Final Technical Report

Development and Evaluation of Ecosystem Indicators for Urbanizing Midwestern Watersheds

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1 Introduction

Urbanization is thought to have negative long-term impacts on stream ecosystems and yet the actual causal relationships between land use change and stream community response have not been well studied. The assumption has been that impervious area within an urban watershed leads to increased runoff during storm events. Such runoff might then contribute to greater erosion, stream channel instability, water quality degradation, and aquatic habitat disturbance. Yet specific predictions on long-term hydrologic impacts, expected changes in stream quality, and stream ecosystem response have been lacking. The focus of this project is on the development of predictive indicators of urbanization that are applicable to midwestern watersheds and stream ecosystems. Predictive models of hydrologic change are combined with physical, chemical, and ecological measurements in urbanizing watersheds to develop a practical approach to assessing urban stream condition and forecasting impacts of future land-use change on aquatic resources.

1.1 Rationale

Natural ecosystems in the Midwest have been impacted by human activities since the mid 1800s. This region is now dominated by corn and soybean fields interspersed with islands of forest, prairie, and wetland. While the economic value of cropland and the ecological value of undeveloped land are well-recognized, both land uses are now vulnerable to expanding urbanization. Well over 50% of the population in this region lives in cities, and the margins of urban and suburban areas are continuing to encroach on rural landscapes. Ecological structure and process within highly fragmented rural landscapes are poorly understood, even as these factors change in response to the growing influence of urban and suburban areas. This situation leads to questions concerning the ability of watersheds to support natural ecosystems on a long-term basis. Ultimately, we need to know what modifications in human activity and land use are needed to increase the long-term survival of natural systems while providing for human commodity needs. The cumulative effects of expanding urban and suburban development on habitats in Midwestern watersheds will depend on both the total area developed and the pattern of that development. For example, a pattern of scattered, low-density housing may cause a different degree of change than clustered, higher-density development. Construction on former (subsurface drained) cropland may produce different types of hydrologic and habitat alteration than similar developments placed in or near forest fragments or wetlands.

Recognition of the ecological importance of human-dominated landscapes and the case for their incorporation into ecological research have been clearly presented by McDonnell and Pickett (1990). They advocate using urban-rural gradients to study such fundamental issues as hierarchy theory and disturbance regimes at various spatial scales. Three major aspects of urbanization (structural, biological, and socio/economic) produce a suite of effects related to ecosystem structure and process. Naiman et al. (1995) recommended the expansion of long-term research on the effects of watershed alterations to examine the influence of land use activities on hydrologic regimes and the subsequent response of biota due to change in nutrients and habitat structure (Figure 1).

While this conceptual scheme for ecological research was envisioned as a gradient between urban and natural (undisturbed) landscapes, the actual pattern is more complex. Highly urbanized areas are still relatively uncommon, as is natural, undisturbed land. Instead the typical gradient in the Midwest is between mixed rural landscapes (disturbed, fragmented) and low-density suburban development. Urbanization generally proceeds across the watershed as a series of clumps that grow and merge over time. Both of these land use types have physical/chemical, biotic, and socio/economic aspects analogous to those shown. However, the linkages among these are still very poorly understood.

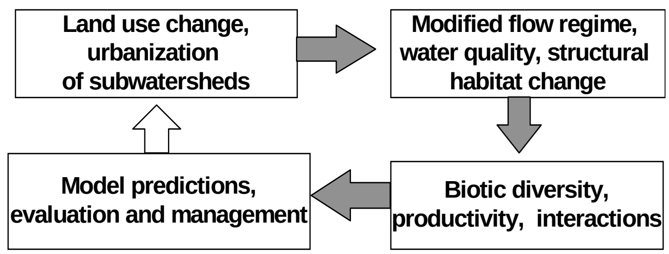

Figure 1. Conceptual approach to evaluating effects of land use change, modified from Naiman et al.(1995).

Environmental indicators: Although environmental impact assessment using dynamic modeling of linked physical and biologic systems would provide the most complete application of our knowledge, such approaches are limited by both the lack of suitable data for most areas of interest, and by the time and expertise required to run such models. In response to this limitation, there is great interest in the development of "indicators", which are simpler variables that provide an index of impacts without requiring full scale modeling. Indicators range from representative elements of the biologic system being impacted (e.g., sensitive fish species), to measures of the physical changes that are impacting the biologic system (e.g., suspended sediment or nutrient concentration), and measures of the alterations in watershed conditions that are driving the physical changes (stressors such as population density and percent impervious area) (Table 1, adapted from Claytor, 1996). If the goal is to monitor existing impacts, then biological system indicators would

Table 1.
URBANIZATION INDICATORS

Physical & Hydrologic:
Stream widening/downcutting
Physical habitat monitoring
Impacted dry weather flows
Increased flooding frequency
Stream temperature monitoring

Water Quality:
Pollutant constituent monitoring
Non-point source loadings
Exceedance frequencies of standards

Biological Indicators:
Fish assemblage
Macroinvertebrate assemblage
Composite indicators (e.g. IBI)
Other biological indicators (e.g. mussels)

Programmatic Indicators:
Permitting and compliance

Growth and development metrics

be selected, whereas if the goal is to develop watershed management strategies, then stressors are the primary focus. In any indicator approach, a major potential limitation is provided by the extent to which changes in the indicator are representative of changes in the more complex system. Indicators established primarily using correlation or regression techniques have statistical validity for the data set on which they were based, but extrapolation beyond the conditions of the original data set is far less reliable. This is because a statistical link does not prove a physical causative relationship, so outside the original conditions of the study, the predictability of the relationship between the indicator and the system is unknown. A solution to this significant limitation in the indicator approach is to link indicators and system behavior through process modeling.

1.2 Objectives

The focus of this project has been on the development of predictive indicators of urbanization that are applicable to Midwestern watersheds and stream ecosystems, and more importantly on developing and testing an objective methodology for such ecosystem indicators using process models to examine stream response to changing land use. Objectives were to:

1. quantify the impacts of urbanization on hydrologic regimes, water quality, and habitat structure of stream ecosystems using paired experimental watersheds, and to develop linked models that accurately predict these impacts.

2. use the linked models as a virtual laboratory within which to generate and test indicators of urbanization and hydrologic change in terms of responses of fish and macroinvertebrate communities.

3. use these models and indicators to assess the response of stream communities to alternative urbanization scenarios with extension to larger watersheds in the region.

2 Study Sites

2.1 Watersheds

The project examined eight watersheds in central Indiana that are in transition from rural to urban. Seven are in the greater Indianapolis area (Marion County) and one is located near Purdue University (Tippecanoe County)(Figure 2). Additional nearby streams were used for particular aspects of the project. The Marion county streams are all tributaries of the West Fork White River (HUC 05120201); the Tippecanoe County streams drain directly to the Wabash (HUC: 05120108).

Both counties are within the Eastern Cornbelt Ecoregion (Omernik and Gallant, 1988), which was originally forested but is now dominated by extensive cropland. Typically 75% of the land in this region is row cropland (mainly corn and soybeans) and 25% is in permanent pasture, small woodlots, or urban areas. The Indianapolis area was composed of row crop agriculture prior to urbanization over the past four decades. While Marion County is centrally located in the region, western Tippecanoe County represents a transition zone toward the Central Cornbelt ecoregion, which was originally tall-grass prairie. Both Counties also belong to the Tipton Till Plain physiographic province, a depositional plain of low relief with postglacial stream erosion. The surficial geology consists of glacial till and outwash sands and gravels.

Regional hydrology of the area is dominated by thunderstorms occurring throughout the year but primarily concentrated in the late-winter and early-spring. Perennial stream flow is maintained by surface and subsurface flow from the uplands, which have a high potential for erosion. Baseflow contributes significantly to the total flow in this region.

Drainage area for the survey sites ranged from 6.9 to 66.7 km^2 while wetted width ranged from 2.3 to 12.9 m. Land use ranged from predominantly agricultural (0% high density urban) to predominantly urban (68% high density urban). Only two study streams in Indianapolis, Pleasant Run and Pogue's Run, receive storm sewer discharge. Pogue's Run was not used for fish sampling since it is diverted underground within the city and probably has blockage to fish passage.

Nonpoint source (NPS) pollution of nutrients to the streams in this region ranks among the highest in the United States. The Middle Wabash-Little Vermilion watershed ranks 2087th out of 2110 watersheds for agricultural runoff (in the top 99%), with 95% for nitrogen runoff, and 98% for sediment delivery. Similarly, the Upper White River ranks in the top 99% of all watersheds for ag runoff, 97% for nitrogen, and 93% for sediment delivery. Unlike other parts of the country, watersheds in this region actually experience a decline in sediment and nutrient NPS as land use shifts from agricultural to urban. Thus it is unlikely that Midwestern streams will respond to urbanization in quite the same way as streams in more forested regions.

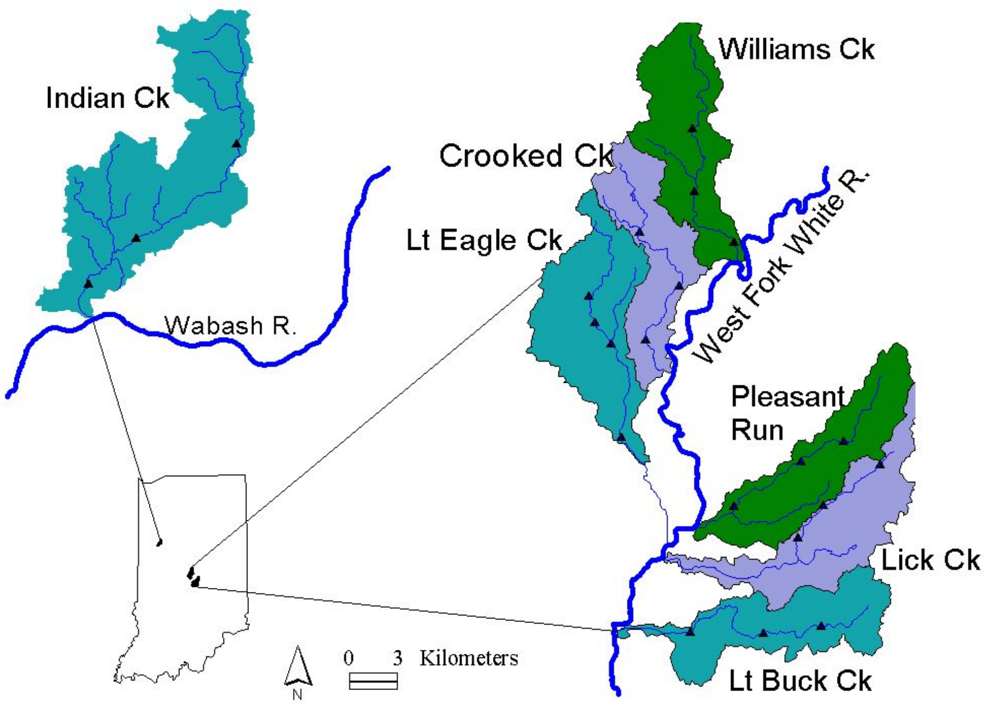

Figure 2. Project watersheds. Black triangles show the upper, middle, and lower sampling sites on each stream.

Marion County is by far the more urban of the two counties since it is dominated by the City of Indianapolis, the 12th largest city in the nation. Population density in Marion County is currently 2173 persons per sq. mile, while Tippecanoe County has only 298 persons per sq. mile. Growth rates between 1990 and 2000 were 7.9% and 14.1% respectively. The West Fork of the White River constitutes approximately 1/20th of the state land area, but supports 1/3 of the state's population.

2.2 Land use classification

The City of Indianapolis did not have a recent land use map available (although they do have a zoning map). Two land use maps were developed for this project using remote sensing, one for 1997 representing pre-study conditions, and one for 1999 representing study period conditions. Two land use maps also allowed us to assess land use change over a short time period, as well as over longer time periods using previous land use maps (1990s and 1980s). Production of the maps and data acquisition was supported both by this grant, and a contract from the City of Indianapolis.

High-resolution SPOT satellite images were acquired for July 1997 and a land use classification was carried out to produce the land use map. Classification was performed using ERDAS Imagine GIS Image processing software at Purdue University. All classification processes were performed on raw data to prevent loss of information due to processing steps such as rectification and re-coding. A supervised classification technique with maximum likelihood parameter derivation was used to divide the data into different land use categories based on spectral signatures. Since spectral signatures of ground features differ significantly, it is possible to distinguish between land uses. For the purpose of classification, about 100 –150 training points were collected from each of the three images based on knowledge of the

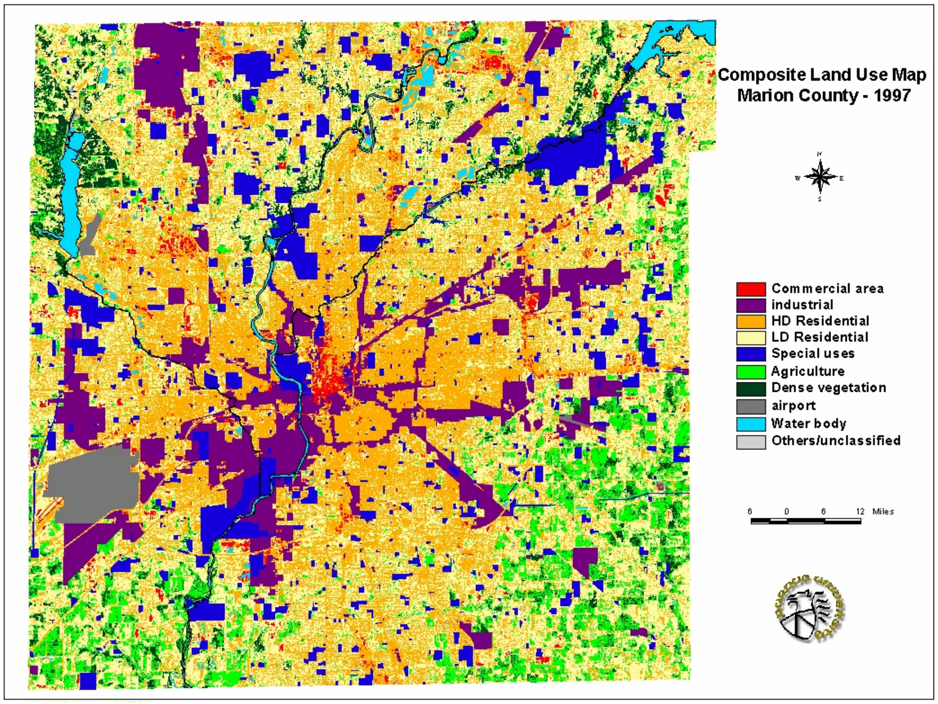

Figure 3. Land use classification map for Marion County.

field area. The whole classification process was repeated where results did not fit known land uses. Once the classification process was completed, re-coding was performed to minimize the number of classes. A rectification model was developed using the raw data, which was later used with the processed data to register the classified image to the UTM coordinate system. Once all the images were registered, they were joined to produce a mosaic of the whole area. Individual watershed areas were later extracted from this image. The land use map had seven classes (agricultural, low-density urban, high-density urban, commercial, forest, and water body) and this was then refined by adding four more classes (industrial, airport, institutional and special uses) using GIS and zoning data acquired from the City of Indianapolis. This composite land use map (Figure 3) forms the basis of the land use data used in comparing ecological and geomorphological conditions along gradients of urbanization, and is also useful for city planners in the Indianapolis Department of Metropolitan Development.

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For the ecological indicator research, watersheds above each stream sampling site were defined using the watershed delineation procedures of ArcView (ESRI Inc.). Elevation data used to delineate watersheds were obtained from 1:24 000 scale digital elevation models (USGS). Land use characteristics of the watersheds upstream of each site were determined from supervised classification of 1997 multispectral satellite images (SPOT Inc.) using ERDAS/Imagine v.8.3 image processing software. One image included the Indianapolis metropolitan area while another image included the Indian Creek watershed in Tippecanoe County (Figure 4).

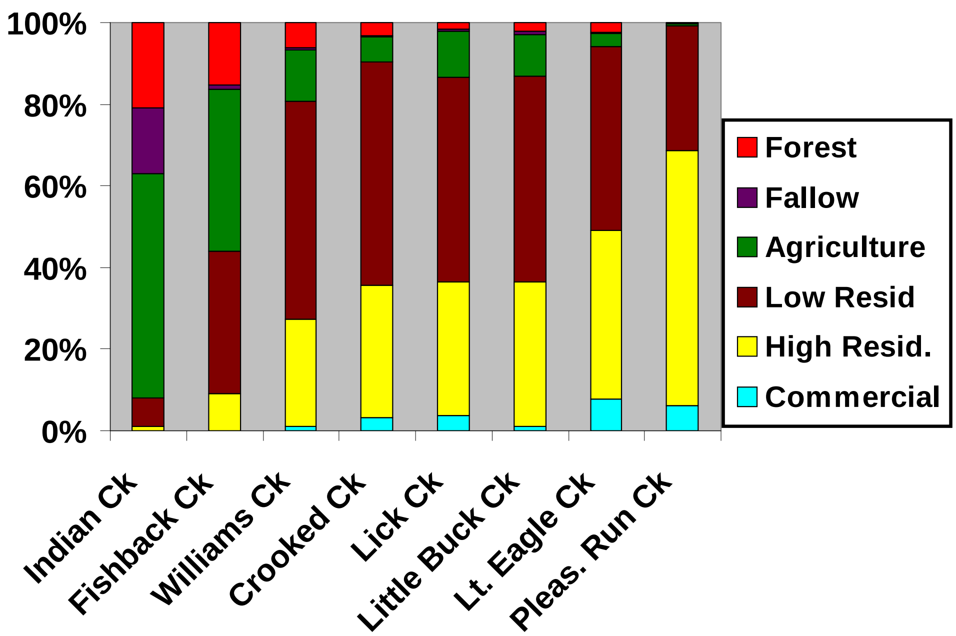

Figure 4. Land use types for streams used in this project.

2.3 Watershed imperviousness estimation

Watershed imperviousness was determined from two digital databases provided by the City of Indianapolis and which were independent of the land use databases described above. The first was a digital map of line features representing roads and streets while the second was a digital database of outlines of buildings. The roads and streets file was converted to a polygon coverage by constructing a 10 m buffer around the linear features. A union of the two maps was assumed to represent the total area of impervious surfaces in Marion County that includes the city of Indianapolis. Since the headwaters of Crooked and Williams Creeks and all of Indian Creek were not contained within the database, watershed imperviousness for those cases was based on a simple linear regression relationship between watershed imperviousness and the urban land use data described above (Figures 5 and 6).

Imperviousness = 0.3714 (high density urban + commercial) + 0.0043

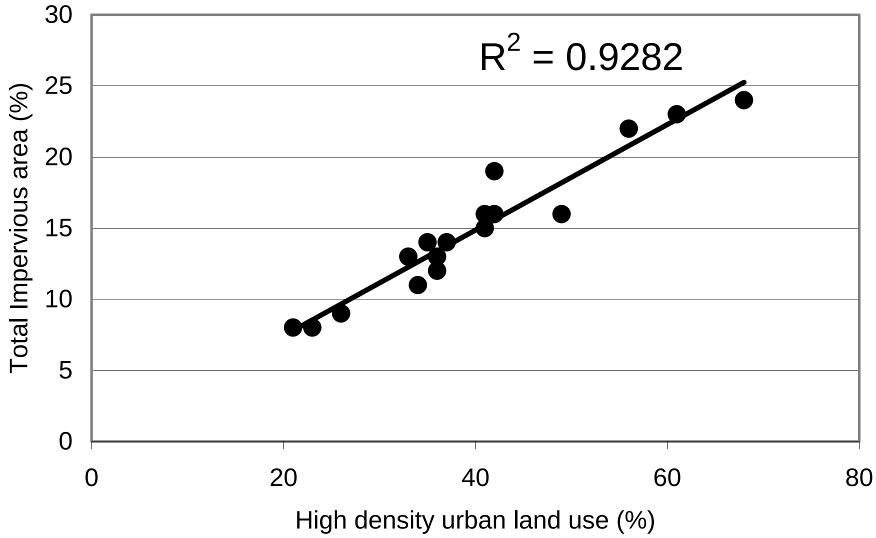

Figure 5. Relationship between satellite-derived urban land use and total impervious area derived from digital outlines of roads and buildings in Marion County.

Riparian land use upstream from each study site was calculated as the cumulative percentage of forested land cover within a 100 m buffer on each side of the stream. Forested land use was derived using a bufferanalysis to extract the land use data on each side of the creeks.

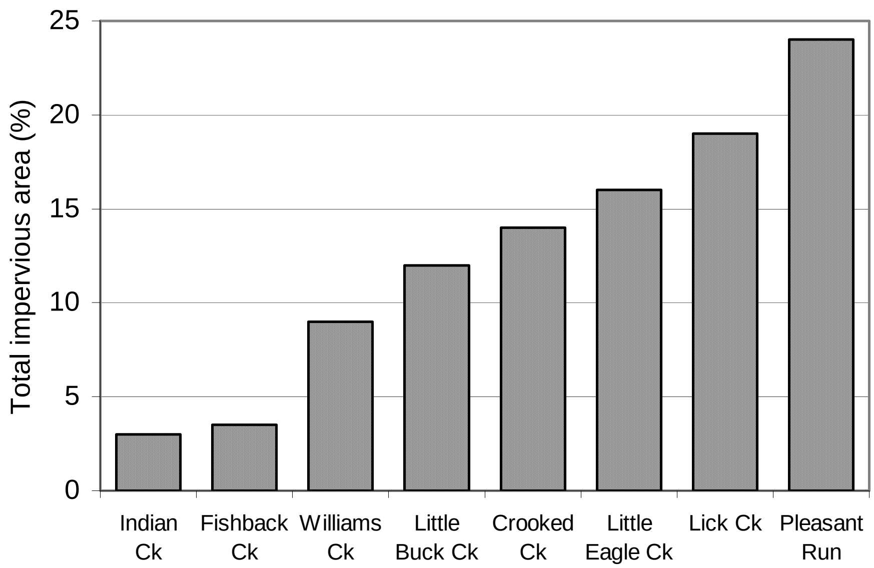

Figure 6. Total impervious area in study watersheds.

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3 Hydrologic and Temperature Modeling

The small streams addressed in this project are runoff-dominated and highly variable in their flow characteristics. Stream biota may be affected adversely by transient conditions occurring at peak or low flow. Physical factors affecting the flow and temperature of such streams were examined by the development of a numerical model of one-dimensional unsteady stream flow and temperature (see Appendix for Younus et al. 2000). A complete description of the model, equations used, key assumptions, performance and test results are included in that reference.

The model accounts for the effects of arbitrary stream geometry, variable slopes, variable flow regimes, and unsteady boundary conditions. The model was verified using stream flow measured at two sites in Little Pine Creek, Tippecanoe County, Indiana, from May 15 to June 9, 1998. The model predicted the diurnal variability in the stream flow well (Figure 7). The standard error of estimation was 50 L/s for the stream flow range from 400 L/s to 3200 L/s during the simulation period. In addition, the model can simulate cross-sectional averaged velocities, shear stress velocities, and water depth variability. The hydrodynamics portion of the model can be used to predict shear stress and substrate mobility in urbanizing channels so that these physical habitat characteristics can be correlated with the responses of fish and macro-invertebrate communities.

The model was also successful in predicting temperature fluctuations every 15 min over the 25 day measurement period (also Figure 7). The thermal transport model accounts for the effects of solar radiation, air temperature, relative humidity, cloud cover, wind speed, heat conduction between water and streambed, subsurface flow, and shading by riparian vegetation. The two most significant factors in determining stream temperature under these conditions were found to be subsurface inflow and solar radiation, both of which could be altered by land use changes. The model has application to predicting thermal limitations for fish and other aquatic organisms that may be stressed during summertime, low flow conditions.

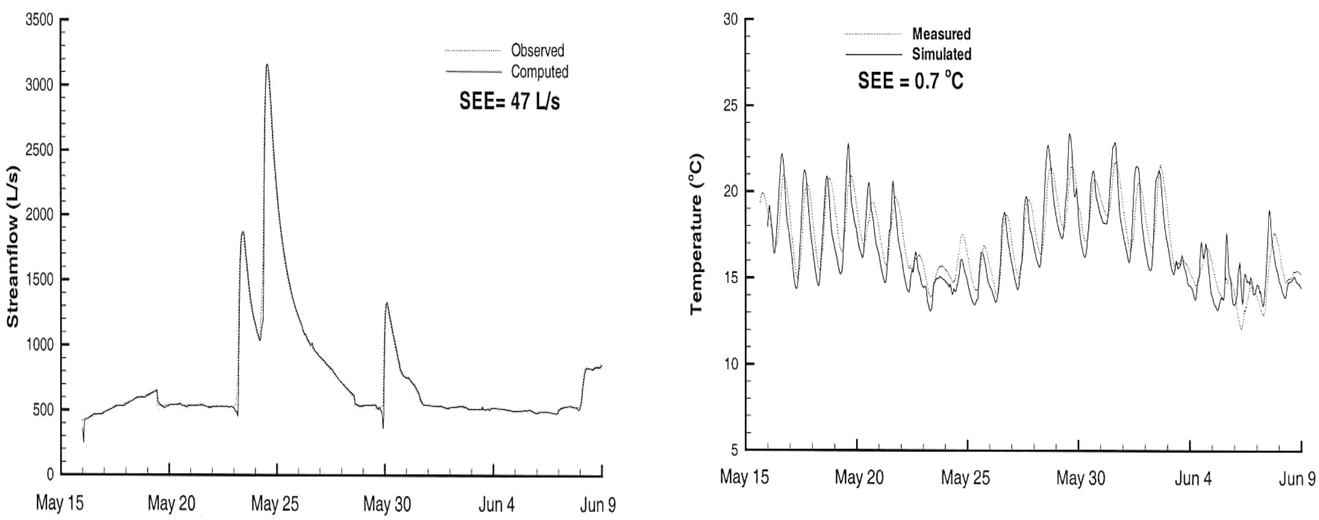

Figure 7. Observed and computed stream flow and temperature for Little Pine Creek.

4 Dissolved Oxygen and P/R Modeling

The streams in this study experience significant diurnal fluctuations in dissolved oxygen (DO) caused by photosynthesis (P) and respiration (R) of the periphyton community. Low dissolved oxygen (less than approximately 5 mg/L) combined with high summer temperatures can stress many types of stream fishes and insects. The factors determining diurnal DO fluctuations were studied by applying a model using the Extreme Value Method based on the maximum and minimum deficits of DO over diurnal cycles (see Appendix for Wang et al. manuscript, which gives a complete description of the model, its assumptions, and test results). Estimated rates were used to characterize stream community productivity (P) and respiration (R). Long-term simulations of DO dynamics in stream using a one-day metabolism rate and the traditional mass balance equation of DO were explored. The basic equation relating the rate of change of DO concentration was augmented with dimensionless relationships between metabolism rate and discharge. This approach successfully predicted 15 min oxygen readings taken in two of the study streams over a 50 day period (Indian and Little Eagle Creeks) (Figure 8).

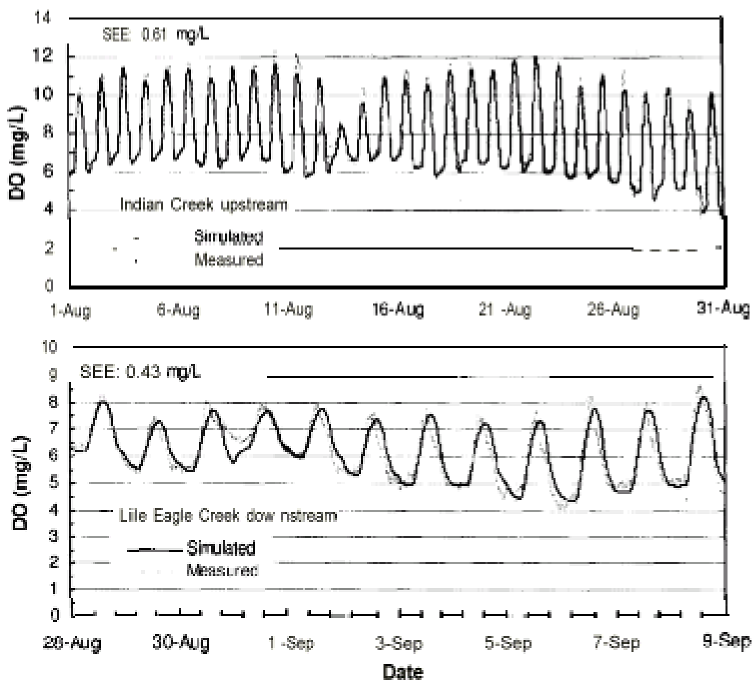

Figure 8. Field observations and simulations of DO dynamics for Indian and Little Eagle Creeks using the parameters estimated by the extreme value method.

As can be seen in Figure 8, DO did not decline below about 5 mg/L during the hottest part of the summer, suggesting that oxygen stress is not likely to be the limiting factor for stream communities in these particular stream reaches, although they may be very significant elsewhere (Smale and Rabeni, 1995). The model results also show that photosynthetic oxygen production was temporarily depressed during an August 11-13 spate in Indian Creek, which has a well-developed periphyton community. This reduction (Figure 9) is most likely due to a temporary reduction in light penetration (i.e. increased depth and turbidity) rather than to a loss of periphyton by scouring. If that had occurred, the rebound in photosynthetic DO production would be much slower.

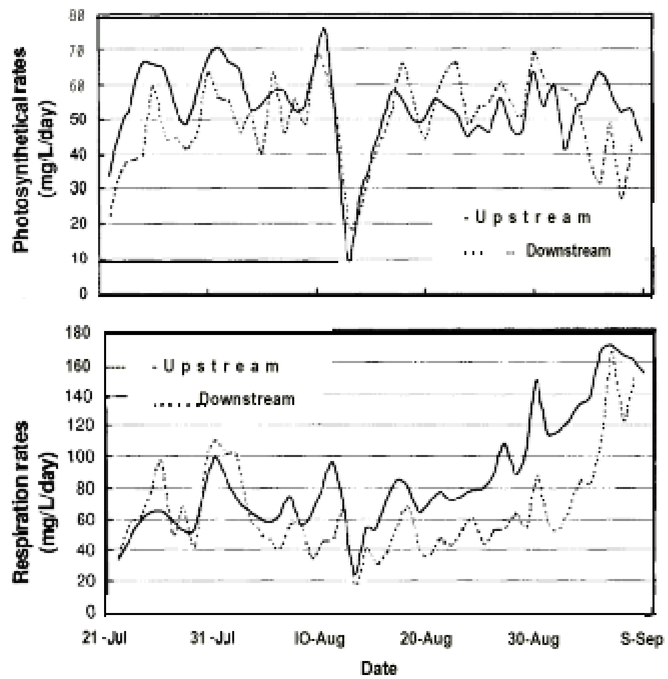

Figure 9. Estimate photosynthetic and respiration rates for two sites in Indian Creek.

The model developed by Wang et al. is much simpler than the physically-based model originally envisioned in the project proposal. That approach would have required six transport equations for organic nitrogen, ammonia, nitrite, nitrate, phosphorus, and dissolved oxygen. The Extreme Value Method reported here requires much less input data and has been shown to work so well for these streams that the added complexity of the model proposed earlier is not justified.

5 Long-term Hydrologic Response Prediction

Even though ecological impacts may be defined by relatively short-term storm events, long-term hydrologic responses to land use change at the watershed scale are also important to understand and to forecast in light of urban sprawl. A long-term hydrologic impact assessment (L-THIA) model to predict annual runoff and NPS pollutant loading has been developed using the curve number method (see Appendix for Bhaduri et al. 2000, which gives a full description of this approach). A sensitivity analysis of these model is reported by Grove et al., in press (see Appendix). Input requirements include long-term climatic records, soil types, and land use information. The model is linked to a GIS for generation and management of model input and output data, and is directly accessible at a web site (Figure 10).

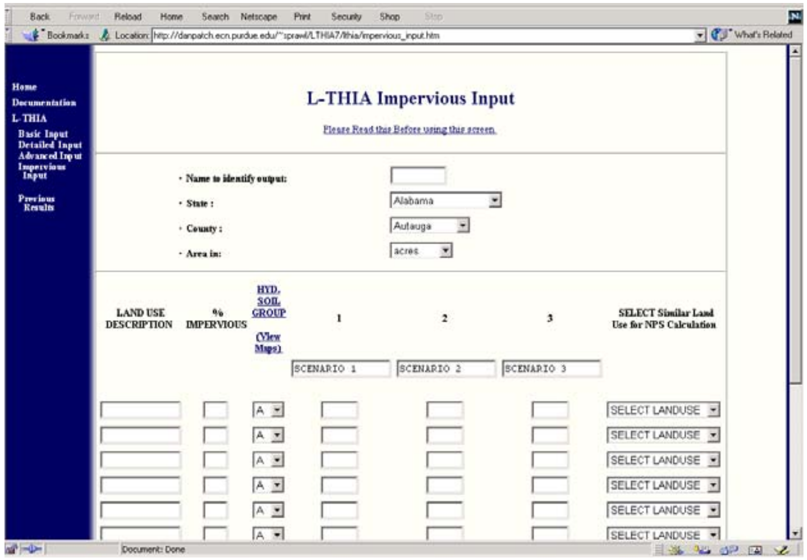

Figure 10. Web site for input of land use and soils data for L-THIA model application (URL is given in Section 16).

5.1 L-THIA Evaluation

The L-THIA model was applied to Little Eagle Creek, which has undergone an 18% increase in urban (impervious) area between 1973 and 1991 and a subsequent increase in stream flow (Figure 11). The urbanization has lead to a calculated 80% increase in annual average runoff volume and estimated increases of more than 50% in annual average loads for Pb, Cu, and Zn.(Figure 12) At the same time, estimated nutrient (N and P) loads calculated by L-THIA decreased by 15% as a result of conversion of agricultural land. Water quality sampling done in conjunction with the biological surveys for this project confirmed that nutrient concentrations in Indianapolis streams are quite low compared to nearby rural streams. Annual mean values of 1.8 mg/L nitrate and 0.12 mg/L total P in Marion Country streams are lower than the 20 mg/L nitrate and 0.2 mg/L P observed in the more rural Indian Creek (Tippecanoe Co.). Increased runoff volume in the urban streams would partially offset the reduction in nutrient concentrations to produce less of a change in annual loadings.

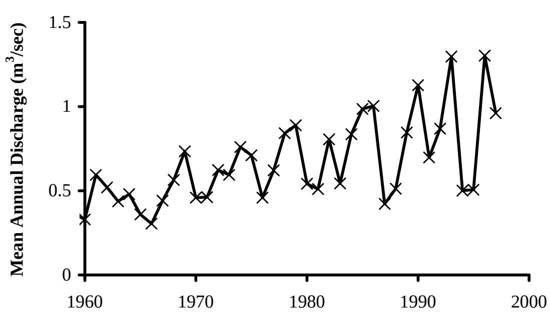

Figure 11. Stream gauge records for Little Eagle Creek (USGS).

The L-THIA curve number method used for long-tern planning purposes gives annual mean runoff. Since we would like to predict daily runoff in order to predict stream channel and ecological impacts, we also ran L-THIA using actual daily rainfall records for the Indianapolis area. Because L-THIA and other curve number methods are sensitive to antecedent soil moisture, the predicted runoff results did not correlate well with observed stream flows. A watershed-scale distributed parameter runoff model such as SWAT would be more suitable for daily runoff simulations and was originally proposed for this project. Considerable effort went in to calibrating and applying SWAT to several watersheds including Lick Creek in Indianapolis. For the period 1993-1997, SWAT gave good results ($R^2 = 0.85$) for the prediction of daily flows in Lick Creek during summer, but less satisfactory results for spring and fall flows (R^2=0.53). SWAT significantly underestimated daily flow at these times, but better results can probably be obtained in the future with additional model calibration.

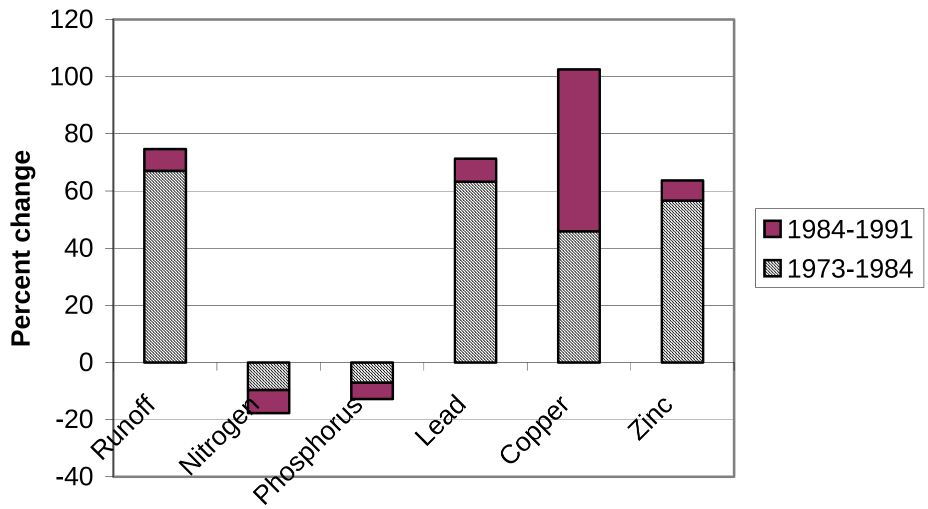

Figure 12. Percent change in average annual runoff volume (x 10^6 m^3), nitrogen (x 10^4 kg), phosphorus (x 10^3 kg), lead (x 10^4 kg), copper (x 10^3 kg), and zinc (x 10^3 kg) over time in the Little Eagle Creek watershed (Source: Bhaduri et al. 2000).

5.2 Application to Regional Land Use Planning

The L-THIA modeling approach is expected to be highly useful to regional planners because of its convenience and simplicity. A statewide extension program called "Planning with POWER" was launched in 2000 by the Purdue University Extension Service with support from the Illinois-Indiana Sea Grant Program (URL below). Its purpose is to promote watershed planning at the local level through web-based education, workshops, and publications. The L-THIA model has been adopted as part of its training and planning process. We have also provided the geomorphology and ecological results of this STAR project to POWER and will be collaborating on several publications for use throughout the state.

6 Geomorphic Stability of Urban Channels

Stream channel morphology is a critical link between land use and aquatic habitat quality, as channel stability is affected by levels of urbanization. The increased runoff predicted by L-THIA and observed in USGS gauge records for Indianapolis streams should lead to stream channel enlargement, either by lateral (bank) or downward erosion of the bed. This would be expected to cause changes in biota sensitive to substrate disturbance.

Channel stability was quantified for seven of the study streams in this project (Table 2) (see Appendix for complete report by Doyle et al. 2000). As a first step in this geomorphic approach, indicators of geomorphic stability were used to assess the relative geomorphic stability between study sites of various levels of urbanization. Preliminary assessment was accomplished based on determination of dominant channel processes and the state of the channel relative to local structures (e.g., bridge pier footings, abutments, hanging culverts, etc.). In addition, we tested the ability of existing and proposed measures of channel stability to discriminate between stable and degrading channels in an urban setting. The existing measures consist of two qualitative methods (e.g., rating schemes), five existing quantitative methods (stream power, unit stream power, excess shear stress, bankfull discharge per unit watershed area, recurrence interval of bankfull discharge), and one proposed quantitative measure (recurrence interval of critical discharge).

Table 2. General study site geomorphic characteristics: values represent reach average conditions.

Site	Drainage area (km^2)	% High density urban	Gradient	D_{50} (mm)	Q_{bf} (m^3/s)	Width to depth ratio
Little Eagle-Upper	17.8	32	0.004	35.9	22	8.1
Little Eagle-Lower	66.7	28	0.002	30.2	21	12.2
Little Buck-Upper	15.9	2	0.002	38.7	9	9.7
Little Buck-Middle	33.4	11	0.002	25.3	17	7.7
Little Buck-Lower	52.3	7	0.001	15.4	16	12.1
Crooked	40.9	11	0.010	41.4	43	12.0
Indian	72.1	0	0.003	47.5	27	8.9

Overall, urbanization was not a distinguishable causal factor in channel instability, but certain measures of channel instability were associated with higher levels of urbanizations. The two qualitative stability assessment methods did not give exceptional results in distinguishing between what were considered degrading and stable sites (Table 3). Both methods seem to equate channel stability with channel uniformity. Based on our results, the two qualitative methods of stability analysis are best suited for local channel instability (i.e., road crossing scale) but not necessarily for identifying larger-scale instability, such as incision of a channel resulting from urbanization. This is an important finding as recent governement field manuals have suggested using these qualitative methods for identifying watershed-scale channel instabilty, which our results indicate may not be appropriate in comparison to available quantitative measures (Table 4), which were much more successful at identifying unstable channels.

Most of the sites surveyed were representative of what was considered to be typical of urban channels: relatively straight reaches (a result of past channelization in many cases), occasional levees, and many local influences such as culverts, riprap, and other bank protection which was frequent but discontinuous along the banks. For preliminary analysis of the effects of urbanization on geomorphic stability, the sites were divided into high urbanization, medium urbanization, and low urbanization (Table 5). Using both the qualitative and quantitative measures of channel stability, very few measures revealed significant differences between the groups segregated by urbanization. Those assessment methods which discriminated between groups of urbanization class (with $p < 0.10$) include excess shear stress, bankfull discharge recurrence interval, and critical discharge recurrence interval (Table 6). Results of comparison using all other measures of channel stability, including the qualitative methods, were insignificant ($p > 0.10$).

Table 3. Comparison of average qualitative stability values for stable and degrading sites. Note: higher ratings are associated with greater instability

	Average		Significance of difference[a]
	Stable sites	**Degrading sites**	**p =**
Johnson et al % of total possible	46.9	52.1	0.13[*]
Pfankuch % of total possible	57.1	66.4	0.14[*]

[a] significance of difference in means tested using one-tail t-test assuming unequal variance; unequal variance tested using F-test ($p < 0.05$); [*] indicates that difference in variance was insignificant and that one-tail t-test assuming equal variance was used

The channels in our study area are responding to urbanization by incising (Figures 13,14). Hence, the current stability of a channel is a function of not only the level of watershed urbanization, but also the length of time of urbanization in the watershed, as well as the relative location of the site within the watershed. Study sites lower in watersheds most-likely responded to urbanization years ago, while upper sites are currently responding to the more recent urban sprawl covering the upper portions of the watershed. Sediment derived from the incising reaches in the upper portions of the watershed is being delivered to the lower reaches as deposition, thus leading to degrading upper reaches and stable (perhaps aggrading) lower reaches.

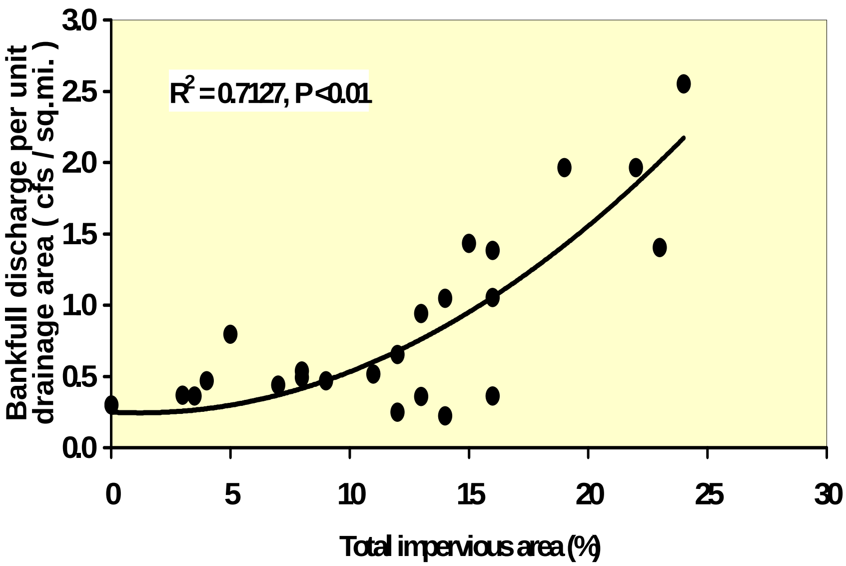

Figure 13. Channel enlargement occurs with increasing watershed imperviousness in the Indianapolis area.

Figure 14. Little Eagle Creek, middle site, showing downcutting and unstable banks. Martin Doyle provides scale.

In our analysis of urbanization using channel stability indicators, we found that only three stability measures varied between levels of urbanization. While the high urbanization sites and medium urbanization sites contained both stable and unstable channels, they still had different average values of the three indicators when compared to the low urbanization sites, indicating some adjustment of measured parameters due to urbanization. Because the initial effect of urbanization on channels is an increase in runoff, the fact that bankfull discharge and critical discharge recurrence intervals were different between the various levels of urbanization indicates that urbanization is affecting channel morphology, watershed hydrology, or both. That is, urbanization has most likely caused a certain degree of channel adjustment as well as increasing the frequency of discharge events greater than the critical discharge.

Table 4. Comparison of average quantitative stability values for stable and degrading sites

	Average[a]		**Significance of difference**[b]
	Stable sites	**Degrading sites**	**p =**
τ (N/m^2)	30.4	99.7	0.01
τ_e (N/m^2)	1.0	2.7	0.03
Q_{bf} per watershed area (m^3/s/km^2)	0.44	1.15	< 0.01
Q_{bf} recurrence interval (days)	139	627	0.05
Q_c recurrence interval (days)	150	35	< 0.01
Ω, (W/m)	400	2539	0.05
ω (W/m^2)	30	170	0.04

[a] represents the average of all cross section values

[b] significance of difference in means tested using one-tail t-test assuming unequal variance; unequal variance tested using F-test ($p < 0.05$)

Table 5. Percent of watershed high density urban and subsequent subdivision of sites for analysis

Site	% High density urban	Classification
Little Eagle-Upper	32	High
Little Eagle-Lower	28	High
Little Buck-Upper	2	Low
Little Buck-Middle	11	Medium
Little Buck-Lower	7	Medium
Crooked	11	Medium
Indian	0	Low

Table 6. Comparison of the average stability measures for channels with high, medium, and low levels of urbanization. Only those comparisons with $p < 0.10$ are included.

	Average[a]			Significance of comparisons between groups[b]		
	High	Medium	Low	High vs. med	High vs. low	Med vs. low
τ_e (N/m^2)	1.3	2.1	0.8	-	0.03*	0.04
Q_{bf} recurrence interval (days)	312	143	147	0.09	-	-
Q_c recurrence interval (days)	88	82	183	-	0.07	0.02

[a] represents the average of all cross section values

[b] significance of difference in means tested using one-tail t-test assuming unequal variance; unequal variance tested using F-test ($p < 0.05$); * indicates that difference in variance was insignificant and that one-tail t-test assuming equal variance was used

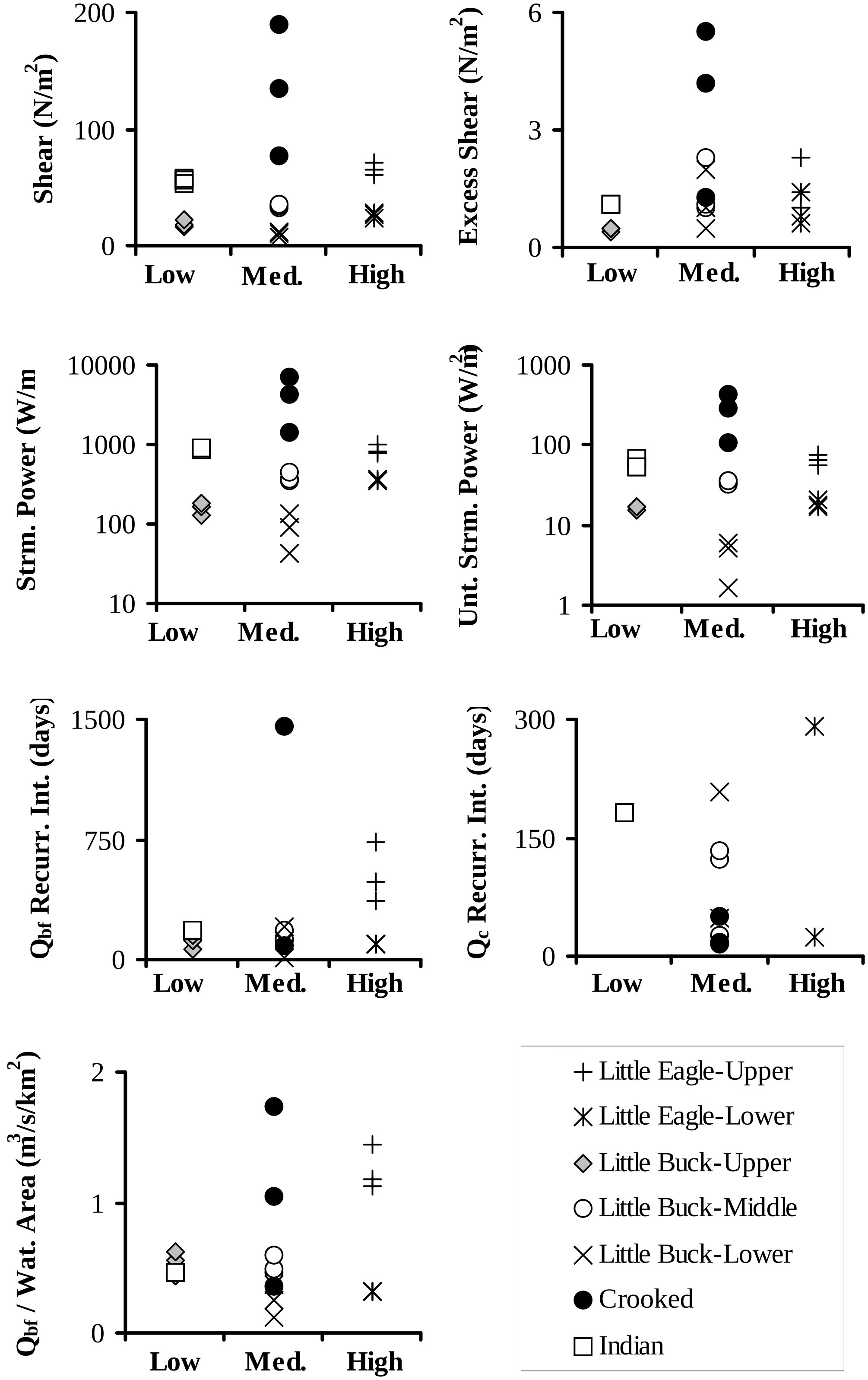

Figure 15. Quantitative stability assessment results for comparison of effects of various levels of urbanization on channels (This is a corrected version of Figure 6 from Doyle et al. 2000).

7 Periphyton Communities in Urban Streams

Stream periphyton, macroinvertebrates, and fish communities were sampled in seven of the study streams and evaluated as possible urbanization indicators for this study. Periphyton was collected during relatively low flow in July-August 1999 and July-August 2000 at upper, middle, and lower stream site (Deitch 2001). All selected sites were in riffle zones corresponding to the collection sites for macroinvertebrates. Twelve pieces of epilithic substrate were selected at each site. Each rock was removed and scrubbed with a hard-bristled brush to remove periphyton. Aluminum foil was used to measure the area of substrate sampled. Periphyton samples were filtered, dried, weighed, and ashed for AFDM determination. Chlorophyll content was determined at middle sites in 2000. Bi-weekly water samples were also taken at these sites in 2000 to establish ambient nutrient (N and P) levels in these streams. Field and laboratory methods generally followed the QA/QC recommendations of the USGS NAWQA program (Moulton et al. 2000).

Periphyton biomass was similar between the rural and urban streams receiving significantly different NPS pollution, which suggests that nutrient levels (N and P) are not critical factors structuring these periphyton communities. Rather, light levels, which are generally higher at downstream sites or sites without riparian shading, appear to determine the amount of growth. High periphyton biomass was correlated with large diurnal fluctuations in dissolved oxygen. Little Eagle Creek had the lowest periphyton production, and also had the smallest dissolved oxygen changes of any stream measured.

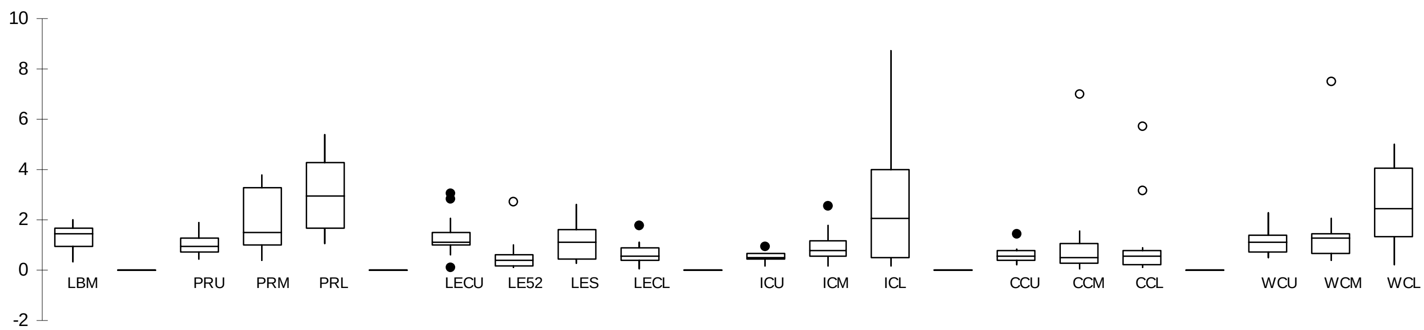

Figure 16. Box plots of biomass (ash-free dry mass in mg/cm^2) of riffle periphyton in urbanizing Indianapolis streams sampled in 1999. Streams are represented by upper, middle, and lower sites (U,M,L respectively). A significant increase in periphyton production occurs at most downstream sites due to increased exposure to light.

The effect of increased water velocity and shear stress on periphyton communities was investigated using artificial streamside channels (Deitch 2001). Two channels received water from a typical agricultural stream with high nutrients (20 mg/L nitrate, 0.08 mg/L total P) and two channels with lower levels (6 mg/L nitrate and 0.06 mg/L total P). After a full periphyton community developed (5 weeks of undisturbed growth), each channel was subjected to a series of three simulated low-intensity hydrologic disturbances, one per week. Each disturbance caused an increase in shear stress of approximately 3.5 X times normal conditions. The community that developed under higher nutrients was composed of thick Chlorophytes and had a biomass of 20-40 mg/cm^2 ash-free dry mass; the lower-nutrient community was mostly composed of loose Chromophyte filaments with 2-7 mg/cm^2 AFDW. The two communities responded to the series of disturbances differently. Results indicated that increased water velocity and shear stress affected periphyton accrual over both nutrient conditions. Higher nutrient levels yielded greater resistance and lower nutrients gave greater resilience to the disturbances. In both treatments, periphyton production recovered to previous levels within one week after disturbance. Greater community resistance under higher nutrients suggests that the high-nutrient community could support a more stable community of herbivores than the low-nutrient community; different biomass and community composition suggest that each periphyton community would support different biotic members.

8 Freshwater Mussels as Indicators of Urbanization

The diversity of freshwater mussels (Unionidae) is particularly high in the rural streams of Indiana but many species are declining throughout the Midwest. While mussels were not a part of the original plan for this project, we were fortunate to obtain survey data for all of the small streams in Tippecanoe County (Myers-Kinzie et al., 2001). These observations (presence/absence) were matched with the channel stability metrics described above to examine the relationship between mussel occurrence and habitat (see Appendix for the full report by Myers-Kinzie et al. *in press*). Most mussel species have a preference for substrates in stable, non-erosional sites. Increased runoff and channel instability could jeopardize the depositional habitats in urbanizing streams. Thus we found that the urban streams have reduced mussel populations (found at 28% of urban sites) compared to similar rural streams (found at 46% of sites). Sites with unstable channels and high shear stress (whether in urban or rural areas) were negatively correlated with the presence of mussels (Figure 17). Because of their slow growth, long life spans, poor dispersal, sensitivity to erosion, and complex reproductive requirements, it is clear that freshwater mussels are especially vulnerable to physical habitat degradation. The fact that they have not been routinely surveyed or studied as ecological indicators is a concern since their habitat requirements are so distinct from other stream organisms.

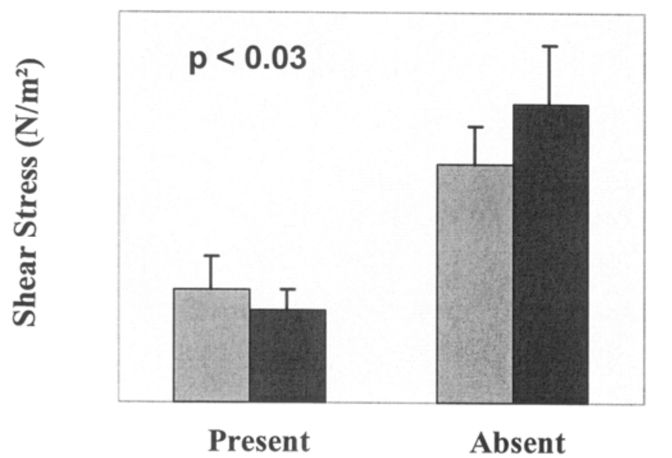

Figure 17. Sites where mussels were found had significantly lower shear stress than sites without mussels. This applied both to streams in rural (light bars) and urban areas (dark bars).

9 Aquatic Insects as Indicators of Urbanization

This section represents the work of Melody Myers-Kinzie, with assistance from a large number of Purdue personnel including Margaret Wacker, Laban Lindley, Peter Mascenik, Vickie Poole, Cameron Guenther, Cecil Rich, Martin Doyle, and Amy Covert. A manuscript is currently in preparation.

9.1 Introduction

The rational behind many biotic assessment protocols is that macroinvertebrate taxa differ in their tolerance to environmental stresses such as chemical pollution. Other forms of stress, such as hydrologic instability, streambed scour and watershed development may also affect macroinvertebrate communities (Quinn and Hickey, 1990; Scarsbrook and Townsend, 1993). In a study of New Zealand rivers, more developed watersheds had lower diversity, taxonomic richness, and numbers of Ephemeroptera, Plecoptera, and Trichophtera species than less developed watersheds (Quinn and Hickey, 1990).

Karr and Chu (1999) define a metric as an “attribute empirically shown to change in value along a gradient of human influence.” Metrics typically involve the response of biotic communities to one or more forms of habitat degradation. These biotic communities may be fish (Karr et al., 1986) or macroinvertebrates (Maxted et al., 2000; Barbour et al., 1996). Multiple metrics increase the probability of accurate assessment (Fore et al., 1996).

Urbanization of a watershed results in multiple stresses to streams, including lower summer base flows and loss of large woody debris (Finkenbine et al., 2000), while increasing mean annual flow (DeWalle et al., 2000). Percent of impervious area summarized multiple consequences of urbanization (Karr and Chu, 1999). In a study of fish communities in southeastern Wisconsin streams, Wang et al. (2000) found that a threshold of environmental damage occurs at about 10 percent imperviousness. Since 1945, urbanization in the Indianapolis area has proceeded along a corridor extending from the White River upstream to the tributaries, and small streams were the most common sties for construction (White, 1996). Many urban streams in Indianapolis show signs of instability in the form of incision and channel widening (Doyle et al., 2000).

Regional differences must be accounted for when evaluating the results of a multimetric macroinvertebrate biomonitoring protocol. Previous studies of bioassessments have included streams in Florida (Barbour et al., 1996), Oregon (Fore et al., 1996) and the mid-Atlantic coastal plain (Maxted et al., 2000). In diverse regions, not only do the native invertebrate populations differ, but the human impacts also vary. In Florida, streams are degraded by non-point source pollution (Barbour et al., 1996), while those in Oregon are affected by logging (Fore et al., 1996).

One macroinvertebrate metric is taxa richness. Invertebrate species richness decreases as disturbance frequency increases (Robinson and Minshall, 1986). Total taxa richness has been found to be a suitable metric for discriminating among sites (Barbour et al., 1992; Kerans and Karr, 1994; Fore et al., 1996; Barbour et al., 1996; Maxted et al., 2000) and is expected to decrease with increased human impact (Kerans and Karr, 1994).

The proportion of the macroinvertebrate community with the primary habit designation as “clinger” has been used to evaluate stream condition (Richards et al., 1997; Maxted et al., 2000). Common and widespread insect taxa considered to be clingers (Merritt and Cummings, 1996) include the families Hydropsychidae, Heliopsychidae, Philopotamidae, and Hydroptilidae of the order Trichoptera (caddisflies); the family Heptageniidae of the order Ephemeroptera (mayflies); the family Elmidae of the order Coleoptera (beetles); and Simuliidae of the order Diptera (true flies). These taxa are expected to proportionally decrease with increasing physical disturbance to the stream habitat.

The orders Trichoptera and Ephemeroptera are generally considered to be sensitive to perturbation, and are included in the often employed EPT (Ephemeroptera-Plecoptera-Trichoptera) metric (e.g. Barbour et al., 1997; Maxted et al., 2000) which measures either the number or proportion of these taxa. A way to evaluate evenness within these orders is to calculate the ratio of Hydropsychidae/total Trichoptera and Baetidae/total Ephemeroptera, as the families Baetidae and Hydropsychidae are considered to be more tolerant of disturbance (Barbour et al., 1997)..

9.2 Field collection and analysis

A total of 17 sites in six creeks were sampled during July and August 1999. These were Crooked Creek, Little Buck, Little Eagle, Pleasant Run, and Williams in Marion County, and Indian Creek in Tippecanoe County. An additional creek, Fishback Creek in Marion County, was intended to be included but was not sampled due to abnormally low flow during the study period (United States Geological Service, 1999).

Only riffles were sampled during this study. The validity of single habitat sampling is supported by Angradi (1996) who found that all functional groups of macroinvertebrates were well-represented in riffles in Appalachian headwater streams

On each creek, three sites (upper, middle, and lower) were selected. At each site, three riffles were sampled for benthic macroinvertebrates and periphyton. Riffles had been selected previously for additional research concerning geomorphological characteristics, habitat quality, and fish populations. A total of 180 samples were collected for macroinvertebrates in 1999. The upper and lower sites on Little Buck Creek were not sampled due to no-flow conditions during 1999.

In 2000, the middle sites on Crooked Creek, Little Buck Creek, Little Eagle Creek Williams Creek, and Pleasant Run, and the middle and lower sites on Indian Creek were sampled again. In addition, middle and lower sites on Lick Creek, and upper sites on Fishback Creek and Little Buck Creek were sampled. The lower site on Little Buck Creek was not sampled due to being completely dry. Fishback Creek flows into Eagle Creek reservoir, and the lower site was not sampled because it was pooled.

In 2001, the middle and lower sites on Indian Creek were sampled to provide additional data to clarify differences in data between the previous two sampling years. The lower site on Fishback Creek was also sampled in 2001.

Macroinvertebrate sampling methodology was adapted from Karr and Chu (1999). Aquatic insects were collected with a 1 ft^2 Surber sampler having a mesh size of 500 μm.

Four samples were taken from each riffle, making a total of 12 samples at each site. Sampling began at the downstream end of the riffle and proceeded upstream until the four samples were taken. Two of the Surber sites were positioned on the upstream portion of the riffle, and two of the Surber sites were positioned on the downstream portion of the riffle. A flag was used to mark the spot of each Surber.

The Surber sampler was placed on the streambed with the opening of the net facing upstream, with the brass frame being held firmly on the substrate. With one person holding the brass frame underwater, another person washed the loose insects off the rocks into the net. The largest particle from each sample was saved, and the intermediate axis length measured. The remaining substrate was stirred with a small rake to a depth of 10 centimeters to loosen organisms in the interstitial spaces. The net was rinsed thoroughly after each sample to prevent cross-contamination.

After collection, the sample was transferred to 250 milliliter Nalgene bottles. The Surber sampler net was examined carefully, and any remaining clinging insects were removed and added to the sample bottle. Once the sample was completely transferred to the sample jar, it was preserved with 70% ethanol. The samples were labeled for site, riffle, and replicate number on the outside of the jar with a water-proof marker, and on the inside of the jar with pencil on a slip of water-proof paper.

Macroinvertebrates in samples were separated from debris such as sand and leaves by hand-sorting with the aid of magnifying lenses. Karr and Chu (1999) suggest that 100 organism sub-samples, such as those used by the Rapid Bioassessment Protocol, are inadequate for multi-metric assessments, as rare taxa are likely to be missed.

Identifications were made using a dissecting microscope (10X) to the level of genus without the use of subsampling (except for Chironomidae) using the taxonomic keys of Pennak (1978), Merritt and Cummins (1996), and McCafferty (1983). Chironomids were subsampled and identified to genus by mounting the head capsules on microscope slides and examining at 40X. The entire insect collection for this project is archived in the Department of Forestry and Natural Resources at Purdue University.

Calculation of metrics and watershed-level analysis by Single-factor ANOVA: At each site, data from individual samples were combined to give a mean proportion and then metrics were calculated from these mean proportions. Metrics included proportion of clingers, scrapers, filterers/collectors, EPT organisms, chironomids, dominant two taxa, and Diptera, and ratios of Hydropsychidae/Trichoptera and Baetis/Ephemeroptera. Other metrics considered but eliminated during the early phases of the study due to very low numbers included proportions of omnivores, predators, and shredders. For 1999 data, single-factor ANOVA was performed on each metric by considering each site in a watershed to be a replicate of a treatment.

Site-level analysis of metrics versus percent imperviousness: Using percent imperviousness as the independent variable, various metrics were tested with regression. These metrics include numbers of non-chironomid and EPT genera, and family-level Hilsenhoff indices (Hilsenhoff, 1987). Proportions of filters/collectors, Diptera, scrapers, chironomids, and clingers were also tested, as was the ratio of *Baetis* to total Ephemeroptera, and the number of insect families for middle and lower sites.

Correlation: To determine which metrics would provide redundant information with other metrics, a correlation matrix was analyzed. A positive or negative correlation coefficient of greater than 0.75 indicated that two metrics provided redundant information. Metrics included in the correlation matrix were family Hilsenhoff index, number of insect families, number of EPT genera, number of non-chironomid genera, proportion clinger taxa, ratio of *Baetis* to total Ephemeroptera, proportion Chironmidae, and proportion total Diptera. Correlation matrices were done separately for 1999 and 2000 data.

Regression with Johnson index of channel stability: Regression analyses were performed using a modified Johnson index of channel stability (Johnson et al., 1999) as the independent variable, and proportion clingers, and percentage of the first and second dominant taxa as the dependent variables. Modification of the Johnson index is described in Doyle et al. (2000).

Logistic regression: Logistic regression was used to test for the probability that rare taxa (present at only one site per sampling year) will occur as a linear function of site imperviousness.

The number of clinger taxa was converted into a dichotomous variable as in Richards et al. (1997) by designating those sites with numbers of taxa less than or equal to the median (7) to be low frequency, and those sites with greater than 7 clinger taxa to be high frequency. Data from the two sampling years could not be combined because Little Buck Middle changed classification from low to high frequency from 1999 to 2000. Logistic regression was used to test the probability that the frequency of clinger taxa will occur as a linear function of site imperviousness for both 1999 and 2000 data. Because several sites became dry during the warm months of 1999, a logistic regression analysis was also performed on the high or low frequency of clinger taxa in 2000 versus the dichotomous independent variable of dry or not dry (perennial flow) the previous year.

Inter-annual variability of metrics: During the 2000 sampling of Indian Creek, the riparian area of the lower site was experiencing construction for silivicultural activities. The middle site of Indian Creek was sampled in 2000 shortly after a high water event. Because of the large variability in data for Indian Creek middle and lower sites between 1999 and 2000, those sites were sampled again in 2001. Because data for 2001 were similar to 1999, it was considered that the circumstances during the time of sampling in 2000 created some anomalous results. Regression analyses were repeated for proportion of clinger taxa with out data from Indian Creek lower.

Metrics were compared for sites that were repeated in 2000. Data from 1999 and 2000 were compared for replicated sites for the metrics proportion of clinger taxa, and the proportion of *Baetis* to total Ephemeroptera. Because there was no significant difference between the two sampling years, regression analyses were also performed on these metrics with combined data for the two years.

9.3 Results

A total of 35 insect families (Table 7) were found in the 8 watersheds sampled. No organisms of the order Plecoptera (stoneflies) were found during the collections.

Table 7. Summary of insect families present, indicated by x, by watershed for all collections. (IC=Indian Creek; Wm=Williams Creek; Cr=Crooked Creek; LB=Little Buck Creek; LE=Little Eagle Creek; Li= Lick Creek; PR=Pleasant Run; FB= Fishback Creek)

		IC	Wm	Cr	LB	LE	Li	PR	FB
Trichoptera	Hydropsychidae	x	x	x	x	x	x	x	x
	Philoptamidae	x	x	x	x	x		x	
	Polycentropodidae	x							
	Heliopsychidae	x							
	Lepdisotomidae	x							
	Hydroptilidae	x	x		x	x	x	x	x
Ephemeroptera	Heptageniidae	x	x	x	x	x	x	x	x
	Baetidae	x	x	x	x	x	x	x	
	Caenidae	x	x	x	x	x	x	x	x
	Oligoneuridae	x				x			
	Tricorythidae	x			x	x			
Coleoptera	Chrysomelidae		x						
	Elmidae	x	x	x	x	x	x	x	x
	Heliodidae	x							
	Psephenidae	x	x						
	Hydrophilidae	x							x
	Heteroceridae		x						
Diptera	Chironomidae	x	x	x	x	x	x	x	x
	Simuliidae	x	x	x	x	x	x	x	

	Tabanidae	x							
	Empididae	x	x	x	x	x	x	x	
	Stratiomyidae	x							
	Tipulidae	x	x	x	x	x	x	x	x
	Ephydridae					x		x	
	Culcididae	x							
	Dolichopodidae	x				x		x	
Odonata	Aeshnidae	x							x
	Coenargrionidae	x	x			x			
	Calopterygidae	x	x			x	x		
	Gomphidae	x							
	Macromidae		x						
Megaloptera	Corydalidae	x							
	Sialidae	x	x				x		
Lepidoptera	Pyralidae				x				
Collembola	Isotomidae	x							

1999 collections ANOVA and linear regression: Using single-factor ANOVA of sites grouped by watershed (Table 8), the most significant differences were revealed by the proportion clingers and the ratio of *Baetis* to total Ephemeroptera, each with p values of less than 0.0001. Also showing significant differences were proportion chironomids, proportion Diptera (Figure 18), and proportion of filterers/collectors.

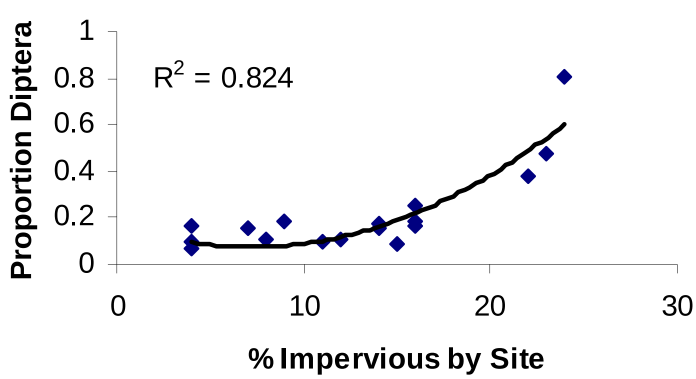

Figure 18. Dipteran proportion increases with imperviousness.

Table 8. Single-factor analysis of variance of site mean proportions grouped by watershed for 1999 collections.

Metric tested	**P value**
Proportion clingers	< 0.001
Baetis/Ephemeroptera ratio	< 0.001
Proportion chironomids	0.001
Proportion Diptera	0.002
Proportion Filters/collectors	0.01
Percent of the dominant taxon	Ns
Proportion predators	Ns
Proportion EPT taxa	Ns
Hydropsychidae/Trichoptera ratio	Ns
Proportion scrapers	Ns
Proportion shredders	Ns

Regression of metrics using site imperviousness as the independent variable (Table 9) showed that many metrics respond to increased imperviousness. The proportion of Diptera, family Hilsenhoff index, the ratio of *Baetis* to total Ephemeroptera, and proportion of Chironomidae increased significantly, while the proportion of clinger taxa, number of non-chironomid genera, number of Ephemeroptera, Plecoptera and Trichoptera (EPT) genera and proportion of filterers/collectors significantly decreased. The proportion of scrapers showed no significant trend.

Table 9. Site-level regression of insect metrics vs. percent imperviousness for 1999 collections.

	Expected response to increasing imperviousness	**R^2**	**p**
Proportion Diptera *	Increase	0.84	< 0.001
Proportion clinger taxa	Decrease	0.76	< 0.001
Family Hilsenhoff Index *	Increase	0.74	< 0.001
Insect taxa richness lower and middle sites	Decrease	0.60	0.005
Baetis/Ephemeroptera ratio	Increase	0.49	0.002
Proportion Chironomids	Increase	0.48	0.002
Number non-chironomid genera	Decrease	0.45	0.003

Number EPT genera	Decrease	0.35	0.012
Proportion filterers/collectors	Decrease	0.30	0.023
Proportion scrapers	Decrease	0.15	ns

* Second order quadratic function

2000 Collections linear regression: Using regression analysis for 2000 data, the ratio of *Baetis* to total Ephemeroptera significantly increased with increasing imperviousness. Other metrics tested showed no significant trend. Analyses were affected by one of the rural sites, Indian Creek lower, experiencing disturbance in the form of silvicultural construction activities, possibly leading to a dramatic increase in the proportion of chironomids, from 5% to 41%, in one year. However, the 2001 collections contained 8% Chironomidae, indicating a return to previous conditions. When regression analyses were repeated without the data from Indian Creek lower, the proportion of clinger taxa showed a significantly (r2 = 0.39, p=0.05) decreasing trend with increasing imperviousness. However, the proportion of Diptera, proportion of chironomids, proportion filterers, number of non-chironomid genera, number of insect families, and family Hilsenhoff biotic index still showed no significant trend.

Indian Creek middle was sampled in 2000 shortly after a period of high water. Repeat sampling in 2001 produced similar data to 1999, indicating that the year 2000 sampling effort may have been negatively affected by the recent high flows. Overall proportions of organisms in 2000 were similar to 1999, although the diversity and abundance was less. Repeating regression analyses for 2000 on the number of insect families and number of non-chironomid genera with out data from this site still did not show a statistically significant trend to increasing imperviousness (Table 10).

Table 10. Site-level regression of insect metrics vs. percent imperviousness for 2000 collections.

	Expected response to increasing imperviousness	**R^2**	**p**
Baetis/Ephemeroptera ratio	Increase	0.48	0.018
Proportion clingers	Decrease	0.19	ns
Number of non-chironomid genera	Decrease	0.10	ns
Number of insect families	Decrease	0.04	ns
Percent filterers/collectors	Decrease	0.02	ns
Number EPT genera	Decrease	0.12	ns
Proportion Diptera	Increase	0.003	ns
Hilsenhoff Family Biotic Index	Increase	0.001	ns
Proportion Chironomids	Increase	0.06	ns

Linear regression against Johnson index of channel stability: For 1999 data, regression of the Johnson index of channel stability (Table 11) against the proportion of clingers showed a decrease at more unstable channels, while the proportion of the first and second dominant taxa increased. However, for 2000 data, no significant trends were seen, even with the exclusion of the anomalous data from Indian Creek lower.

Table 11. Regression of Johnson index of channel stability against proportion clingers and proportion of the dominant two taxa. Data from 2000 were analyzed both with and without Indian Creek lower site, which had experienced a short-term disturbance before insects were sampled.

	Response to instability	**1999**		**2000**		**2000 without lower Indian**	
Metric		**R^2**	**p**	**R^2**	**P**	**R^2**	**p**
Proportion Clingers	Decrease	0.37	0.01	0.10	Ns	0.09	ns
Proportion 1st and 2nd dominant	Increase	0.35	0.01	0.06	Ns	0.01	ns

Logistic regression: For 1999 data, there was a significant (p=0.04) probability that rare taxa would be present as a linear function of site imperviousness. For 2000 data, the probability was nearly significant (p=0.08). The probability of the frequency of numbers of clinger taxa as a linear function of site imperviousness was nearly significant (p=0.06) for 1999 data, but was not significant for 2000 data. The probability of the frequency of numbers of clinger taxa as a linear function of whether sites were dry in 1999 was not significant (p=0.14).

Comparison of 1999 and 2000 insect data: Sites that were sampled in both 1999 and 2000 are compared for key metrics in Tables 12 through 19. Among replicated sites, the caddisfly family Hydropsychidae was either first or second dominant family in five out of seven sites in 1999 (Table 12), and six out of seven in 2000 (Table13). Only Little Buck middle did not have Hydropsychidae as either first or second dominant family.

Table 12. First and second dominant families and percentage of each for 1999 for replicated sites.

	1999			
Site	First dominant family	%	Second dominant family	%
Indian Creek-middle	Hydropsychidae	53	Chironomidae	12
Indian Creek-lower	Hydropsychidae	67	Tricorythidae	10
Pleasant Run-middle	Chironomidae	46	Baetidae	36
Little Buck-middle	Elmidae	57	Baetidae	19
Crooked-middle	Hydropsychidae	37	Baetidae	29
Williams-middle	Hydropsychidae	63	Baetidae	11
Little Eagle-middle	Hydropsychidae	37	Baetidae	27

Table 13. First and second dominant families and percentage of each for 2000 for replicated sites.

	2000			
	First dominant family	%	Second dominant family	%
Indian Creek-middle	Hydropsychidae	66	Chironomidae	8
Indian Creek-lower	Chironomidae	41	Hydropsychidae	31
Pleasant Run-middle	Baetidae	37	Hydropsychidae	33
Little Buck-middle	Elmidae	26	Simulidae	24
Crooked-middle	Hydropsychidae	34	Simuliidae	30
Williams-middle	Hydropsychidae	58	Chironomidae	12
Little Eagle-middle	Hydropsychidae	52	Baetidae	17

Table 14. Combined percentage of first and second dominant families for replicated sites, and percent change from 1999 to 2000.

Site	**1999**	**2000**	**change**
Indian Creek-middle	65	74	+ 9
Indian Creek-lower	77	72	-5
Pleasant Run-middle	82	73	-9
Little Buck-middle	76	50	-26
Crooked-middle	66	64	-2
Williams-middle	66	64	-2
Little Eagle-middle	64	69	+ 5

Table 15. Percentage of clinger taxa for replicated sites in 1999 and 2000, and percent change.

	Clingers		
Site	1999 (%)	2000 (%)	change
Indian Creek-middle	70	81.9	+11.9
Indian Creek-lower	75	47.7	- 27.3
Pleasant Run-middle	16.0	45.0	+ 29
Little Buck-middle	70.0	63.5	- 6.5
Crooked-middle	62.0	81.0	+ 19
Williams-middle	74.0	78.7	+ 4.7
Little Eagle middle	62.0	74.3	+ 12.3

Table 16. Number of insect families and family level Hilsenhoff Biotic Index (HBI), including water quality category, for replicated sites in 1999 and 2000, and percent change.

	Number insect families			Family HBI		
Site	1999	2000	change	1999	2000	change
Indian Creek-middle	17	8	-9	4.09 v. good	4.10 v. good	+ 0.01
Indian Creek-lower	17	20	+ 3	4.01 v. good	4.81 good	+0.80 *
Pleasant Run-middle	10	9	-1	4.91 good	4.40 good	- 0.51
Little Buck-middle	11	9	-2	4.12 v. good	5.09 fair	+ 0.97 *
Crooked-middle	10	9	-1	4.22 v. good	4.69 good	+ 0.47 *
Williams-middle	12	12	+1	3.99 v. good	4.26 good	+0.27 *
L. Eagle-middle	10	11	+ 1	4.14 v. good	4.38 good	+0.24 *

* = different water quality category

Table 17. Percent chironomids and number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) genera for replicated sites for 1999 and 2000, and percent change of each.

	Chironomid %			**Number EPT genera**		
Site	1999	2000	change	1999	2000	change
Indian Creek-middle	12	8	-4	8	4	-4
Indian Creek-lower	5	41	+36	9	12	+3
Pleasant Run-middle	46	14	-32	5	6	+1
Little Buck-middle	7	20	+13	7	7	0
Crooked-middle	8	5	-3	6	6	0
Williams-middle	4	12	+8	7	7	0
Little Eagle	7	6	+1	7	7	0

Table 18. Percent filterers and Diptera for replicated sites for 1999 and 2000, and percent change of each.

	Filterers %			Diptera %		
Site	1999	2000	Change %	1999	2000	Change %
Indian Creek-middle	63.7	71.7	+ 8.0	16.2	13.5	- 2.7
Indian Creek-lower	70.4	34.9	- 35.5	6.9	43.8	+ 36.9
Pleasant Run-middle	11.9	42.2	+ 30.3	47.8	20.9	- 26.9
Little Buck-middle	13.1	35.8	+22.7	9.1	43.9	+34.8
Crooked-middle	41.8	64.5	+ 22.7	17.2	38.4	+21.8
Williams-middle	75.6	65.5	-10.1	10.2	21.3	+11.1
Little Eagle	40.0	67.9	+27.9	8.3	20.3	+12.0

Table 19. Percent Ephemeroptera and ratio of *Baetis* to total Ephemeroptera for replicated sites for 1999 and 2000, and percent change of each.

	Ephemeroptera %			Baetis/Ephemeroptera %		
Site	1999	2000	Change %	1999	2000	Change %
Indian Creek-middle	21.1	8.9	-12.2	30.1	76.2	+ 46.1
Indian Creek-lower	20.3	10.3	-10.0	21.4	43.3	+21.9
Pleasant Run-middle	39.4	40.5	+1.1	92.5	97.7	+5.2
Little Buck-middle	20.9	15.3	-5.6	90.8	93.1	+2.3
Crooked-middle	32.8	12.6	- 20.2	89.5	95.4	+5.9
Williams-middle	11.9	3.7	- 8,2	94.9	94.2	- 0.7
Little Eagle	29.9	17.3	- 12.6	90.2	96.6	+6.4

Overall trends: When combining data from all sampling years, the proportion of clingers decreased significantly (r2=0.70, p <0.001) in a linear function as imperviousness increased. A weaker relationship (r2=0.25, p=0.02) existed between decreasing proportion of clingers and increasing channel instability as indicated by the Johnson index. The proportion of *Baetis* to total Ephemeroptera increased significantly (r^2=0.76, p<0.001) in a log function as the imperviousness increased.

9.4 Discussion of indicators

Choice of independent variable: Imperviousness was a more useful predictor variable than the Johnson index of channel stability. Doyle et al. (2000) found that the Johnson index showed little difference between seven of the sites in the present study that ranged from 0% to 32% high-density urban land use.

One of the most urban sites, Little Eagle-lower, was rated as the most stable, in part due to rip-rap protected levees (Doyle et al., 2000).

EPT metrics: Analysis of variance did not reveal any significant differences in proportion of EPT taxa between watersheds. Using linear regression, the number of EPT genera present showed a weak relationship with imperviousness in 1999, and was not significant in 2000. Although the number of EPT genera in replicated sites generally showed good agreement from 1999 to 2000, Indian Creek middle showed a net loss of four genera, from eight to four. However, there were 10 EPT genera in the 2001 collections from this site. This was likely caused by the timing of the 2000 collection shortly after a period of high flows that may have dislodged many organisms. The EPT metric was further weakened by the total lack of Plecoptera in the study sites. Most of the macroinvertebrate communities in the streams in the study were also similar in that they had large representations of the Hydropsyschidae genera *Hydropysche* and *Cheumatopsyche*. The ratio of Hydropsychidae to total Trichoptera failed to distinguish between sites.

Family-level Hilsenhoff Biotic Index: All sites scored as very good or good, with the exception of Pleasant Run lower in 1999 and Little Buck middle in 2000, which scored as fair, indicating water quality was not the primary limiting factor in determining macroinvertebrate communities. Regression of family-level Hilsenhoff index against site imperviousness was significant for 1999 data, however, no relationship was seen for 2000 data. As this index was developed in Wisconsin, Resh and Jackson (1993) recommend being aware of regional differences when using this index. While the family-level Hilsenhoff biotic index has many advantages in water quality studies, it is not in itself adequate to detect changes brought about by increased watershed imperviousness from urbanization.

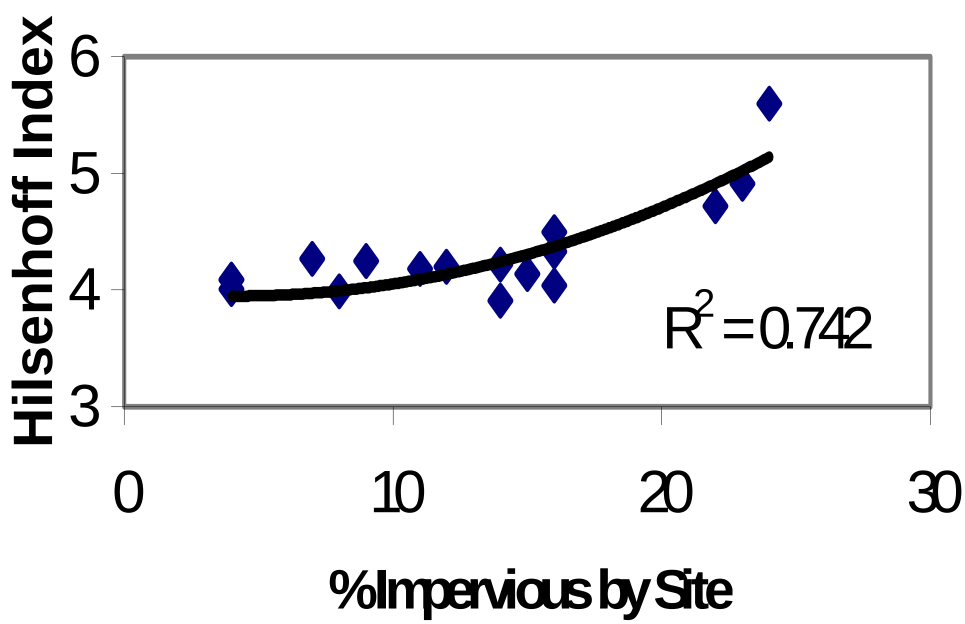

Figure 19. Trend in Hilsenhoff Family-level index with imperviousness.

Baetis/ Total Ephemeroptera: There was a higher occurrence of *Baetis* relative to total Ephemeroptera in the more urbanized streams. *Baetis* mayflies were found to be more resistant to spates than other common taxa in prairie streams (Miller and Golladay, 1996). Baetis spp. may have ten generations per year, and are streamlined and capable of short bursts of rapid swimming (Merritt and Cummings, 1996). Even if a spate decreases their numbers in a stream, they can rapidly recolonize after a disturbance (Allan, 1975; Fisher et al., 1982). Robinson et al. (1990) suggest that differences in dispersal abilities of lotic macroinvertebrates influences the time needed for recolonization after a disturbance, and that organisms that disperse by crawling will be slower colonizers than *Baetis*.

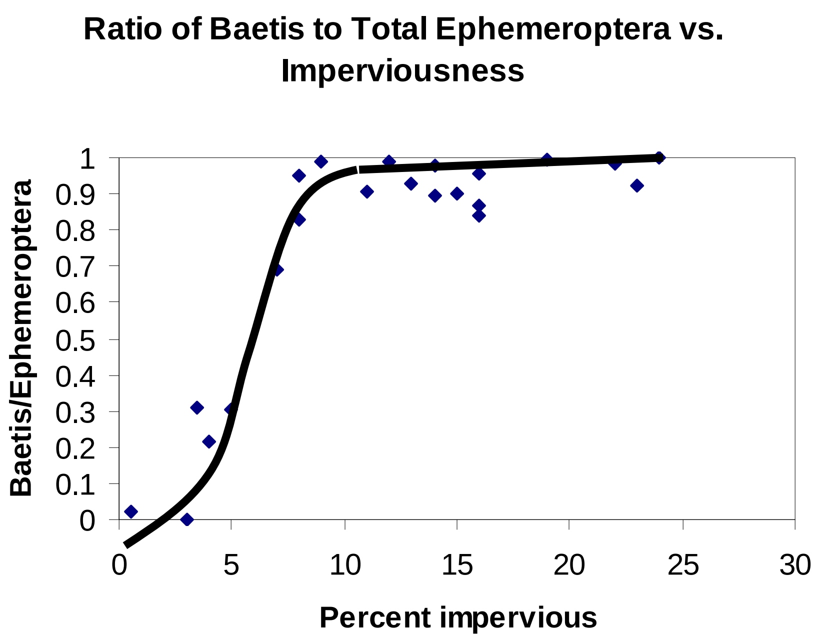

Figure 20. Ratio of Baetis to total Ephemeraoptera as a function of imperviousness. The trend line predicts a 50% Baetis ratio at approximately 6% imperviousness.

In less urbanized streams, other mayfly families such as Heptageniidae comprise a higher percentage of the total Ephemeroptera . In a study of the common heptageniid, Stenocron interpunctatum (McCafferty and Huff, 1978) in Wildcat Creek (north central Indiana), the population had generation times varying from one per year to three every two years. The number of generations of mayflies per year can be influenced by local flow and temperature conditions (Rader and Ward, 1989; Jacobi and Benke, 1991).

Seasonal effects need to be taken into consideration when assessing the proportion of *Baetis*. In Idaho, Robinson et al. (1990) found a decline in relative abundance from summer to fall in a Baetis species. In the present study, sampling at the highly urbanized Pogues Run upper site in late November, 1999 produced 0% *Baetis,* while repeat sampling at the same site in July, 2000 showed that Baetis was the predominant taxon, at 43% of the total macroinvertebrate collection.

Clinger taxa: The proportion of clinger taxa decreased as the degree of imperviousness increased. This is likely the result of higher peak flows associated with increased imperviousness and urbanization. Filterers such as Hydropsychidae are exposed to current to obtain food, while scrapers such as Heptageniidae may be dislodged into the current from their high-risk, exposed, periphyton-rich areas (Merritt and Cummings, 1996). The probability that high numbers of clinger taxa would occur decreased as imperviousness increased, although this was not significant at the 0.05 level. More data points would likely increase the statistical significance.

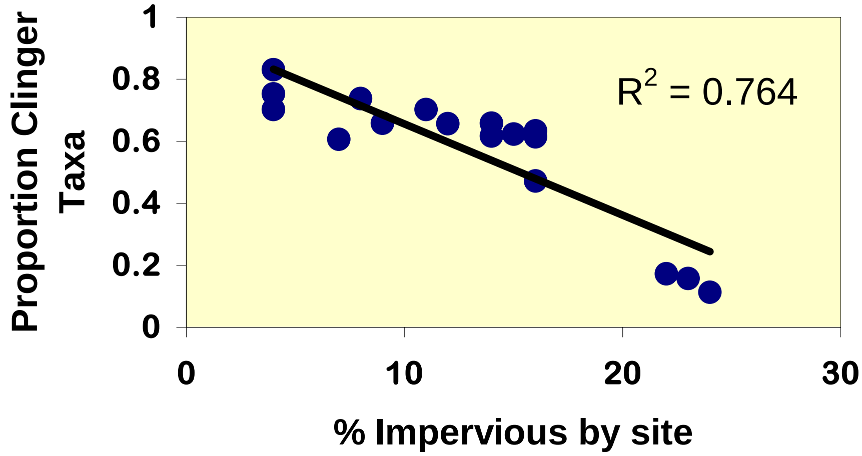

Figure 21. Decline in clinger proportion with increasing imperviousness**.**

Occasionally intermittent streams: For the year 2000 collections, some of the metric results were unexpected for some of the less urbanized streams. The year 1999 was exceptionally dry, and three of the less urban sites (Little Buck upper, Fishback upper, Indian Creek lower) were dry for an extended period. Feminella (1996) suggests that streams that are occasionally intermittent (riffles stop flowing in dry years) can affect insect populations the next year. Feminella also suggests that year-to-year differences in riffle permanence may account for year-to-year differences in insect assemblages. Miller and Golladay (1996) found that the genera *Cheumatopsyche*, *Chimarra*, and *Baetis* are flow-dependent and therefore limited in intermittent streams, while *Stenelmis* and Stenonema were able to utilize such habitats. Temporal variation due to an unusually dry sampling year affected feeding ecology metrics in a study by Fore et al. (1996) to the extent that they failed to distinguish most and least disturbed sites. The probability that clinger taxa was less likely to occur with high frequency in streams that had been dry at some point the previous year was not statistically significant, although a larger data set might reveal a relationship.

Metrics recommended to evaluate urbanization: Increased imperviousness was most clearly associated with a decrease in the proportion of clinger taxa, and an increase in the ratio of *Baetis* to total Ephemeroptera. The ratio of *Baetis* to total Ephemeroptera was not highly correlated to the other commonly used metrics in this study, and therefore would not provide redundant information. The proportion of clingers was highly negatively correlated with the family Hilsenhoff index, proportion Diptera, and proportion of chironomids in the 1999 data set, but not the 2000 data set and might be redundant with those metrics. However, the proportion of clingers shows a more direct response to increasing imperviousness, and should be included in any index designed to assess the impact of urbanization.

10 Stream Fish Assemblages as Indicators of Urbanization

This section represents the work of Cecil Rich. A thesis is currently in preparation.

10.1 Introduction

The importance of physical environmental variability on species diversity and abundance is well established in the ecological literature (Connell 1978; Sousa 1979; Pickett and White 1985; Petraitis 1989). Spatial habitat heterogeneity and temporal environmental variability in stream systems are thought to provide a habitat template upon which biotic communities are organized (Southwood 1977, 1988; Minshall 1988; Poff and Ward 1990). In the conceptual framework provided by a habitat template, communities present at a site are thought to reflect long-term constraints on the types of species attributes appropriate for local persistence. Poff and Ward (1990) recommend the minimum elements necessary for describing lotic physical habitat templates are the long-term temporal pattern of physical variability (streamflow patterns) in combination with substrate heterogeneity and stability. These variables reflect the commonly held perception that habitat disturbance generated by variable stream flow regimes are dominant factors generating disturbance in many lotic systems (Bain et al. 1988; Poff and Ward 1989; Poff et al. 1997).

Highly variable discharge regimes in streams provide excellent opportunities to study the ecological impacts of flow-related disturbance in structuring lotic communities (Resh et al. 1988; Poff and Ward 1989, 1990; Reice et al. 1990; Poff and Allan 1995; Townsend et al. 1997). Flow extremes have been shown to directly influence local assemblage structure of a wide range of biotic components of stream ecosystems including: fish (Horwitz 1978; Meffe 1984; Poff and Allan 1995), macroinvertebrates (Quinn and Hickey 1990; Fisher and Grimm 1991; Death and Winterbourn 1994), macrophytes (Henry et al. 1996), periphyton (Biggs 1995), and microcrustacea (Robertson et al. 1997). Measures of disturbance that have been used in the majority of studies include: hydrological variables related to flow variability (Horwitz 1978; Jowett and Duncan 1990; Poff and Allan 1995), flood magnitude (Hendricks 1995), and flood frequency (Quinn and Hickey 1990; Scarsbrook and Townsend 1993; Poff and Allan 1995). Additionally, several studies have quantified disturbance to stream benthic communities using a measure of stream bed movement during high discharge events (McElravy et al. 1989; Rader and Ward 1989; Cobb et al. 1992, Death and Winterbourne 1994, Scarsbrook 1995). This approach estimates the depth of flow required to initiate bed movement from an empirical relationship relating substrate size to the force required to move that size particle.

Several studies have found clear differences in the functional composition of fish communities in streams having stable flow regimes versus those having more variable flow regimes (Horwitz 1978; Bain et al. 1988; Poff and Allan 1995). As hydrological variability changed from stable to variable in these systems, fish assemblages varied from specialist species to resource generalists (Poff and Allan 1995). These studies assumed that hydrologic variability was directly or indirectly responsible for the changes in fish assemblage structure but did not test the importance of other potentially important factors under the control of flow regime such as local habitat structure and channel stability. Reductions in habitat diversity have been found associated with land use changes and these changes have been associated with reductions in commonly used habitat elements for some sizes and species of fish (Rabeni and Jacobsen 1993).

In addition to the flushing effects of high and low discharge in streams, other flow-related factors such as habitat structure, frequency of bed disturbance, and channel stability may also influence fish assemblage structure. For example, bankfull discharge events have implications for channel adjustment and stabilities (Carling 1988) while discharge magnitudes and frequencies are thought to determine bedload transport and channel adjustment (Wohlman and Miller 1960). Species which have life history traits dependent on stable substrates such as benthic-dwelling fish or those that spawn on gravel substrate may be particularly sensitive to flow related disturbance (Coon 1987; Limburg and Schmidt 1990). Schlosser (1982) found upstream – downstream gradients in fish abundance and size in headwater streams that he related to differences in habitat heterogeneity and pool depth. Sampling of fish communities before and after floods have shown early life history stages affected more than older fish (Schlosser 1985) with this effect dependent on the timing of reproduction relative to the occurrence of flood events (John 1964; Harvey 1987; Pearsons et al. 1992).

Our objectives in this study were to assess the influence of hydrologic variability in streams on habitat structure, channel morphology, and fish assemblage structure. We hypothesize that streams with more variable hydrologic regimes will have less stable and complex habitats which in turn lead to loss of specialist fish species lacking adaptations which permit persistence in variable environments and a higher proportion of generalist fish species having combinations of life history characteristics that promote persistence in variable environments.

10.2 Sampling and analysis methods

Sites were selected to include varying degrees of urbanization within drainage basins as well as on the basis of availability of stream gauging data and practical logistical factors (e.g., accessibility). Biological and physical data were collected for a 200-m stream segment at each site. We included in the sampling segment the best ecological habitat available as indicated by the presence of a riparian corridor and physically complex habitats (e.g., deep pools and a variety of substratum types). Riparian vegetation armors stream banks, reduces bank erosion, promotes development of scour holes (Keller and Swanson 1979), reduces stream channel siltation (Rabeni and Smale 1995) and moderates chemical and nutrient inputs (Osborne and Kovacic 1993); the importance of complex habitats in supporting diverse and stable biotic assemblages is also well documented (Schlosser 1982; Schlosser 1987). Obvious point sources of pollution such as treated sewage effluents were avoided. Within each watershed three 200-m study reaches spaced approximately equidistantly along the length of each watershed were assessed for land use, geomorphic, habitat, and biotic characteristics. Land use characteristics of watersheds were defined to quantify the extent and location of urban areas. Sampling methods for each set of variables are described below.

10.2.1 Fish community sampling methods

Fish community composition was determined in each 200-m stream reach during base flow conditions with a 2-person crew. This involved single-pass backpack electrofishing with a Smith-Root Model 12 battery powered electrofisher. To insure that small benthic fish such as darters were not missed, care was taken to slowly and methodically sample areas of complex cover such as large riffle substrate and deep pools. Fish were identified, enumerated, and measured, and then returned to the stream. A reference collection of representative species was archived to document proper species identification at the Purdue University Forestry Department. We determined fish assemblage variables from the original species data using techniques such as the Index of Biotic Integrity (IBI) and variables representing species traits were derived from a range of published sources (Pflieger 1975; Trautman 1981; Ohio EPA 1989; Detenbeck et al. 1992; Poff and Allan 1995). Continuous variables such as the morphological traits were calculated as an average of the value for each species present at a site while categorical variables such as functional feeding group were calculated as a proportion. For example, if five of ten species at a site were benthic invertivores, that site would receive a value of 0.5 for the benthic invertivore feeding group variable. Continuous variables were calculated as an average of literature values for species present at a site.

10.2.2 Physical habitat characterization

We assessed physical habitat at each survey site following biotic sampling. A suite of variables was measured that included those commonly used to characterize stream habitat (Table 20). These included measures of longitudinal habitat dimensions, woody debris, bank condition, substrate characteristics, and a qualitative habitat index (Qualitative Habitat Evaluation Index (QHEI); Ohio EPA 1989). This method rates a stream reach based on six metrics that include substrate, instream cover, channel morphology, riparian zone and bank erosion, pool and riffle quality, and map-derived gradient. The total QHEI score is derived by adding the components of each metric and then summing the metric scores to obtain the total score

Table 20. Site habitat variables measured at each study reach.

Variable	Methods
% shallows	Visual estimate of the percentage of wetted area < 10 cm in depth
% deep pools	Visual estimate of the percentage of wetted area as pools > 0.5 m in depth
Maximum depth	Depth of deepest part of stream reach
% pools	Measure of the percentage of reach length composed of pool habitat
% riffles	Measure of the percentage of reach length composed of riffle habitat
Pool depth	Measure of the average depth of pools
% eroded banks	Visual estimate of the percentage of banks that were washed or sloughing
Woody debris	Length of large woody debris >10 cm diameter per 100 m of stream channel

Channel geomorphic stability was assessed using both qualitative and quantitative methods to provide a measure of physical disturbance to stream ecosystems. Qualitative methods used included the Pfankuch (1978) method and the Johnson et al. (1999) method. Both methods incorporate qualitative ratings of channel stability characteristics while the Johnson et al. method combines qualitative ratings with simple hydraulic and sediment transport calculations. Visual assessments of stability indicators were made along the 200 m reach where biotic and habitat assessments were conducted. Quantitative measures of channel geomorphic condition were calculated based on standard sediment transport equations relating channel morphological characteristics including cross-sectional area, slope, and substrate composition to quantitative descriptors of the channel (Table 21). Quantitative measures were calculated as a reach average for field measurements made at three cross sections located at riffles within each 200m reach. Methods used in calculating these variables as well as an application of these methods to a subset of the current study watersheds are detailed in Doyle et al. (2000).

Table 21. Quantitative descriptors of channel stability.

Variable	Description
Boundary shear stress	the shear exerted on the bed during bankfull events
Critical shear stress	the shear needed in order to initiate motion of the bed

Critical discharge	the discharge needed to initiate bed motion
Excess shear stress	the ratio of boundary to critical shear stress
Stream power	the rate of doing work

10.2.3 Analytical methods

We used the fourth-corner problem method to relate environmental features of streams to functional characteristics of fish communities. This is a nonparametric permutation technique for directly relating biological and behavioral characteristics of animals to habitat characteristics of the locations at which they are found (Legendre et al. 1997; Legendre and Legendre 1998). While previous techniques for analyzing data of this type generally involved relating species composition against habitat characteristics or against behavioral data, this method estimates parameters describing relationships between habitat characteristics and biology and behavior and testing for the significance of these relationships. Given three data matrices A, B, and C, with A being a species by locality matrix (p rows by n columns), B being a species by ecological trait matrix (p rows by q columns), and C being a habitat variable by locality matrix (m rows by q columns), this method provides a solution to a fourth matrix D, a habitat by ecological trait matrix (m rows by q columns); hence the name fourth-corner problem. The first table A contains data on the presence or absence of k species at m locations while the second table B describes n biological or behavioral traits of the k species, and the third table C contains environmental data on the m locations. While table A is necessarily presence/absence or frequency data, tables B and C can be either quantitative or qualitative data. The fourth-corner method provides a matrix algebra solution estimating the parameters in D by crossing the n biological or behavioral variables to the p environmental variables. The estimated parameters test whether the associations between variables significantly differ from 0 (no association), or from the values they could take if the environment were randomly organized. Because the random component tested in this study is the species found at the sampling sites, matrix A is permuted to allow hypothesis testing. Further, because I am assuming environmental control over individual species (i.e., species are found at locations having appropriate living conditions and select these locations independently of one another), the values within each row are permuted at random within each row (a random ordering of species each permutation).

The general method involves several steps that include formulating hypotheses, computing a test statistic, determining the distribution of the test statistic, and making the statistical decision. The null hypothesis is that the species (stream fish) are distributed randomly among the study sites while the alternative hypothesis is that the species are not distributed randomly among sites. There are two methods that can be used when computing a test statistic for quantitative (environmental characteristics) versus nominal variables (biological characteristics). In the first in which the values are not standardized, the fourth corner statistics are measures of within-group sums of squares that are divided by the total sums of squares to provide normalized measures of within-group homogeneity and take values from 0 to 1. A global statistic that can be calculated for all groups of the nominal variable, and is an F test, tests the null hypothesis that no groups are significantly different from the others in terms of the values of B to which it is associated, and whether at least one group is significantly more homogeneous than would be expected under random allocation of species to sites. In the second test, where values have been standardized to mean 0 and variance 1, the fourth-corner statistic become correlation coefficients between the quantitative variable and the binary descriptors (behavioral variable 1 if true and 0 if false for a species). The correlation coefficients indicate the strength of association of each state of the nominal variable to small or large values of the quantitative variable. To account for the increased probability of a Type I error in the case of multiple simultaneous tests for , we used Holm's (1979) procedure for adjusting individual probabilities. To deal with the problem of each state of the nominal variable being mutually exclusive, multiple regression was used where the quantitative variable was regressed on 1 less than the total number of states of the nominal variable. This provides an R^2 value to test the global hypothesis that at least one state of the nominal variable is significantly correlated to the quantitative variable.

10.3 Results

Physical habitat quality varied from 48.5 to 82.0 on the QHEI scale out of a possible 100. Fish communities ranged from "very poor" (IBI = 22) to "good" (IBI = 50). No "very good" or excellent sites were found. The results show a clear degradation of overall fish habitat quality (QHEI) in relation to urbanization as measured by watershed imperviousness (Figure 22). Overall quality of the fish community (as measured by IBI) likewise declines (Figure 23), although the correlations for biotic variables are lower than for physical variables. We are currently incorporating collection data from 2001 to evaluate a variety of possible metrics other than IBI for use in urbanizing streams (Tables 22 and 23). The proportion of darters and sculpins (Figure 24), for example, is significantly related to channel instability and % imperviousness. However work remains to be done to establish a direct relationship between hydrologic change (critical flow and variance) and specific biological indicators. This will require integration of the hydrology/water quality models discussed above as well as the geomorphic and stream community field data.

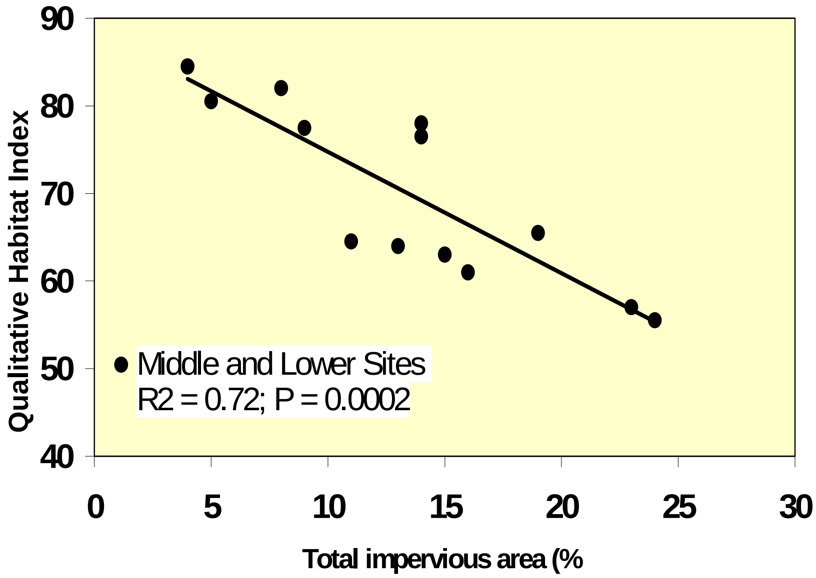

Figure 22. Decline in qualitative habitat index (QHEI) with increasing imperviousness**.**

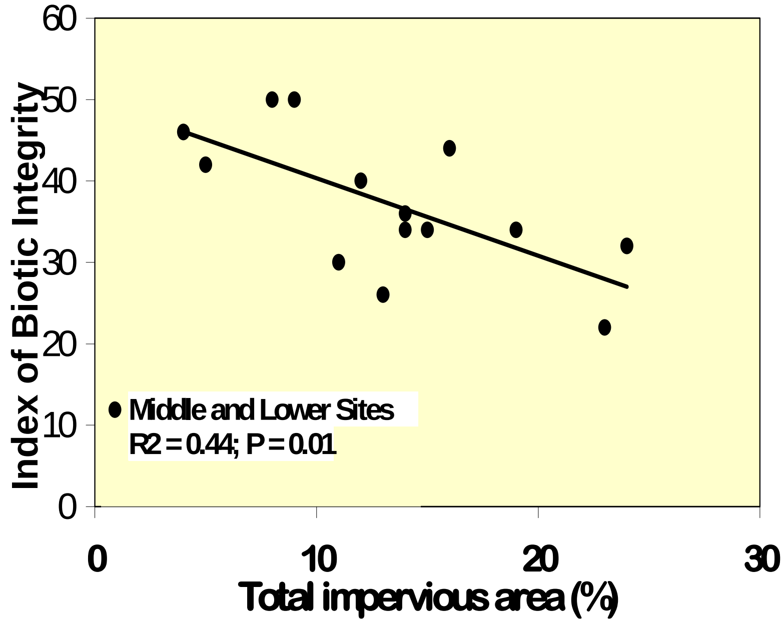

Figure 23. Decline in index of biotic integrity with increasing imperviousness.

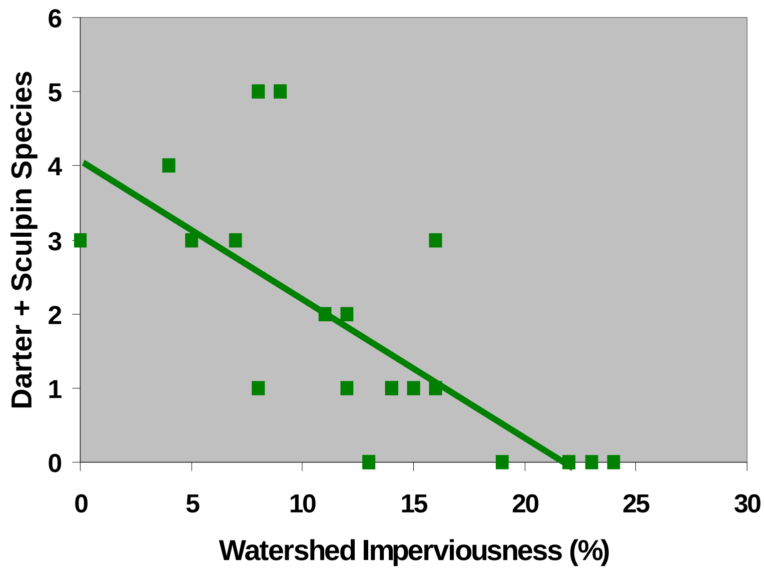

Figure 24. Decline in darters and sculpins with increasing imperviousness**.**

Table 22. Summary of stream sites sampled, drainage area, level of urbanization, and fish (IBI) and habitat indexes (QHEI). Site numbers (1,2,3) correspond to upper, middle, and lower sites respectively.

Stream	Site	Drainage Area, km^2	Wetted width, m	High density urban land use (%)	IBI	QHEI
Crooked Creek	1	18.16	2.3	38	30	57.5
Crooked Creek	2	31.10	5.5	37	34	76.5
Crooked Creek	3	40.88	5.6	35	36	78.0
Fishback Creek	1	23.52	4.5	9	24	72.5
Fishback Creek	2	45.54	7.9	10	30	71.5
Indian Creek	1	15.50	3.1	0	28	48.5
Indian Creek	2	39.00	5.9	0	42	72.5
Indian Creek	3	55.95	5.6	0	46	76.5
Lick Creek	1	27.76	3.5	36	26	71.0
Lick Creek	2	21.71	5.4	33	26	64.0
Lick Creek	3	6.92	5.9	42	34	65.5
Lt. Buck Creek	1	15.95	4.7	21	28	62.0
Lt. Buck Creek	2	33.38	7.1	34	30	64.5
Lt.Eagle Creek	1	15.36	4.6	42	36	73.5
Lt.Eagle Creek	2	31.12	4.9	41	34	63.0
Lt.Eagle Creek	3	66.69	6.2	49	44	61.0
Lt.Eagle Creek	1a	17.74	6.4	41	34	82.0
Pleasant Run Creek	1	20.46	6.5	56	22	52.0
Pleasant Run Creek	2	27.87	7.5	61	22	57.0
Pleasant Run Creek	3	40.94	5.7	68	32	55.5
Williams Creek	1	56.35	5.8	20	42	75.0
Williams Creek	2	43.85	5.7	23	50	82.0
Williams Creek	3	56.35	12.9	26	50	77.5

Table 23. Spearman's Rank Correlation between fish community variables (IBI and individual metrics) and habitat and land use variables.

	High Density Urban	QHEI	Riffle Depth	Average Depth	**Pool volume**
Index of Biotic Integrity	R = -0.073 P = 0.7583	R = 0.666 P = 0.0005	R = 0.706 P = 0.0001	R = 0.497 P = 0.0158	R = 0.404 P= 0.0561
# Species	R = -0.357	R = 0.445	R = 0.353	R = 0.335	R = 0.309

	P = 0.1214	P = 0.0331	P = 0.0980	P = .1178	P = 0.1512
# Darter Species	R = -0.584	R= 0.501	R = 0.347	R = 0.534	R = 0.241
	P = 0.0087	P= 0.0148	P = 0.1045	P = 0.0087	P = 0.2678
# Sensitive Species	R = -0.133	R = 0.409	R = 0.470	R = 0.427	R = 0.596
	P = 0.5761	P = 0.0527	P = 0.0237	P = 0.0422	P = 0.0027
% Tolerant Individuals	R = 0.147 P = 0.5349	R = -0.584 P = 0.0034	R = -0.701 P = 0.0001	R= -0.535 P = 0.0085	R = -0.457 P = 0.0285
% Insectivore Individuals	R = -0.175 P = 0.4599	R = 0.426 P = 0.0429	R = 0.460 P = 0.0272	R = 0.391 P = 0.0650	R = 0.432 P = 0.0396
% Pioneer Individuals	R = 0.185 P = 0.4327	R = -0.514 P = 0.0119	R = -0.505 P = 0.0139	R = -0.359 P = 0.0928	R = -0.088 P = 0.6881
% Simple Lithophils	R = 0.021 P = 0.9296	R = 0.433 P = 0.0388	R = 0.630 P = 0.0013	R = 0.332 P = 0.1220	R = 0.156 P = 0.4777
% Reach As Pool Habitat	R = -0.605 P = 0.0047	R = 0.039 P = 0.8596	R = -0.257 P = 0.2356	R = 0.336 P = 0.1169	R = 0.262 P = 0.2274
% Reach As Riffle Habitat	R = 0.647 P = 0.0021	R= 0.056 P = 0.798	R = 0.393 P = 0.0632	R = -0.178 P = 0.417	R = -0.081 P = 0.7145

11 Conclusions

This project has produced an array of tools for use in evaluating the response of Midwestern watersheds and streams to increasing urbanization. The dynamic hydrology, temperature, and oxygen models are most applicable for engineers, biologists, and water quality managers working with specific stream reaches and for characterizing sites where ecological data is also available. Specific urbanization indicators have been tested using channel geomorphology, freshwater mussels, aquatic insects, and fish that have good potential for widespread application in the region. We plan to continue working with these to further develop the causal connection between hydrologic and geomorphic impacts during storm events and the observed response of stream biota found in this project to be sensitive to urbanization. The results of this project lend credence to the idea that significant ecological changes occur in streams affected by quite low levels of imperviousness (5 – 10 %, as shown by the mussel, *Baetid* , and clinger indicators). Indian Creek in Tippecanoe County is presently at about 3% imperviousness, with rapidly expanding subdivisions within its watershed. Since we have a long record of hydrology, water quality, and biological data for this stream, it warrants further study in the coming years.

The L-THIA approach is ideal for planners and others looking at larger-scale situations. The extension of L-THIA for predicting NPS pollution as well as runoff volume is especially useful for region planning where various sub-watersheds may differ in their soil, terrain, and water characteristics and therefore in pollution potential.

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13 Project Publications

Bhaduri, B. 1998. A GIS-based model to assess the long-term impacts of land use change on hydrology and nonpoint source pollution. Ph.D. Thesis, Department of Earth and Atmospheric Sciences, Purdue University.

Bhaduri, B., J. Harbor, B. Engel, and M. Grove, M., 2000, Assessing watershed-scale, long-term hydrologic impacts of land use change using a GIS-NPS model. Environmental Management, 26(6):. 643–658.

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14 Project Presentations and Talks

Bhaduri, B., Harbor, J., Engel, B., Lim, K. and Jones, D., 1999, Assessing the Long-Term Hydrologic Impact of Land Use Change Using a GIS-NPS Model and the World Wide Web. Environmental Problem Solving with GIS: A National Conference, Cincinnati, Ohio.Proceedings Volume (USEPA).

Deitch, M., and A. Spacie. Effects of storm-induced disturbances on periphyton communities: an experiment in artificial streamside channels. Society of International Limnology World Conference, Melbourne, AU. Feb. 4-9, 2001.

Harbor, J., Long-term hydrological impact assessment for land use change. MACOG Workshop on managing land use change. Goshen, IN, September, 1999.

Harbor, J., Engel, B., Jones, D., Pandey, S., Lim, K. and Muthukrishnan, S. A Comparison of the Long-Term Hydrological Impacts of Urban Renewal versus Urban Sprawl. National Conference on Tools for Urban Water Resource Management & Protection, Environmental Protection Agency, Conference Proceedings.

Lim, K. J., Engel, B. A., Kim, Y., and Harbor, J. "Development of the Long Term Hydrologic Impact Assessment (L-THIA) WWW Systems." 10th International Soil Conservation, Purdue University, West Lafayette, Indiana. 1999.

Myers-Kinzie, M. and A. Spacie. The response of benthic insect communities to urbanization in small midwestern watersheds. North American Benthological Assocn. annual meeting, Keystone CO. May 28-June 1, 2000.

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Rich, C., A. Spacie, M. Doyle, and J. Harbor. Urbanization’s effects on streams: linkages between altered hydrology, geomorphology, habitat and fish assemblage structure. Midwest Fish and Wildlife Conference, Chicago, IL. Dec.8, 1999.

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15 **Supplemental Keywords**

watersheds, ecological effects, ecosystem, indicators, scaling, aquatic, habitat, integrated assessment, engineering, ecology, hydrology, modeling, Midwest, Indiana, IN, EPA Region V, integrated assessment, EPA Region V

16 Relevant Web Sites

The L-THIA model can be run directly at:
http://danpatch.ecn.purdue.edu/~sprawl/LTHIA7/

The Planning with POWER program to assist regional planners is described at:
http://www.planningwithpower.org/

Related research on geomorphology is given at:
http://www.eas.purdue.edu/geomorph/

17 Appendix

The following published papers and manuscripts are appended to this report in the form of PDF documents:

Bhaduri, B., J. Harbor, B. Engel, and M. Grove, M., 2000, Assessing watershed-scale, long-term hydrologic impacts of land use change using a GIS-NPS model. Environmental Management, 26(6). 643–658.

Doyle, M., J. Harbor, C. Rich and A. Spacie, 2000, Examining the effects of urbanization on streams using indicators of geomorphic stability. Physical Geography, 21:155-181.

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Myers-Kinzie, M., A. Spacie, C. Rich, and M. Doyle. Relationship of unionid mussel occurrence to channel stability in urban streams. Verh. International Vereinigung Limnologie (accepted).

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Wang, H., C. Xu, M. Hondzo, V. Poole, and A. Spacie. Stream dissolved oxygen dynamics in urbanized and agricultural watersheds. *Manuscript in preparation*.