since these are day-

Table 8. A process for determining the risk to aquatic life use attainment based on TP and NO3-N concentrations in Ohio rivers and streams. The risk of aquatic life impairment increases as sample statistics increasingly deviate from reference site results (REF) or observed associations between biological community performance and nutrients (ALL data).

Risk of Impairing Aquatic Life	Sample Median	Sample 90th Percentile
None	Less Than REF Median for Aquatic Life Use	Less Than ALL or REF Median for Aquatic Life Use
Minimal	Less Than REF Median for Aquatic Life Use	Less Than ALL or REF 75th %tile for Aquatic Life Use
Low	Greater Than REF Median for Aquatic Life Use	Less Than ALL or REF 75th %tile for Aquatic Life Use
Moderate	Greater Than REF 75th %tile for Aquatic Life Use	Less Than ALL or REF [Median + 2* Interquartile Range] for Aquatic Life Use
Mod./High	Greater Than REF [Median + 2* Interquartile Range] for Aquatic Life Use	Less Than ALL or REF [Median + 2* 75th %tile - Median] for Aquatic Life Use
High	Greater Than ALL or REF [Median + 2* Interquartile Range] for Aquatic Life Use	
Extreme		Greater Than ALL or REF [Median + 2* Interquartile Range] for Aquatic Life Use

time values from streams and rivers with substantial algal activity (particularly under increased TP concentrations) nighttime values are undoubtedly lower and diel swings greater. Gammon (1995) illustrates these D.O. swings in numerous nutrient enriched agricultural streams in Indiana that have had their forested riparian areas removed. While it seems likely that an increased incidence of DELT anomalies on fish is not *directly* caused by TP, indirect effects of TP (increased diel swings in D.O., increased benthic demand of D.O. from decaying algae), and the secondary factors associated with elevated TP, (low D.O. combined with toxics or other chemical stresses below point sources, feedlots, or failing on-site septic systems) do cause DELTS. Direct or indirect effects of TP are all exacerbated by physical changes (*e.g.,* channelization).

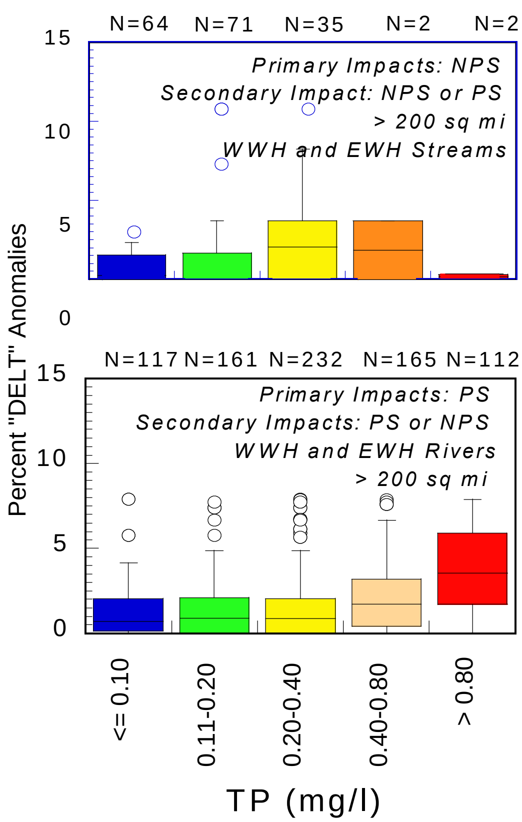


Figure 29. Distributions of percent DELT anomalies versus TP for primarily point source impacted (lower panel) and nonpoint impacted (upper panel) sites. Data are from MWH, WWH and EWH designated wading streams.

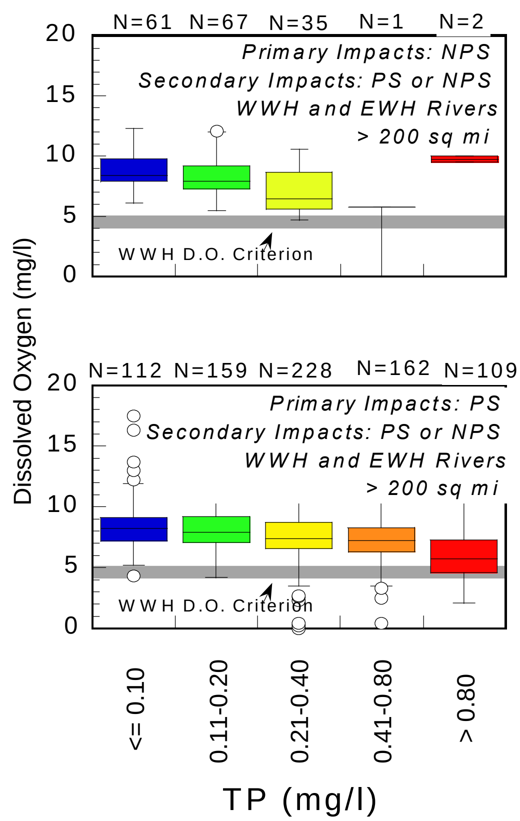


Figure 30. Distributions of dissolved oxygen versus TP for primarily point source impacted (lower panel) and nonpoint impacted (upper panel) sites. Data are from MWH, WWH and EWH designated wading streams.

The patterns observed in headwater and wadeable streams bolster these conclusions. In streams where the primary impact type is point source in origin, DELT anomalies generally increased with TP. Inclusion of toxic-impacted sites from the Ottawa River and HELP ecoregion that coincided with a TP range classes (0.21-0.40 mg/l; Figure 30, upper) added variability to the relationship. The very high incidences of DELT anomalies observed in these streams are a response to toxicity and are largely independent of impacts associated with TP. This is an example where the association between TP and DELT anomalies is casual and not causal. Headwater and wadeable streams that were primarily affected by NPS impacts showed increases in DELT anomalies with increased concentrations of TP (Figure 31, middle). However, the magnitude of increase was low in comparison to point source effects, and was further reduced when sites with a point source related secondary impact type were removed (Figure 31, lower). The results of this analysis indicate that TP is frequently a covariate with other factors resulting in an increased incidence of DELT anomalies, particularly where point source discharges are prevalent. A similar pattern was evident where TP from NPS loadings are high coupled with extensive habitat modifications. In this latter case, the stresses mimic conditions associated with WWTP impacts.

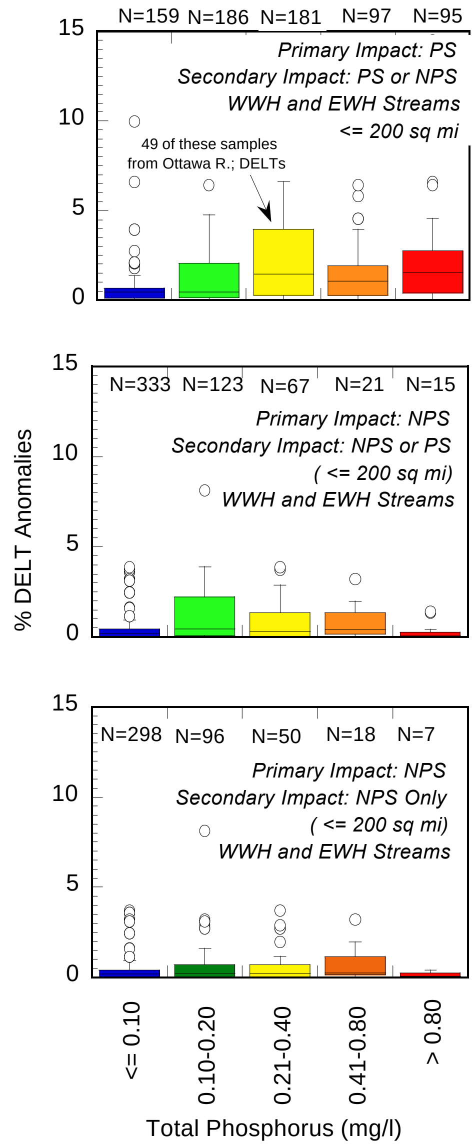


Figure 31. Distributions of percent DELT anomalies versus TP in headwater streams for primarily point source impacted sites (top panel), primarily nonpoint impacted sites (middle panel) and solely nonpoint impacted sites (bottom panel). Data are from MWH, WWH and EWH designated streams

Incorporation of Biosurvey and Habitat Data into TMDLs

U.S. EPA (1991) recognized that non-chemical factors can determine whether a waterbody can attain Water Quality Standards (*e.g.*, designated aquatic life uses). The U.S. EPA (1991) TMDL guidance document states that control measures for quantifiable non-chemical parameters should be developed and implemented in a manner similar to chemical loads. However, no specific guidance has been forthcoming on how such a process should be designed and implemented. Lacking such guidance we here propose a framework for incorporating biological survey results and habitat data into the TMDL process (Figure 32).

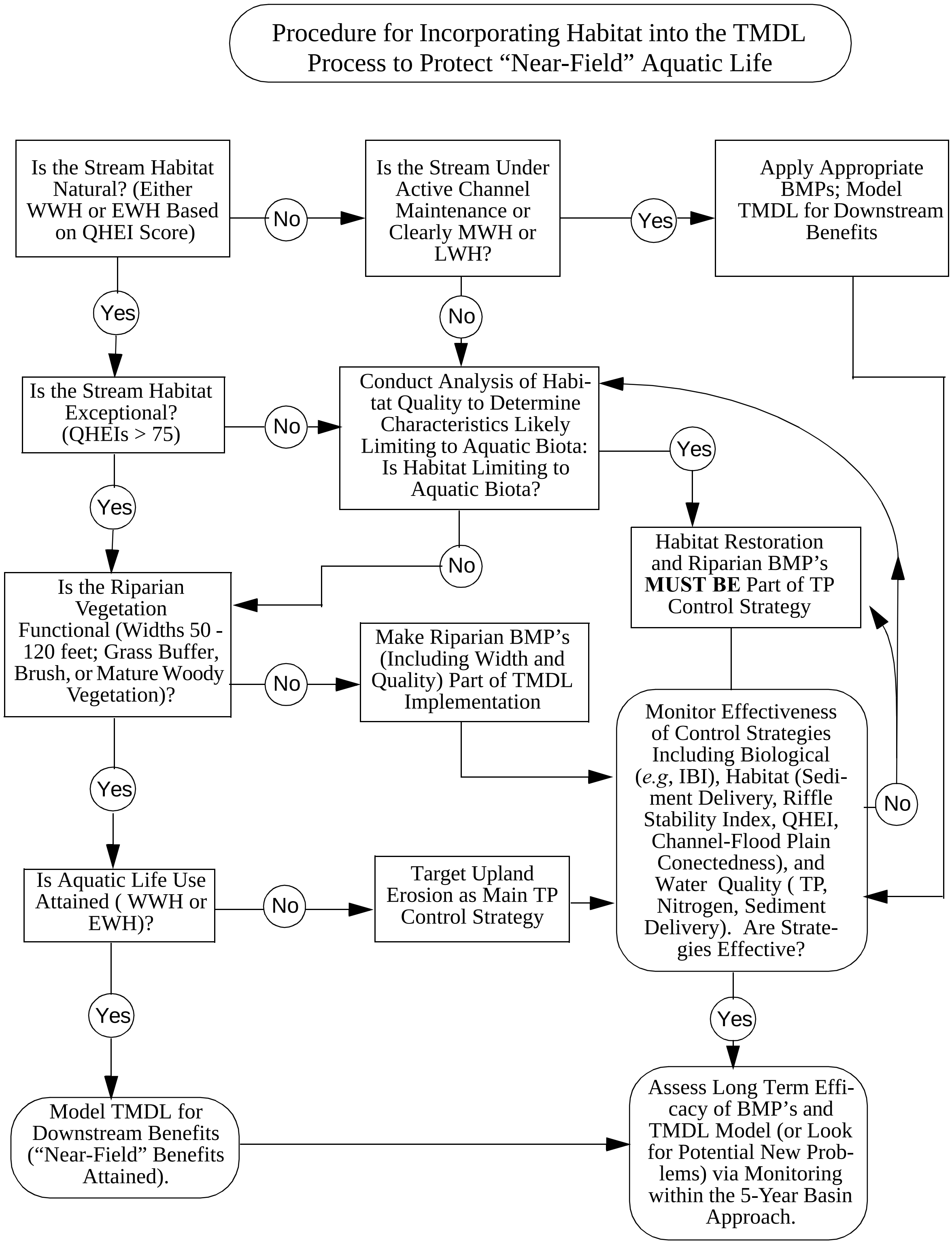


Figure 32. A flow chart summarizing how habitat and biosurvey data can be included in a TMDL process.

TMDLs for nonpoint sources have focused primarily on estimates of nutrient, sediment, and pesticide loadings and are most frequently based on estimates of gross erosion. BMPs are then designed to reduce the loadings of the parameter(s) of interest. The actual effectiveness of the load reduction efforts is generally measured in terms of BMP implementation and is not typically confirmed with ambient monitoring information, although examples do exist (Baker 1988). Implementation (and maintenance) of a BMP is frequently deemed sufficient evidence that restoration goals have been achieved. Restoration goals are inherently embodied in the aquatic life use designations of most state water quality standards. As has been amply demonstrated in Ohio (Ohio EPA 1990; Yoder and Rankin 1995b, Yoder and Rankin 1998), basing effectiveness solely on stressor indicators (i.e., loadings) may be misleading, especially when other stress factors (e.g., habitat) are present.

Information about biological and habitat quality is necessary to ensure a complete evaluation of BMP effectiveness. A flow chart suggesting how such information might be incorporated into the TMDL process appears in Figure 32. The first branch in this process deals with instream habitat. If instream habitat is largely natural and unmodified then the condition of the riparian zone is ascertained. The riparian zone needs to be sufficiently wide and intact to maintain instream habitat over time and provide for assimilative capacity. If this is lacking, then the TMDL should include strategies to protect or restore the riparian zone. If instream habitats are intact and aquatic life uses remain impaired (and point source discharges are not a factor) then upland BMPs should be a major focus. In cases where aquatic life uses are fully attained, then any controls or BMPs should be aimed at maintaining and preempting any emerging threats (*e.g.,* suburbanization). Such controls should be implemented in a proactive manner prior to major land use changes occurring in a watershed. Based on the work of previous investigators (Steedman 1988) the relationship between upland land use, the amount of impervious surfaces, and riparian zone quality must be accounted for in TMDL restoration and protection efforts.

Other branches in the flow chart pertain to situations where instream habitat is a principal limiting factor. If habitat degradation is reversable, then BMPs that either directly or indirectly result in restored instream habitat will be required. Decreasing pollutant loadings in such cases will not likely result in restoration of beneficial uses unless accompanied by instream habitat restoration. The important message here is that if aquatic life use attainment restoration and maintenance are goals of the TMDL process then instream and riparian habitat protection and restoration must be a part of this process. Also implicit in this approach is that upland BMPs will have few realized benefits if habitat restoration is not included. This should have profound consequences when evaluating local projects where extensive and essentially irretrievable habitat modifications exist.

Although site-specific issues have been treated separately from watershed scale issues for practical reasons, the two are inseparably linked. The restoration of natural function in headwater streams (see Figures 3 and 4) will have substantial positive benefits to downstream reaches. For example, streamside forest quality has been noted as the single-most important human controlled factor affecting the structure, function and water quality of streams entering coastal embayments (Sweeney 1992). Although the implementation of upland nutrient and erosion controls will yield positive benefits to downstream waters, the restoration of downstream function and productivity to an acceptable level (*e.g.,* lentic habitats) is partially dependent on the *aggregate* quality of headwater watersheds. Mainstem rivers (orders 5-6 and larger), lakes, reservoirs, and estuaries can be extensively impacted by the secondary effects of headwater modifications. For example, modification of both the headwaters and mainstem of the Missouri River has lead to an estimated 80% reduction in organic carbon from 1892 to 1982 (Hesse *et al.* 1988). The poor condition of most Lake Erie river mouths, embayments, harbors and nearshore areas (Thoma 1992), would substantially benefit from the ecological changes that would result from the sequestering of nutrients and sediment acheived by large-scale habitat and riparian restoration in the headwater streams of the parent watersheds. Such efforts would help to stabilize nutrient and sediment delivery to the lake, and to rehabilitate species dependant on riverine and wetland habitats for spawning (e.g., muskellunge and northern pike).

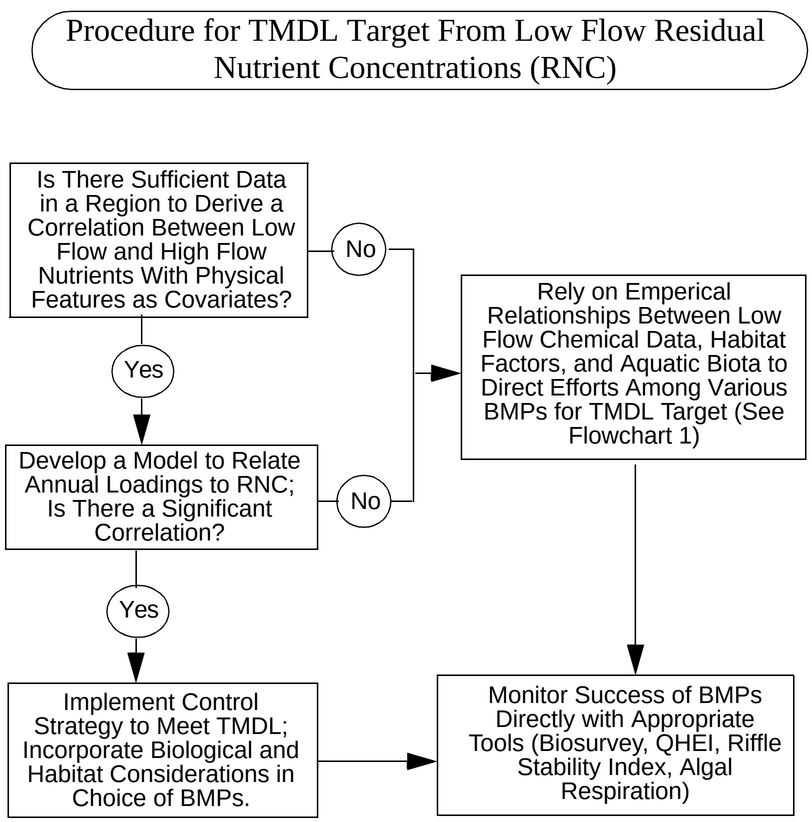


Figure 33. A flow chart summarizing how RNC and high flow estimates of total loads are related and can be incorporated into the TMDL process.

The use of habitat monitoring methods that target the characteristics most limiting to aquatic life, and that are responsive to changes in sediment loads, should be an important part of the TMDL process. In situations where habitat is not a principal limiting factor, biological data can be used to gauge the loading reductions necessary to restore and protect aquatic life if: 1) a sufficient database exists to determine correlations between residual nutrient concentrations (RNCs) and flow-based loadings, and 2) RNCs are sufficiently correlated with flow-related loadings (Figure 33). This project has also identified the factors that have a substantial influence on nutrient delivery, export, and instream processing that should be considered if RNC loading relationships are to be developed for application to wider areas. The principal factors that we have considered include watershed size, ecoregion, stream gradient, and habitat condition. Regional RNC load relationships will need to be stratified by these principal factors at a minimum. Other factors known to be important in affecting nutrient delivery to streams, such as soil type and land use (except as these are included in ecoregions), were not examined. These factors should be considered in any future work and will be more closely examined as subecoregions are delineated for Ohio.

In situations where the information base is insufficient to derive relationships between RNCs and total loadings, the empirical associations between the biota, habitat, and nutrient parameters illustrated here can provide insight into which BMPs are needed to restore and protect aquatic life uses (Figure 33). These are tools for the direct measurement of BMP effectiveness which is much preferable to relying solely on predictive indications of success. Even where the predicted reductions of nutrient or sediment loadings are accurate and achieved, non-chemical impacts may circumvent achieving aquatic life restoration and protection goals. Thus indicators that measure not only the traditional chemical and physical stresses, but indicators that measure the integrated result of the various factors of land use and instream processes will be needed.

Other Uses for the Regional Refernce Water Chemistry Statistics

The statistics summarized by this study have several potential uses for improving analyses conducted on ambient water quality and biological monitoring results. The most obvious use of these statistics is to compare monitoring results to background conditions for a stream or river of a similar drainage area in the same ecoregion. States routinely use water quality criteria as the benchmarks to determine whether a water quality problem exists. For nutrient parameters, such criteria are generally lacking, thus these types of "benchmarking" comparisons cannot be performed. One product of this study is a system for ranking the relative aquatic life risk of nutrient concentrations in grab samples (Table 6). This is based on the probability of departing from reference concentrations (REF, Appendix 1) and nutrient concentrations that are correlated with ICI or IBI scores (ALL, Appendix 2) consistent with the Warmwater Habitat or Exceptional Warmwater Habitat aquatic life use biocriteria. Six of the qualitative descriptors of risk associated with increasing deviations from the distribution of REF and ALL sites are as follows: 1) no risk; 2) minimal risk; 3) low risk; 4) moderate risk; 5) high risk; and, 6) extreme risk. In this "risk ranking" process the distribution of sampling results is compared to distributions of water chemistry at reference sites (REF) or to water chemistry distributions within ranges of the ICI and IBI (ALL data) that reflect minimally impacted conditions (WWH or EWH index ranges). Specifically, the median and 90th percentile sample values are compared to the median, 75th percentile, or two times the interquartile ranges of the REF or ALL sites. As illustrated in the last column of Table 6, this constitutes a comparison of the distribution of sample values to the reference distribution for the nutrient parameter. The more that the sample values deviate from the normal range (*i.e.*, the 90th or 95th percentile) of reference values, or ranges associated with aquatic life use attainment, the higher the probability that the sample site will not attain the criteria associated with the EWH or WWH aquatic life uses. In other words, from the empirical results, we rarely have found biological index values meeting WWH or EWH criteria associated with nutrient values deviating strongly from the normal reference distribution. By including associations between RNCs and the biological indices, this approach goes beyond efforts to characterize environmental conditions based solely on ranges of statistical deviation (*e.g.*, multiples of standard deviations) from known reference conditions (*e.g.*, Kelly and Hite 1984 for sediment chemistry) without consideration of biological associations.

Based on the associations between habitat, nutrients, and the aquatic communities, "management criteria" or goals can be derived for restoring aquatic life in streams. Table 7 summarizes some QHEI attribute management "criteria" that would be useful endpoints in a management scheme to restore or protect instream physical habitat. Ideally, the particular habitat components that appear to be the limiting factors can be selected as the primary management goals. Similar management criteria can be chosen for the nutrient parameters. Management options for total phosphorus can be based on the data in Appendix 1 and the "low risk levels" summarized in Table 6. Then, to derive management criteria to act as goals for restoring a warmwater stream that has the potential to be a "high quality" warmwater stream, we would first identify the habitat needing restoration or protection. Then we would set a realistic nutrient goal. Here, a realistic goal is derived by comparing existing instream concentrations to ranges in Appendix 1, or to suggested “criteria” in the Summary, and then setting a goal based on the median, 75th percentile or other range of the reference condition (*i.e.*, Appendix 1). For example if existing concentrations exceed the 90th percentlile of the reference range, a realistic goal may be the 75th percentile. An accumulation of these short-term objectives would be needed to acheive long-term watershed based restoration goals, so the process becomes iterative.

Table 9. QHEI attribute ìcriteriaî that can serve as management goals or endpoints for efforts to restore, enhance, or protect aquatic life in streams.

Attribute	"Criteria" WWH	EWH
Number of any Modified Attributes	≤4	≤2
High Influence Modified Attributes	≤1	0
Substrate Metric Scores	≥13	≥15
Substrate Embeddedness Score	≥3	4
Channel Metric Score	≥14	≥15
Overall QHEI	≥60	≥75

The decision making process outlined here is not intended as a substitute for the ambient biological monitoring necessary to characterize site-specific conditions, detect other types of impairment, and to examine trends or changes in biological integrity related to BMP implementation. Relying only on chemical data as a definitive assessment of aquatic life use attainment status constitutes the inappropriate substitution of an exposure indicator as a response indicator. However, this system can be a useful tool for interpreting ambient chemical *and* biological data to discern whether nutrients or other water chemistry parameters are having negative effects on aquatic life. In addition, such information can be incorporated into the TMDL/Load Allocation process where an empirical approach is used to select the appropriate BMPs. This process will be further enhanced by deriving RNC load relationships and linking these to ambient biological performance (Figure 33). It is important to remember that correlations between water quality and the biological indices may or may not be causative, especially in small and large rivers (drainage areas > 200 mi^2. Total phosphorus, for example, because of the propensity to enter the water attach to sediment particles may be a partial surrogate for sedimentation, which in turn may be the ultimate mechanism of biological impairment. Thus BMPs which reduce phosphorus delivery to streams may also reduce sediment delivery rates making total phosphorus a potentially useful indicator for sedimentation. However, this relationship need to be demonstrated emperically.

Application Examples: Twin Creek and Bokes Creek

An example of how the nutrient chemistry and habitat data may be used to aid in the interpretation of the biological results is illustrated with results from Twin Creek and Bokes Creek, both are wadeable streams located in the ECBP ecoregion. Twin Creek is a tributary of the Great Miami River located west of Dayton and flows through Darke, Preble, and Montgomery counties, all are dominiated by agricultural land use. Although there are localized impacts from point sources and agricultural activities, Twin Creek has excellent habitat quality, and largely meets EWH criteria.

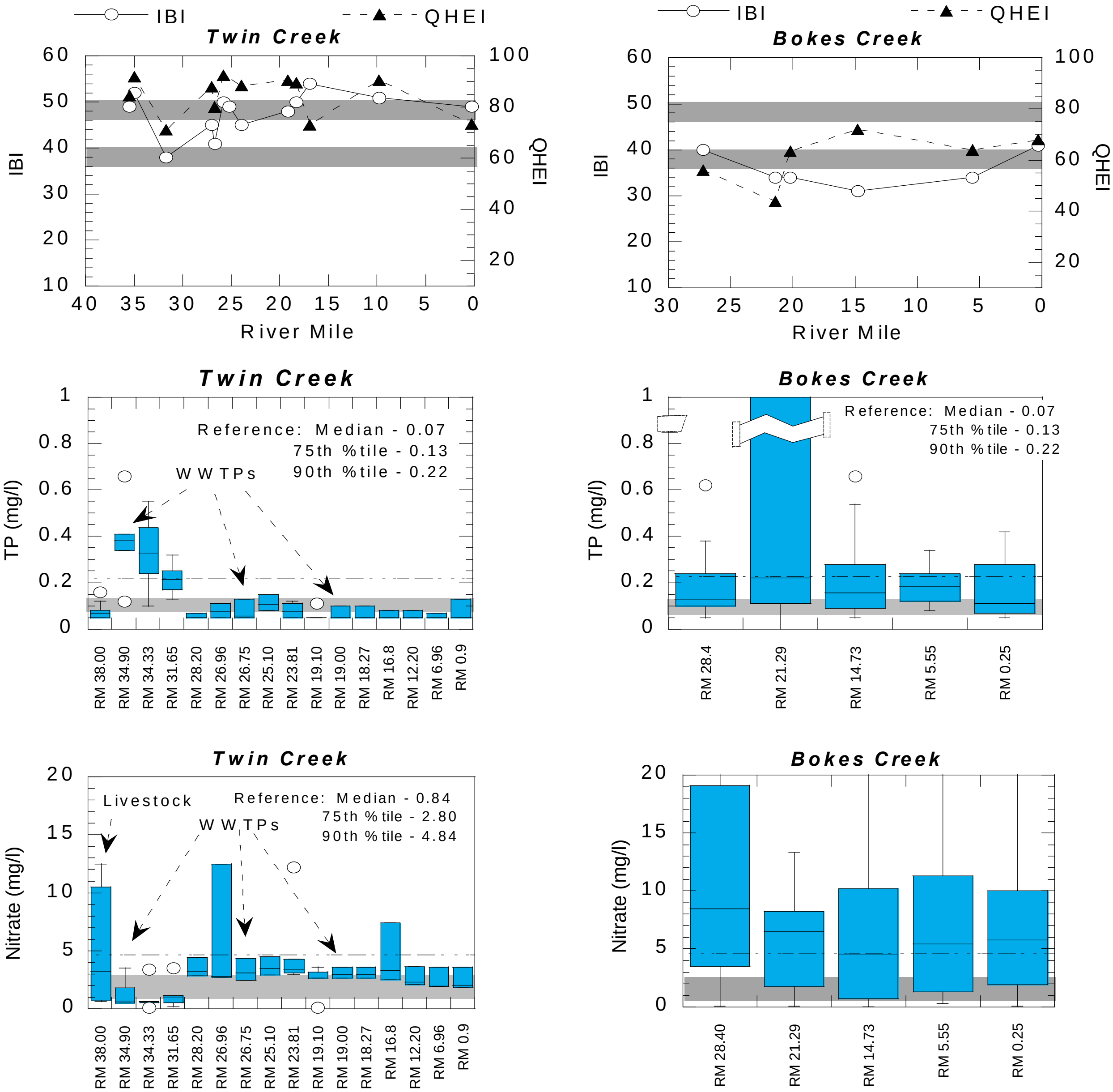
Bokes Creek is a tributary of the Scioto River and flows through Union, Logan, and Delaware counties, also having landscapes dominated by agriculture. Nutrient enrichment from row crop agriculture and poultry operations is of concern in the watershed. Because of the presence of these types of historical problems, this subbasin has been selected for testing the TMDL/LA process in an attempt to better manage nonpoint source loadings of nutrients. Biological performance ranges from good to poor and the results have recently been summarized (Ohio EPA 1991, 1994).

Figure 34 summarizes some pertinent monitoring data collected in these streams. The upper panels illustrate the longitudinal IBI and QHEI patterns for Twin Creek (left) and Bokes Creek (right). The biological communities found in Twin Creek generally attain the EWH use criteria and are superior to those found in Bokes Creek (Figure 34, upper left and right). Total phosphorus and nitrate concentrations are significantly higher in Bokes Creek than in Twin Creek (Figure 34, middle left and right). The Bokes Creek watershed has more intensive agriculture than does Twin Creek as well as large scale poultry operations with associated land treatment of manure. Although habitat quality is somewhat degraded in Bokes Creek (several sites are recovering from past channelization, riparian encroachment, and snagging; Ohio EPA 1994) the habitat quality is sufficient to support the WWH use. Ohio EPA (1991, 1994) attributed observed impairments to a combination of encroachment of intensive row crop agriculture on riparian and instream habitats and nutrient enrichment from land application of manure in the watershed.

Table 8 summarizes the biological, nutrient chemistry, and habitat data for the two streams with the darkness of shading in the table cells increasing with an increasing deviation from reference or biologically acceptable conditions. Bokes Creek had more sites deviating from the reference or biologically acceptable conditions for each parameter analyzed, including both nutrient chemistry and habitat parameters, than Twin Creek. Although Twin Creek had 4 of 6 sites deviating from reference conditions for nitrate-N, as was indicated earlier, the nitrate-N/biological index associations were much weaker than that observed for TP, especially for the EWH streams such as Twin Creek.

Although the water chemistry data are useful for screening watersheds for *potential* problems in relationship to reference or biologically acceptable conditions, the comparatively wide variability in the associations between nutrient parameters and biological performance compared to associations of the latter with habitat firmly indicates the need for the universal consideration of *both* attributes, at a minimum, when attempting to devise nonpoint *and* point source control strategies to protect or restore aquatic life. Full restoration of WWH biological performance in Bokes Creek will require the consideration of a combination of nutrient reductions *and* habitat restoration.

There were many high flow nitrate-N and TP values in the Bokes Creek database. There was little association between nitrate-N and flow; however, TP was positively correlated with flow (Figure 34, lower right). However, to adequately calibrate flow-concentration curves that can be translated into appropriate targets, flow-concentration relationships for least impacted watersheds should also be established. Such efforts need to incorporate those factors (discussed earlier) that affect nutrient dynamics and include stream gradient, habitat and riparian quality. In lieu of having these relationships better developed, the empirical approach (Figure 33) can be useful and cost-effective for deciding how to restore or protect aquatic life in Ohio's rivers and streams.



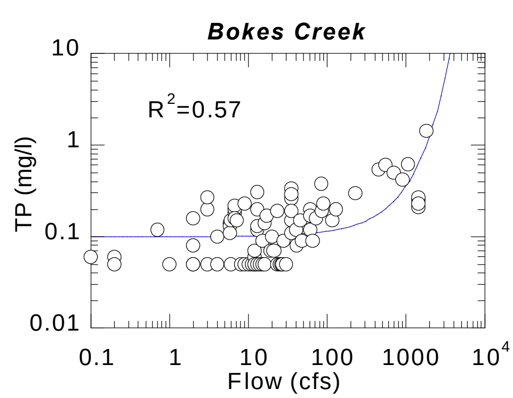


Figure 34. IBI, QHEI, and TP information for Twin Creek (left panels) and Bokes Creek (right panels) to illustrate associations between habitat, biology and nutrient chemistry outlined in this report. On the TP and N graphs, the shading represents the median and 75th percentiles and the horizontal line the 90th percentile of reference sites in the ECPB ecoregion for wadeable streams.

Little Miami River

Examination of the factors that are limiting biological communities in the Little Miami River also benefited from the analyses in this document. The approach outlined here was applied to data from the Little Miami River and East Fork Little Miami River (Ohio EPA 1995b). Both streams are designated as Exceptional Warmwater Habitat. In these rivers both % DELT anomalies and total phosphorus are higher than other EWH streams in the state (Figure 35). The % DELT anomalies were similar to levels observed in the Cuyahoga River in 1991. The pattern in this figure suggests a significant association between TP and % DELT anomalies and is also associated with cumulative effluent in these waters. The statewide examination of the relationship between TP and % DELT anomalies (see Figures 27-30) indicated that TP was most strongly correlated with DELT anomalies where: (1) point sources were the primary source of impairment, or 2) nonpoint source impacts were especially severe (*i.e.*, extreme nutrient enrichment and extensive channel modifications mimicked stresses associated with WWTPs).

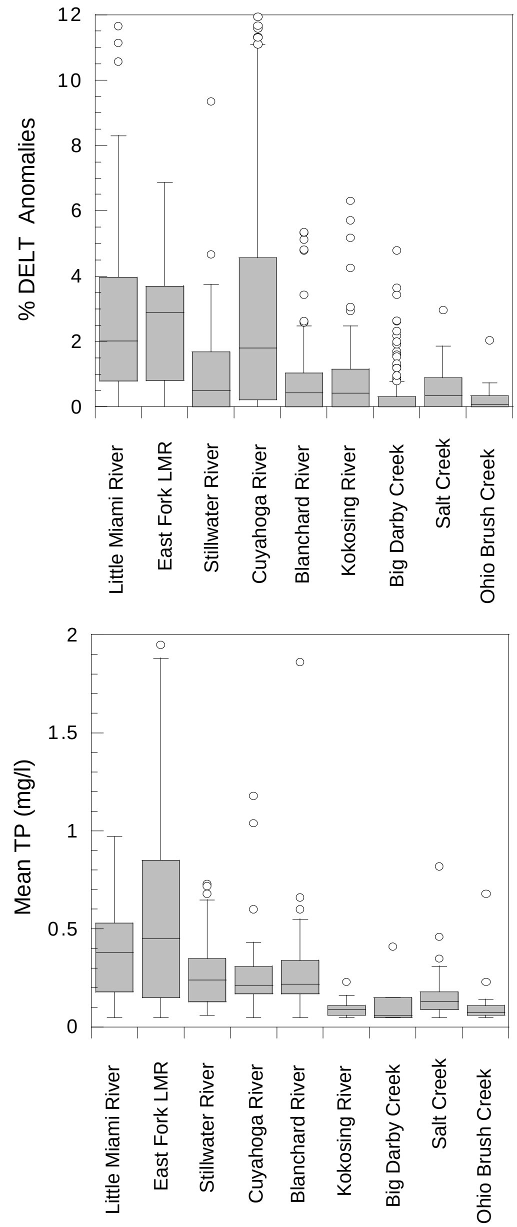


Figure 35. Distributions of percent DELT anomalies (top panel) and Mean TP (bottom panel) for nine Ohio rivers.

Figure 36 illustrates the concentration of TP by sample location in the upper Little Miami River (upper panel), lower Little Miami River (middle panel), and East Fork Little Miami River (lower panel). The background expectations for each river are portrayed by shading (median and 75th percentile) and dotted (90th percentile) or dashed (twice the interquartile range above the median). The qualitative risk of impairing the aquatic life use in each river based on a comparison of the ambient TP at each site with the reference TP is labeled vertically for each site. It is clear from Figures 36 and 37 that the concentrations of TP and TSS in certain portions of the Little Miami River are well above concentrations observed in reference streams in Ohio. EWH streams are particularly associated with low TP concentration and a low variance of values (*e.g.*, twice the IQR above the median is typically less than the 90th percentile). Ranges of TP concentrations given in the middle panel of Figure 36 are associated with deleterious ecological changes that occur in medium to large rivers when nutrient cycling is shorten, and production is driven by instream processing (autochthonous input). The East Fork of the Little Miami River (lower panel, Figure 36) has ranges of TP, except the upstream-most site at

RM 15.6, that are predominantly categorized as high or extreme values corresponding with impacts observed in this stretch (Ohio 1995b).

TSS concentrations closely mirror TP concentrations in these rivers (Figures 36 and 37). TP is often delivered to streams attached to solids particles such as sediment or suspended particles in WWTP plant discharges. The mean concentrations of TSS and TP are positively related on a logarithmic scale in the Little Miami River, a common pattern. There is a strong association between increasing TP concentrations and decreasing biological condition in streams across Ohio, especially in EWH streams. The Little Miami River shows a similar trend. Clearly, some factor associated with elevated TP and TSS is the causative factor for this pattern. In streams where DELT anomalies and TP are high, D.O. is often low, and bacterial levels of human or livestock origin are often elevated. Because of such co-varying factors, approaches to reducing risks to aquatic life from point source discharges will need to focus on strategies that reduce TSS and TP, and protect streams from increasing nonpoint threats though erosion control, habitat protection and restoration, and riparian protection and restoration.

Future Research

Other landscape factors, specifically soil type and glacial features, were originally planned to be investigated in this study. However, lack of sufficiently accurate information prompted a postponement of these analyses. A GIS based approached is needed to make data easier to obtain and manipulate. The pattern of nitrate-N concentrations observed in the Eastern Corn Belt Plains ecoregion suggests that such subregional analyses are warranted. After the completion of the delineation of subecoregions for Ohio, the relationships between nutrient loadings, habitat, and biological performance presented here can be reexamined in this new spatial framework.

A critical need for the success of watershed and nonpoint source management efforts is the development of relationships between land use and riparian zone quality similar to that developed by Steedman (1988) for Toronto urban streams, for Michigan streams developed by Allan *et al*. (1997), Johnson *et al*. (1997) and Richards *et al*. (1995). A sufficient instream database exists throughout Ohio for biological performance, habitat, and water quality information. However, usable information about riparian zone quantity and quality is lacking and the development of suitable surrogates is being explored. The goal of such an investigation would be to produce estimates of riparian zone quality and quantity relative to land use throughout the agricultural region of Ohio needed to consistently achieve and maintain the biological criteria for the WWH and EWH use designations. One clear need is to develop a relationship between the degree of the riparian encroachment and modification in small streams and impairment of the WWH or EWH criteria in higher order streams and the major mainstem rivers. Such a relationship could have a major effect on the direction of NPS management efforts to protect and restore Ohio's rivers.

Another area of needed research is the relationship between phosphorus, and algal abundance and species composition over similar spatial scales entertained in this study. The composition of algal communities in lentic environments is influenced by both the concentration of TP and nitrogen, and the ratios of nitrogen to phosphorus (Hecky and Kilham 1988). Although the dynamics of nutrient limitation are not similar between lotic and lentic environments (Bothwell 1989), nutrient limitation of periphytic communities does occur in rivers and streams (Wu et al. 1996), a phosphorus-chlorophyll *a* relationship exists for rivers and streams (Van Nieuwenhuyse and Jones 1996), and algal composition is influenced by changes in nutrient concentrations and ratios (Mulholland et al. 1995). Nutrient enrichment tends to shift the algal community dominance away from diatoms toward filamentous and bluegreen algae. Based on what we now know, a reduction in nitrogen concentrations in the absence of phosphorus reduction could stimulate production of blue-greens. Consequently, the target nutrient "criteria" tabulated in the Summary section fortuitously reflect N:P ratios $\geq$ 10.

Recent modifications to fluvial geomorphological characterization of stream types (Rosgen 1994) that incorporate stream gradient, stream morphology, and stream substrate characters, may allow an improved targeting of those streams that are likely to be the most sensitive to increased nutrient loadings (*e.g.,* low gradient, alluvial, meandering streams) or those that may be insensitive to the same loadings (*e.g.,* high gradient, high flow, headwater streams). The Rosgen stream classification system could also prove useful in explaining intra-ecoregion variation in habitat and biological quality in streams.

Although many details regarding the ecology of stream systems need to be researched, this paper and a large volume of research elseware (U. S. National Research Council 1992) have established the immediate need to institute aquatic habitat and riparian corridor restoration. The National Academy of Sciences has called for the restoration of 400,000 miles of streams and rivers over the next 20 years and they emphasize that physical restoration of riparian zones and restoration of river hydrology (fluctuating flows) need to be given priority if successful restoration is to occur (U. S. National Research Council 1992). Perhaps Ohio needs to develop a similar, measurable goal for its waters.

Table 10. Application of ecological risk factors associated with nutrient chemistry and habitat data in Bokes Creek and Twin Creek.

Location	Biological Data		Water Column Chemisty (Median/90th %tile)			Selected Habitat Parameters				
RM	IBI/ICI	Attainment Status	TP	NO_3-N	NH4	Riparian Metric	Embedd-edness	Substrate Metric	Channel Metric	MWH Attributes
Bokes Creek (1992)										
27.2/26.2	32/44	Partial	0.13 (0.24)	8.45 (19.10)	0.05 (0.14)	5.5	1.5	9	9	7
21.3/21.4	29/50	Partial	0.28 (1.27)	6.87 (8.21)	0.14 (0.40)	4	1.5	13.5	12.5	9
20.2/20.5	29/34	Non	–	–	–	5	1	4.5	10.5	6
14.8/14.8	37/40	Partial	0.14 (0.28)	6.91 (11.1)	0.05 (0.22)	6.5	3	17	17	2
5.6/5.6	37/42	Partial	0.18 (0.24)	7.21 (11.3)	0.07 (0.10)	5	3	16.5	14	2
0.3	49/44	Full	0.11 (0.28)	5.79 (10.0)	0.05 (0.06)	6	2	14.5	14.5	0
Twin Creek (1986)										
35.5/35.8	49/44	Full	0.05 (0.09)	0.60 (1.95)	0.05 (0.05)	4	3	19	19	0
35.0/35.0	52/34	Partial	0.39 (0.41)	0.51 (2.02)	0.71 (0.90)	9	3	18	19	0
31.7/31.7	38/44	Partial	0.22 (0.29)	1.11 (2.32)	0.05 (0.05)	4	2	14	14	2
27.0/28.5	45/50	Partial	0.05 (0.07)	3.24 (4.00)	0.05 (0.05)	8	3	16	19	0
26.7/–	41/-	Non	0.06 (0.10)	3.12 (3.81)	0.05 (0.05)	9	3	17	19	1
25.8/25.8	50/48	Full	0.11 (0.13)	3.48 (4.09)	0.05 (0.05)	8.5	4	20	19	0
23.9/23.9	45/50	Partial	0.05 (0.09)	3.33 (3.89)	0.05 (0.05)	8.5	3	20	19	0
18.3/18.2	50/40	Partial	0.05 (0.08)	2.97 (3.32)	0.05 (0.05)	10	3	17	19	0
16.9/16.6	54/50	Full	0.05 (0.07)	3.30 (5.58)	0.05 (0.05)	7	2	14	16	0
Shading Criteria										
			Table 4 Appendix 1	Table 4 Appendix 2	Table 4 Appendix 4	> 7	> 2	18-20	19-20	0-1
						6-7	2	15-17	16-18	2-3
Overall qualitative ratings for parameters and relation to shading:		Excellent	Good	Fair	Poor	4-5	< 2	11-14	12-15	4-6
Ecological Risk Ratings from Table 6		Ratings: 1-3	Ratings: 4-5	Ratings: 6-7	Ratings: 8-9	<4		≤ 10	≤ 11	≥ 7

Reference Statistics from Table 6 for IBI 40-49, ECBP, Wadeable Streams: Total Phosphorus - Median = 0.07; 75th %tile=0.24; Med + 2 IQR = 0.41
Nitrate - Median = 1.27; 75th %tile = 3.00; Med + 2 IQR = 4.89
Total Ammonia - Median = 0.05; 75th %tile = 0.07; Med + 2 IQR = 0.09

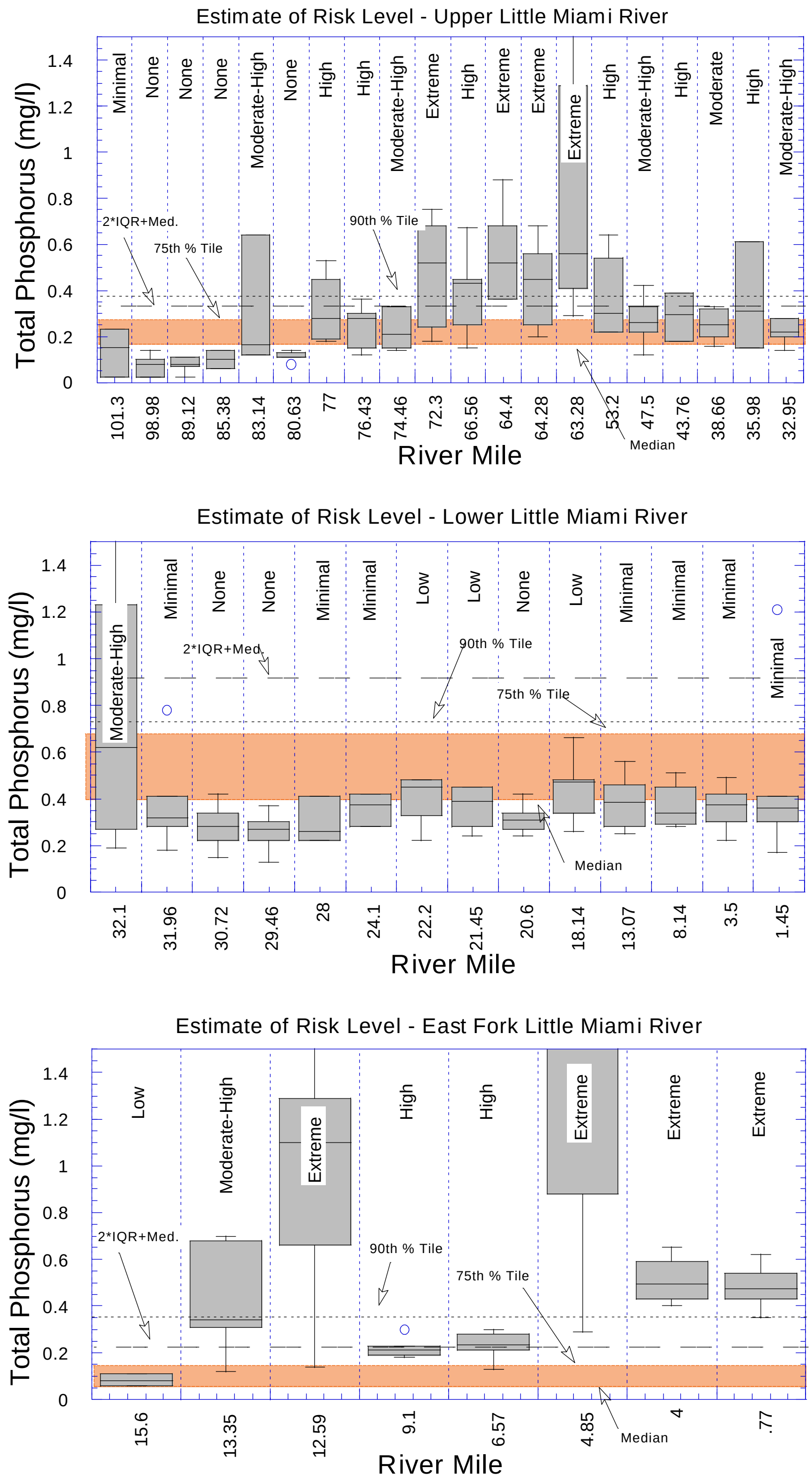


Figure 36. Total phosphorus versus river mile for the upper Little Miami River (top), lower Little Miami River (middle), and East Fork Little Miami River (bottom). Shading represents the median to 75th percentiles of reference streams, the dotted line the 90th percentile, and the dashed line two times the interquartile range above the median. Narrative risk descriptions refer to the risk of impairment to the IBI based on deviations of TP sample medians and 75th percentiles from refernce conditions.

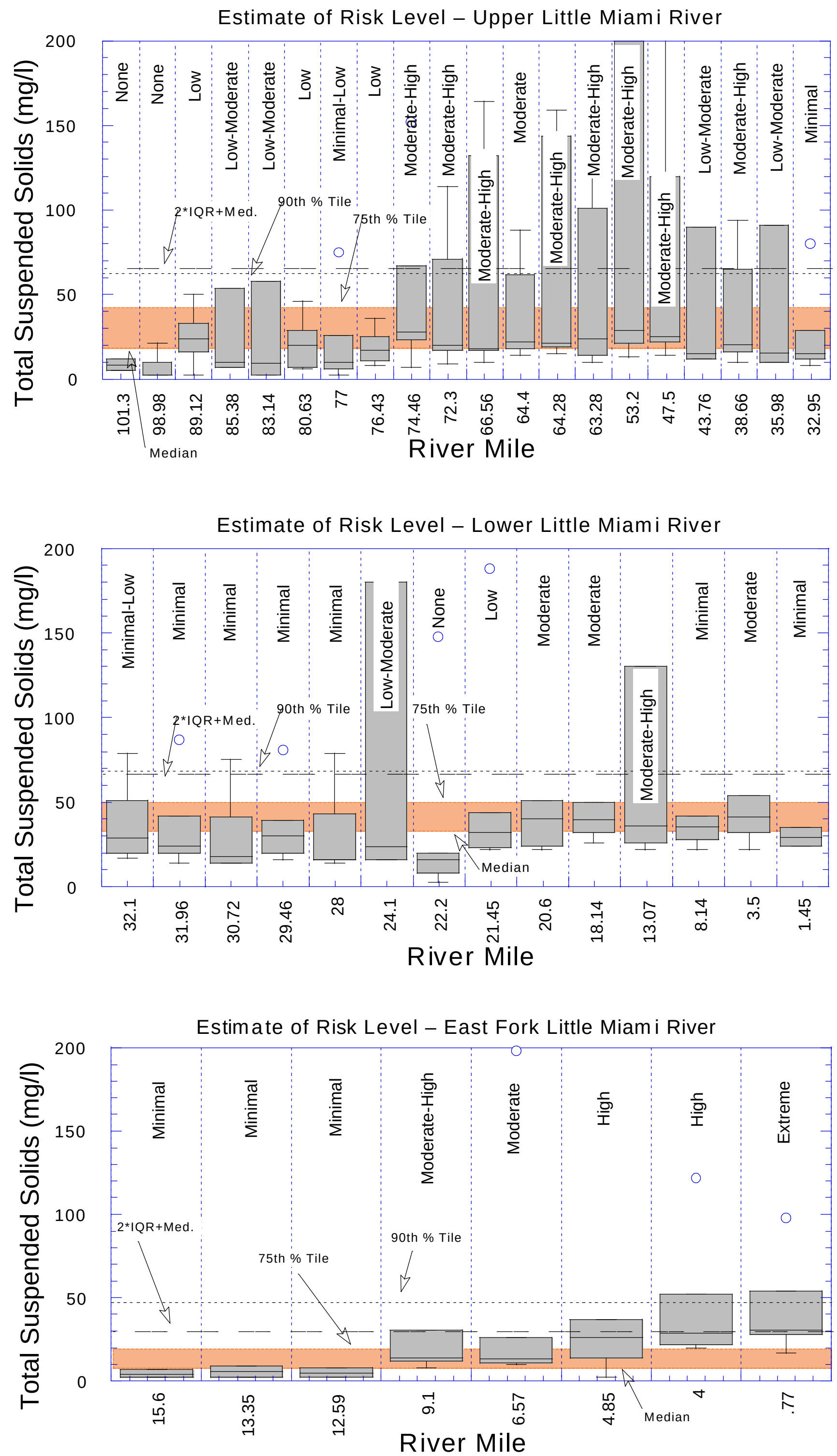


Figure 37. Total suspended solids versus river mile for the upper Little Miami River (top), lower Little Miami River (middle), and East Fork Little Miami River (bottom). Shading represents the median to 75th percentiles of reference streams, the dotted line the 90th percentile, and the dashed line two times the interquartile range above the median. Narrative risk descriptions refer to the risk of impairment to the IBI based on deviations of TP sample medians and 75th percentiles from refernce conditions.

References

Allan, J. D., D. L. Erickson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. Freshwater Biology 37: 149-161.

Baker, D. B. 1985. Regional water quality impacts of intensive row-crop agriculture: A Lake Erie basin case study. J. Soil Water Cons. 40: 125-132.

Baker, D. B. 1988. Sediment, nutrient, and pesticide transport in selected lower Great Lake tributaries. USEPA, Region V, Great Lakes National Program Office. EPA-905/4-88-001 (GLNPO Report No. 1).

Barling, R. D. and Moore, I. D. 1994. Role of buffer strips in management of waterway pollution; A review. Env. Mgmt. 18(4): 543-558.

Benke, A. C., Hall, C. A. S., Hawkins, C. P., Lowe-McConnell, R. H., Stanford, J. A., Suberkropp, K., and J. V. Ward. 1988. Bioenergetic considerations in the analysis of stream ecosystems. J. N. Am. Benth. Soc. 7(4): 480-502.

Bilby, R. E. and G. E. Likens. 1980. Importance of organic debris dams in the structure and function of stream ecosystems. Ecology 61: 1107-1113.

Bowden, W. B. 1987. The biogeochemistry of nitrogen in freshwater wetlands. Biogeochemistry 4:313-348.

Brady, N. C. 1990. The nature and properties of soils. Tenth Edition. Macmillan Publishing Co., New York.

Chichester, F. W., and C. W. Richardson. 1992. Sediment and nutrient loss from clay soils as affected by tillage. Journal of Environmental Quality 21(4): 587-590.

Cooper, J. R., Gilliam, J. W., Daniels, R. B., and W. P. Robarge. 1987. Riparian areas as filters for agricultural sediment. Soil Sci. Soc. Am. 51: 416-420.

Cummins, K. W. 1974. Structure and function of stream ecosystems. BioScience 24: 631-641.

DeShon, J. E. 1995. Development and application of an invertebrate community index (ICI), pp. 217-244. *in* Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making, Davis, W.S. and Simon, T.P. (eds.). Lewis Publishers, Boca Raton, FL.

Dodd, W. K, Jones, J. R. and E. B. Welch. 1998. Suggested classification of stream trophic state: Distributions of temperate stream types by chlorophyll, total nitrogen, and phosphorus. Wat. Res. 32: 1455-1462.

Dolan, D. M. 1993. Point source loadings of phosphorus to Lake Erie: 1986-1990.

Fausch, D.O., Karr, J.R. and P.R. Yant. 1984. Regional application of an index of biotic integrity based on stream fish communities. Trans. Am. Fish. Soc. 113: 39-55.

Fennesy, M. S. and J. K. Cronk. 1997. The effectiveness and restoration potential of riparian ecotones for the management of nonpoint source pollution, particularly nitrate. Critical Reviews in Environmental Science and Technology 27(4): 285-317.

Gallant, A. L., T. R. Whittier, D. P. Larsen, J. M. Omernik and R. M. Hughes. 1989. Regionalization as a Tool for Managing Environmental Resources. EPA-600-3-89-060. U. S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon.

Gammon, J. R. 1995. An environmental assessment of the streams of Putnam County, Indiana and vicinity with special emphasis on the effects of animal feedlots. A report to Heritage Environmental Services, Indianapolis, Indiana. 124 pp.

Gorman, O. T. and J. R. Karr. 1978. Habitat structure and stream fish communities. Ecology 59: 507-515.

Gregory, S. V., F. J. Swanson, W. A. McKee and K. W. Cummins. 1991. An ecosystem perspective of riparian zones. Bioscience 41: 540-551.

Grimm, N. B., 1988. Role of macroinvertebrates in nitrogen dynamics of a desert stream. Ecology 69: 1884-1893.

Hesse, L. W., Wolfe, C. W., and N. K. Cole. 1988. Some aspects of energy flow in the Missouri River ecosystem and a rationale for recovery. Pages 13-28 *in* N. G. Benson (editor). The Missouri River: the resources, their uses and values. Northcentral Division, American Fishery Society, Special Publication No. 8.

Johnson, L. B., C. Richards, and G. E. Host. 1997. Landscape influences on water chemistry in Midwestern stream ecosystems. Freshwater Biology 37: 193-208.

Kappesser, G. B. 1994. Riffle Stability Index: A procedure to evaluate stream reach and watershed equilibrium. U.S. Department of Agriculture, Forest Service, Jefferson National Forest, Roanoke, Virginia.

Kaushik, N. K., Robinson, J. B., Stammers, W. N., and H. R. Whitely. 1981. Aspects of nitrogen transport and transformation in headwater streams. Pages 113-139 *In*: Perspectives of Running Water Ecology, Louck, M. A. and Williams, D. D., editors. Plenum Press.

Karr, J.R. 1995. Protecting aquatic ecosystems: clean water is not enough, pp. 7-14. *in* Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making, Davis, W.S. and Simon, T.P. (eds.). Lewis Publishers, Boca Raton, FL.

Karr, J. R. 1991. Biological integrity: a long-neglected aspect of water resource management.

Ecological Applications 1: 66-84.

Karr, J.R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6(6): 21-27.

Karr, J. R. and I. J. Schlosser. 1977. Impact of nearstream vegetation and stream morphology on water quality and stream biota. EPA-600-3-77-097. U. S. EPA, Environmental Research Laboratory, Athens, Georgia.

Karr, J. R. and I. J. Schlosser. 1978. Water resources and the land-water interface. Science 201: 229-234.

Karr, J. R. and D. R. Dudley. 1981. Ecological perspective on water quality goals. Env. Mgmt. 5: 55-68.

Karr, J. R., Toth, L. A., and G. A. Garman. 1983. Habitat preservation for midwest stream fishes: principles and guidelines. USEPA, Office of Research and Development, Corvallis, Oregon EPA-600/3-83-006.

Kelly, M. H. and R. L. Hite. 1984. Evlauation of Illinois stream sediment data: 1974-1980. Illinois Environmental Protection Agency, Division of Water Pollution Control.

Klotz, R. L. 1988. Sediment control of soluble reactive phosphorus in Hoxie Gorge Creek , New York. Canadian Journal of Fisheries and Aquatic Sciences 45: 2026-2034.

Larsen, D. P., Omernik, J. M., Hughes, R. M., Rohm, C. M., Whittier, T. R., Kinney, A. J., Gallant, A. L., and D. R. Dudley. 1986. Correspondence between spatial patterns in fish assemblages in Ohio streams and aquatic ecoregions. Env. Mgmt. 10: 815-828.

Lamberti, G. A. and V. H. Resh. 1983. Stream periphyton and insect herbivores: an experimental study of grazing by a caddisfly population. Ecology 64(5): 1124-1135.

Leopold, L. B., M. G. Wolman and J. P. Miller. 1964. Fluvial processes in geomorphology. W. H. Freeman and Company, San Francisco, California..

Lowrance, R. R., R. Todel, J. Fail, O. Hendrickson Jr., O., R. Leonard, and L. Asmussen. 1984. Riparian forests as nutrient filters in agricultural watersheds. BioScience 34: 374-377.

Lowrance, R, Altier, L. S., Newbold, J. D., Schnabel, R. R., Groffman, P. M., Denver, J. M., Correll, D. L., Gilliam, J. W., Robinson, J. L., Brinsfield, R. B., Staver, K. W., Lucas, W., and A. H. Todd. 1995. Water quality functions of riparian forest buffer systems in the Chesapeake Bay Watershed. USEPA, Nutrient Subcommittee of the Chesapeake Bay Program. EPA 903-R-95-004.

Malanson, G. P. 1993. Riparian landscapes. Cambridge University Press, Cambridge, Great

Brittain.

Meyer, J. L., McDowell, W. H., Bott, T. L., Elwood, J. W., Ishizaki, C., Melack, J. M., Peckarsky, B. L., Peterson, B. J., and R. A. Rublee. 1988. Elemental dynamics in streams. J. N. Am. Benth. Soc. 7(4): 410-432.

Miltner, R. J. and E. T. Rankin. 1998. Primary nutrients and the biotic integrity of rivers and streams. Freshwater Biology 40: 145-158.

Minshall, G. W., Peterson, Cummins, K. W., Bott, T. L., Sedell, J. R., Cushing, C. E., and R. L. Vannote. 1983. Interbiome comparison of stream ecosystem dynamics. Ecological Monographs 53: 1-25.

Minshall, G. W., K. W. Cummins, R. C. Peterson, C. E. Cushing, Bruns, D. A., Sedell, J. R., and R. L. Vannote. 1985. Developments in stream ecosystem theory. Can. J. Fish. Aquat. Sci. 42: 1045-1055.

Minshall, G. W. 1988. Stream ecosystem theory: a global perspective. J. N. Am. Benth. Soc. 7(4): 263-288.

Morisawa, M. 1968. Streams, their dynamics and morphology. McGraw-Hill, New York.

Naiman, R. J., Decamps, H., Pastor, J., and C. A. Johnston. 1988. The potential importance of boundaries to fluvial ecosystems. J. N. Am. Benth. Soc. 7(4): 289-306.

Naiman, R. J., Decamps, and M. Pollock. 1993. The role of riparian corridors in maintaining regional biodiversity. Ecological Applications 3(2): 209-212.

Newberry, D. G. S. 1992. Management of urban riparian systems for nitrate reduction. Prepared for the United States Environmental Protection Agency, Region V, Chicago, IL

Newbold, J. D., Elwood, J. W., O'Neill, R. V., and A. L. Sheldon. 1983. Phosphorus dynamics in a woodland stream ecosystem: a study of nutrient spiraling. Ecology 64: 1249-1265.

Nototny, V. and G. Chesters. 1981. Handbook on Nonpoint Pollution: Sources and Management. Van Nostrand Reinhold Co., New York, New York.

Odum, E. P. 1971. Basic Ecology. Saunders College Publications, Philadelphia, PA.

Ohio Environmental Protection Agency. 1987a. Biological criteria for the protection of aquatic life: Volume I. The role of biological data in water quality assessment. Division of Water Quality Monitoring and Assessment, Surface Water Section, Columbus, Ohio 43266.

Ohio Environmental Protection Agency. 1987b. Biological criteria for the protection of aquatic life: Volume II. Users manual for biological field assessment of Ohio surface waters.

Division of Water Quality Monitoring and Assessment, Surface Water Section, Columbus, Ohio 43266.

Ohio Environmental Protection Agency. 1989a. Biological criteria for the protection of aquatic life: Volume III. Standardized biological field sampling and laboratory methods for assessing fish and macroinvertebrate communities. Division of Water Quality Monitoring and Assessment, Columbus, Ohio 43266.

Ohio Environmental Protection Agency. 1989b. Addendum to biological criteria for the protection of aquatic life: Volume II. Users manual for biological field assessment of Ohio surface waters. Division of Water Quality Monitoring and Assessment, Surface Water Section, Columbus, Ohio 43266.

Ohio Environmental Protection Agency. 1991a. Ohio EPA manual of surveillance methods and quality assurance practices. Ohio EPA, Division of Environmental Services, Columbus, Ohio 43210.

Ohio Environmental Protection Agency. 1991b. Biological and water quality study of Mill Creek and selected tributaries and Bokes Creek. Ohio EPA, Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, Ohio 43266.

Ohio Environmental Protection Agency. 1992. Ohio water resource inventory. Volume I: Summary, status, and trends. E. Rankin, C. Yoder, and D. Mishne, editors. Ohio Environmental Protection Agency, Division of Water Quality Monitoring and Assessment, Columbus, Ohio 43266.

Ohio Environmental Protection Agency. 1994. Biological and habitat study of Bokes Creek, Union and Delaware Counties, Ohio. OEPA Tech. Rept. No. EAS/1994-1-2. Ohio EPA, Division of Surface Water, Ecological Assessment Section, Columbus, Ohio 43266.

Ohio Environmental Protection Agency. 1995. Biological and water quality study of Little Miami River and selected tributaries, Clarke, Greene, Montgomery, Warren, Clermont, and Hamilton Counties, Ohio. Volume I. OEPA Tech. Rept. No. MAS/1994-12-11. Ohio EPA, Division of Surface Water, Monitoring and Assessment Section, Columbus, Ohio 43266.

Omernik, J.M. and A.L. Gallant. 1988. Ecoregions of the upper midwest states, EPA/600/3-88/037, U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon.

Phillips, J. D. 1989. Fluvial sediment storage in wetlands. Water Resources Bulletin 25: 867-873.

Platts, W. S., W. F. Megahan, and G. W. Minshall. 1983. Methods for evaluating stream, riparian, and biotic conditions. General Technical Report No. INT-138, U. S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station, Ogden, Utah.

Peterjohn, W. T., and D. C. Correll. 1984. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. Ecology 65: 1466-1475.

Pulliam, H. R. 1988. Sources, sinks, and population regulation. American Naturalist 132: 652-661.

Rabeni, C. F. and M. A. Smale. 1995. Effects of siltation on stream fishes and the potential mitigating role of the buffering riparian zone. Hydrobiologia 303: 211-219.

Rankin, E. T. 1989. The qualitative habitat evaluation index (QHEI), rationale, methods, and application. Ohio Environmental Protection Agency, Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, Ohio.

Rankin, E. T. 1995. The use of habitat indices in water resource quality assessments, pp. 181-208. *in* Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making, Davis, W.S. and Simon, T.P. (eds.), Lewis Publishers, Boca Raton, FL.

Richards, C., L. B. Johnson, and G. E. Host. 1996. Landscape-scale influences on stream habitats and biota. Canadian Journal of Fisheries and Aquatic Sciences 53: 295-311.

Robinson, J. S., A. N. Sharpley, and S. J. Smith. 1992. Estimating bioavailable phosphorus loss in agricultural runoff: Method development and application. Pages 375 - 385 in Water Environment Federation 65th Annual Conference & Exposition, 20-24 September 1992, New Orleans, LA.

Rohlich, G. A., and D. J. O'Connor. 1980. Phosphorus management for the Great Lakes. Final Report Phosphorus Management Strategies Task Force. PLUARG Technical Report of the International Joint Commission, Windsor, Onatario, Canada

Rosgen, D. L. 1994. A classification of natural rivers. Catena 22(1994): 169-199.

Schlosser, I. J. 1995. Critical landscape attributes that influence fish population dynamics in headwater streams. Hydrobiologia 303: 71-81.

Sharpley, A. N., Chapra, S. C., Wedepohl, R., Sim, J. T., Daniel, T. C. and K. R. Reddy. 1994. Managing agricultural phosphorus for protection of surface waters: Issues and options. J.Env. Qual. 23: 437-451.

Statzner, B., Gore, J. A., and V. H. Resh. 1988. Hydraulic stream ecology: observed patterns and potential applications. J. N. Am. Benth. Soc. 7(4): 289-306.

Steedman, R. J. 1988. Modification and assessment of an index of biotic integrity to quantify stream quality in southern Ontario. Canadian Journal of Fisheries and Aquatic Sciences 45: 492-501.

Stein, R. A., D. R. DeVries, and J. M. Dettmers. 1995. Food-web regulation by a planktivore: exploring the generality of the trophic cascade hypothosis. Canadian Journal of Fisheries and Aquatic Sciences 52: 2518-2526.

Sweeney, B. W. 1992. Streamside forests and the physical, chemical, and trophic characteristics of piedmont streams in eastern North America. Water Sci. Tech. 26(12): 2653-2673.

Terrell, J. W., Cade, B. S., Carpenter, J. and J. M. Thompson. 1996. Modeling stream fish habitat limitations from wedge-shaped patterns of variation in standing stock. Transactions of the American Fishery Society 125: 104-117.

Thoma, R. F. 1990. A preliminary assessment of Ohio's Lake Erie estuarine fish communities. Ohio Environmental Protection Agency, Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, Ohio 43266.

USEPA. 1991. Guidance for water quality-based decisions: The TMDL process. USEPA, Office of Water, Washington, D.C., EPA 440/4-91-001.

USEPA 1976. Quality criteria for water. USEPA, Washington, D. C.

U. S. National Research Council. 1992. Restoration of aquatic ecosystems: science, technology, and public policy. National Academy Press, Washington, DC.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell and C. E. Cushing. 1980. The river continuum concept. Can. J. Fish. Aquatic Sci. 37: 130-137.

Wallace, J. B. and D. H. Ross. 1982. Seston and dissolved organic carbon dynamics in a southern Appalachian stream. Ecology 63(3): 824-838.

Wallace, J. B., Cuffney, T. F., Webster, J. R., Lugthart, G. J., Chung, K., and B. S. Goldowitz. 1991. Export of fine organic particles from headwater streams: effects of season, extreme discharges, and invertebrate manipulation. Limnol. Oceanog. 36(4): 670-682.

Watson, V. J., Loucks, O. L., Mitchell, J. and N. L. Clesceri. 1979. Impact of development on watershed hydrologic and nutrient budgets. J. Water Poll. Contr. Fed. 51: 2876-2885.

Ward, J. V. 1989. The dour-dimensional nature of lotic ecosystems. J. N. Am. Benth. Soc. 8: 2-8.

Whittier, T. R., Larsen, D. P., Hughes, R. M., Rohm, C. M., Gallant, A. L., and J. M. Omernik. 1987. The Ohio stream regionalization project: a compendium of results. USEPA, Environmental Research Laboratory, Corvallis, OR EPA/600/3-87/025.

Whittier, T. R., Hughes, R. M. and Larsen, D. P. 1988. The correspondence between ecoregions and spatial patterns in stream ecosystems in Oregon. Can. J. Fish. Aquatic Sci. 45: 1264-

1278.

Yoder, C.O. and E.T. Rankin. 1995a. Biological criteria program development and implementation in Ohio, pp. 109-144. *in* Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making, Davis, W.S. and Simon, T.P. (eds.). Lewis Publishers, Boca Raton, FL.

Yoder, C.O. and E.T. Rankin. 1995b. Biological response signatures and the area of degradation value: new tools for interpreting multimetric data , pp. 263-286. *in* Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making, Davis, W.S. and Simon, T.P. (eds.). Lewis Publishers, Boca Raton, FL.

Yoder, C. O. 1991. The integrated biosurvey as a tool for evaluation of aquatic life use attainment and impairment in Ohio surface waters. Pages 110-122 in Biological criteria, research and regulation, proceedings of a symposium, 12-13 December 1990, Arlington, Virginia. EPA-440-5-91-005.U. S. EPA, Office of Water, Washington, D.C.

Glossary

"ALL"- Sites in Ohio EPA's databases where both biological and water chemistry data were collected during the same summer period. Unlike REF sites that meet the criteria to be classified as relatively unimpacted reference sites, ALL sites includeunimpaired as well as heavily polluted sites.

Aquatic Life Use - A designation assigned to a waterbody in Ohio based on the *potential* aquatic life that the water can sustain given the ecoregion potential; (See EWH, WWH, CWH, LRW, Designated use).

Aquatic Life Use Attainment - Defined as the condition when a waterbody has demonstrated, through the use of ambient biological and/or chemical data, that it does not significantly violate biological or water quality criteria for that use.

Bank Stabilization - "Methods of supporting the structural integrity of earthen stream channel banks with structural supports to prevent bank slumping and undercutting of riparian trees, as well as overall erosion. Recommended bank stabilization techniques include those of willow stakes, overlapping riprap, or brush bundles (U. S, National Research Council 1992)."

Biological (Biotic) Integrity - The ability of an aquatic community to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitats within a region (taken from: Ohio EPA 1987; Karr *et al.* 1986).

Biosurvey - In field (ambient) sampling of resident biological organisms to assess biological integrity. For Ohio the accepted methods include pulsed-DC methods of electrofishing for sampling fish and, for sampling macroinvertebrates, Hester-Dendy Multiple Plate Artificial Substrate Samplers and dip nets. Other synonyms: ambient (or instream) biological sampling, biosurveillance.

Channelization - General term applied to stream channel modifications, usually designed to improve drainage of fields and/or prevent flooding, which include channel straightening and widening and often is associated with riparian vegetation removal; these activities almost always result in degraded biological integrity via habitat loss and trophic disturbances.

CPOM - Coarse Particulate Organic Matter.

Degradation - A lowering of the existing water quality or biological condition in Ohio's surface waters.

Designated Use - The purpose or benefit to be derived from a waterbody, *e.g.,* drinking water, aquatic life.

DP - Dissolved Phophorus.

Ecoregion - Regions of geographic similarity based on an overlay of maps of land-surface form, soils, land use, and potential natural vegetation; such regions are likely to contain similar aquatic communities.

Ecoregion Criteria - Biological index values that represent the base level of what minimally impacted communities should achieve in a particular ecoregion.

Electrofishing - Method of collecting fish by stunning them with electrical current from a gas-powered generator; the stun is temporary and fish are released unharmed after processing. Processing includes species identification, counting, weighing, and examining for external anomalies. These results are used to calculate the Index of Biotic Integrity (IBI) and the modified Index of Well-Being (Iwb).

Eutrophic - This refers to a highly “productive” body of water that has high concentrations of organic matter, nutrients, and algae.

Exceptional Warmwater Habitat (EWH) - Aquatic life use designed to protect aquatic communities of exceptional diversity and biotic integrity; such communities usually have high species richness, often support rare and endangered species and/or an exceptional sport fishery.

Far-field - Effects of an activity that are observed downstream or away from that activity. For example, phosphorus that enters the stream through the removal of riparian vegetation may have effects far downstream in pools of mainstem rivers or impoundments.

FPOM - Fine Particulate Organic Matter.

Hester-Dendy Multiple Plate Sampler - A sampling device for macroinvertebrates which consists of a set of square hardboard plates (approximately a surface area of one square foot) separated by spacers of increasing width. Aquatic macroinvertebrates colonize or reproduce on this device which is placed instream for six weeks during the summer. Counts of individuals and species are used in calculation of the Invertebrate Community Index (ICI). (See Invertebrate Community Index).

Impacted - This refers to the situation where there is suspected impairment based on the presence of sources (*e.g.,* nonpoint source survey). In such cases there is evidence that some changes or disturbance has occurred to the stream, but there is no quantitative data to establish whether aquatic life uses are actually being impaired.

Impaired - This refers to the situation where there is monitored level data that establishes a violation of some water quality or biological criterion, and hence, an impairment of the designated use .

Index of Biotic Integrity (IBI) - An ecologically-based index that uses fish community data and summarizes them as 12 ecological metrics that can be classified into three categories: species richness, species composition, trophic composition, and fish density and condition (Karr 1981; Karr *et al.* 1986).

Index of Well-Being (Iwb) - A composite index of diversity and abundance measures (density and biomass) based on fish community data (Gammon 1976; Gammon *et al.* 1981).

Invertebrate Community Index (ICI) - An index of biological condition based on ten metrics that measure various structural and tolerance components of macroinvertebrate communities in Ohio streams (DeShon *et al.*, unpublished; OhioEPA 1987).

Limited Resource Water (LRW) - An aquatic life use assigned to those streams with very limited aquatic life potential, usually restricted to mine drainage streams or very small streams (<3 sq. mi. drainage area) in urban areas with limited or no flow during the summer

Modified Warmwater Habitat (MWH) - Aquatic life use assigned to streams that have irretrievable, extensive, man induced modifications that preclude attainment of the Warmwater Habitat Use (WWH); such streams are characterized by species that are tolerant of poor chemical quality (fluctuating dissolved oxygen) and habitat conditions (siltation, habitat simplification) that often occur in modified streams.

Modified Habitat Attributes - These are habitat characteristics, extracted from the QHEI, that areassociated with degraded biotic conditions (low IBIs). These are divided into "high" influence attributes and "moderate" influence attributes on the basis of the statistical strength of their relation with the IBI.

Near-field - Effects of an activity that are observed adjacent or close to to that activity.

Nonpoint Pollution Source - Diffuse sources of pollutants such as urban storm water, construction, farms and mines that are usually delivered to waterbodies via rain runoff and water infiltration.

Performance - See Biological Integrity

Point Source of Pollution - Any source of pollution that arises from a single identifiable point, such as a discharge pipe of an industry or WWTP.

Pollutant Loading - Amount (mass) of a compound discharged into a waterbody per unit of time, for example, kg/day.

PP - Particulate Phosphorus; phosphorus attached to sediment particles.

QHEI (Qualitative Habitat Evaluation Index) - A qualitative habitat index designed as a screening tool to help in assigning designated uses and as an aid in interpreting changes in aquatic communities.

REF - Sites with biological or water chemistry data that meet the criteria to be classified as relatively unimpacted reference sites. For this analysis existing biosurvey-based reference sites (BioC - see below) were supplemented with sites with water chemistry data only (BWQ).

Reference Site - A relative unimpacted biosurvey site that is used to define the expected or potential biological community within a region such as a ecoregion; in Ohio reference sites were used to calibrate the ICI and IBI.

Residual Nutrient Concentrations (RNC) - Concentrations of nutrients (nitrates, phosphorus) typical of low/normal flow conditions in streams.

Rheophilic - Organisms that are “current loving”; usually reserved for organisms that are obligate riffle dwellers.

RFBS (Riparian Forested Buffer Systems) - An acronymn that distinguishes forest riparian buffer strips from other (*i.e.*, grassed) types of stream side vegetation.

Surrogate Measures of Biotic Integrity - Chemical parameters designed to protect aquatic life if they are not exceeded instream. Because they are indirect measures of aquatic community integrity, and mostly derived from laboratory toxicity tests, they are termed “surrogate” (*i.e.*, substitute) measures of biotic integrity.

Summer-Fall Low Flow - Flow regime typical for the Summer to mid-Fall period for Ohio streams characterized by the lowest flows of the years, with base flows in perennial streams supplied by ground water, and occassional surface and subsurface runoff from rain events.

Threatened Streams - These are streams that are currently meeting their designated uses but because of obvious trends (see urban encroachment) or qualitative data are thought to be declining in quality and may become degraded in the future without changes in current practices.

Toxic Substances - Any substance that can cause death, abnormalities, disease, mutations, cancer, deformities, or reproductive malfunctions in an organism.

TP - Total Recoverable Phosphorus.

Urban Encroachment - Increased development in a watershed, especially where it affects the floodplain, riparian zone, and runoff characteristics of a basin.

Use Designation - See “Designated Use”.

Warmwater Habitat Attributes - These are habitat characteristics, extracted from the QHEI, that areassociated with degraded biotic conditions (low IBIs). These are divided into "high" influence attributes and "moderate" influence attributes on the basis of the statistical strength of their relation with the IBI.

Wasteload Allocation - The portion of a streams capacity to assimilate pollutants without violating water quality standards allotted to existing (or future) point sources (*e.g.,* WWTPs)[1]; *i.e.*, the loading (kg/day) of a pollutant allowed to be discharged by a source without violating water quality standards.

Waterbody/Waterbody Segment - A length of stream, based on Ohio EPA's mapping system (Division of Environmental Planning and Management), defined for analysis of water quality trends for this report. Each stream segment is approximately 10 miles in length; there are over 3800 stream segments currently defined for Ohio. Each lake is also a separate waterbody.

Water Quality Standards - The rules set forth for establishing stream use designations and water quality criteria protective of such uses the surface waters of the state.

305(b) - Section of the Clean Water Act that requires a biennial report to assess the progress of the Clean Water Act programs.

Data Transformations

For such a distinct wedge of points, we fit a series of lines through the upper 10% of the points as described by Blackburn *et al.*1990. We have modified the methods of Blackburn *et al.*1990 to attempt to define changes in slope when threshold responses appear to be non-linear or a combination of more than one linear response. For example, a parameter might have no observable effect over some range of the parameter and after some point show a distinct pattern of declining biological assemblages with increases in concentration, until perhaps the assemblage is eliminated. The methods described by Blackburn and employed here work well with large datasets, as we have here, but become more problematical with sparse data sets. Because of the problems with calculating high percentiles in groups that have few data points (*e.g.*, maximum value likely estimate of 95th percentile, for data categories with fewer than 10 values the upper 50% of the points were included in the regression and or data categories with fewer than 20 values the upper 25% of the points were included in the regression. Such a change generally resulted in regression lines more similar to that which were drawn by eye. Because maximum values were often the result of calculating a 95th percentile regression with sparse data, sparse intervals were generally overestimates of 95th percentiles which effected the slope of the regression.

We visually examined relationships between parameters and the IBI or ICI to determine where changes in slope occur and only calculated regressions through the linear portions (*i.e.*, where assemblage threshold responses are likely. Most relationships were fitted with least squares regressions (solid lines on graphs), however, if the regressions substantially differed from what we would fit by eye (dashed line), both lines are plotted. The regression equations are used to back-calculate the “threshold” parameter value above which a specific biological index value is not likely to occur, on the basis that such values have been rare in the past based on large numbers of samples.

[1]Water Quality Standards for the protection of aquatic life should explicitly or implicitly incorporate biological criteria to expect full protection of aquatic life.

[2]the degree to which larger substrates (*e.g.*, boulder, cobble, gravels) are covered by fine sediments (fine gravel, sand, silt).

ACKNOWLEDGEMENTS

This study could not have been accomplished without the extensive collection of biological and water quality samples collected by Ohio EPA Division of Surface Water staff over the 14 field seasons represented in this study. Ed Herricks provided helpful comments on an early version of this document. This study was funded by a Section 319 grant from the USEPA.