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Evaluating The Illinois Stream Valley Segment Model As An Effective Management Tool

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EVALUATING THE ILLINOIS STREAM VALLEY SEGMENT
MODEL AS AN EFFECTIVE MANAGEMENT TOOL

WARRNER

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EVALUATING THE ILLINOIS STREAM VALLEY
SEGMENT MODEL AS AN EFFECTIVE MANAGEMENT
TOOL

BY

STEPHEN S. WARRNER

THESIS

SUBMITTED IN PARTIAL FULLFILLMENT OF THE REQUIREMENTS FOR THE
DEGREE OF

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CHARLESTON, ILLINOIS

2008

I HEREBY RECOMMEND THAT THIS THESIS BE ACCEPTED AS FULFILLING
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Abstract

Stream habitat assessment procedures have been used to monitor biological potential, habitat restoration, and determine anthropogenic impacts on streams. In the past, the Illinois Department of Natural Resources (IDNR) and the Illinois Natural History Survey (INHS) have determined which stream reaches to sample habitat quality based on historical data, accessibility, and whether or not a reach is representative of an area. To develop a consistent and unbiased procedure for choosing sampling locations, the IDNR and INHS have proposed a technique by which watersheds are divided into homogeneous stream segments called valley segments. Valley segments within each watershed are determined by GIS parameters including surficial geology, predicted flow, slope, and drainage area. To date no research has been conducted to 1) detail the methodology used to test the valley segment model, 2) determine if arcs within a MAVS have similar habitat variables, 3) determine if the stream habitat within a valley segment is homogeneous, and 4) determine if different valley segments have varying habitat variables.

To test the valley segment concept, two adjacent valley segments were randomly selected within thirteen streams in the Embarras River watershed, located in east-central Illinois. Reaches were randomly selected within each valley segment, and a modified USEPA Wadeable Streams Assessment was used to quantify channel, bank, riparian, and floodplain structure. Habitat variables were combined for each stream through a principle components analysis (PCA) in order to measure environmental variation, and two multivariate analysis of variance (MANOVA) tests were performed on PCA axes 1-3 to determine if differences existed within and between adjacent valley segments. Within

each valley segment, reaches were significantly different from each other indicating a relatively heterogeneous physical environment ($3.52 \leq F \leq 19.69$; $p \leq 0.0003$). In the majority of streams, adjacent valley segments were significantly different from each other ($5.37 \leq F \leq 245.13$; $p \leq 0.002$). However, habitat variability within each valley segment was less than variability between adjacent valley segments. Therefore, when utilized at an appropriate spatial scale, this model is an effective management tool for identifying representative sampling locations and extrapolating reach-specific information.

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Introduction

Chemical stressors, exotic species introductions, and habitat degradation continue to impact the quality of North American streams (Karr and Chu 2000; US EPA 2002). While these stressors often vary depending on the dominant land use of the region, in both agricultural and urban landscapes, stream modifications such as channel reconfiguration, dam construction, and riparian zone removal are common sources of habitat degradation and fragmentation (Rankin 1989; Zwick 1992; Rogers et al. 2002). In regions with high agricultural land use, siltation, nitrogen loading, and pesticides are typically the main sources of stream degradation, while in urbanized regions, toxic metals, phosphorus loading, and pathogenic contamination are more prevalent (Crunkilton et al. 1996; Allan et al. 1997; Johnson et al. 1997; Paul and Meyer 2001; Fitzpatrick et al. 2005; Diana et al. 2006; Allan and Castillo 2007).

In Illinois, approximately 80% of the land use is dedicated towards agriculture practices. In order to reduce flooding and increase the amount of arable land, landscape modifications such as tilling of fields, stream channelization, and riparian zone removal are often employed but have deleterious effects on the stream (Rhoads and Herricks 1996; Talmage et al. 2002; Infante et al. 2006). Channel alterations in conjunction with an often lack of best management practices (BMPs) such as contour plowing, reduced tillage, riparian buffer strips, and grassed waterways, have altered the chemical and physical properties of the water while degrading the quality of habitat available for aquatic communities (Rabeni and Jacobson 1999; Meader and Goldstein 2003; Royer et al. 2006; Allan and Castillo 2007).

In order to drain agricultural land effectively, woody debris obstructions are removed and stream channels are widened, deepened, and straightened (Lau et al. 2006; Allan and Castillo 2007). Consequently, the sinuosity (degree to which the channel meanders) of the stream decreases, reducing stream complexity and creating a homogeneous environment for aquatic organisms (Rankin 1989; Harding et al. 1998; USEPA 2004; Oscoz et al. 2005). Along with instream channel modifications, riparian zone removal alters allochthonous inputs into the stream, decreasing potential food sources and habitat availability while increasing sediment runoff (Sheehan and Rasmussen 1999). Sediment carried from agricultural fields into streams settles into the crevices of larger substrate, decreasing substrate heterogeneity and reducing habitat available for macroinvertebrate and fish species (Berkman and Rabeni 1987; Shields et al. 1995; Sheehan and Rasmussen 1999; Wang et al. 2002). In addition, high concentrations of suspended solids within the water column decrease light transmittance and subsequently impact primary production and trophic dynamics (Wetzel 2001; Talmage et al. 2002).

Furthermore, removal of the riparian zone can also increase nutrient and sediment loading, reduce stream shade, and produce altered temperature regimes which disrupts the natural chemical of streams and enriches the water to the point of eutrophication (Harper 1992; Larson and Larson 1996; Mitner and Rankin 1998; Bartholow 2000). In conjunction with increased water temperature, nutrient loading leads to uncontrolled algal production, decreased dissolved oxygen, and increased stress to aquatic communities (Wetzel 2001; Figueroa-Nieves et al. 2006; Morgan et al. 2006; Royer et al. 2006). Thus, temperature and nutrient loading can dramatically change the chemical and

physical properties of a stream (Winger 1981; Johnson et al 1997; Rabeni and Jacobson 1999).

The deteriorating water and habitat quality associated with agricultural activity can have profound impacts on aquatic communities, contributing to changes in community composition including shifts in species diversity, abundance, and even the disappearance of sensitive aquatic species (Karr et al. 1985; Crunkilton et al. 1996; Carpenter et al. 1998; Sheehan and Rasmussen 1999; Meader and Goldstein 2003). Stream segments affected by stream degradation are often characterized by communities dominated by tolerant and/or exotic species, a more simplified food web, and higher incidences of diseases such as deformities, lesions, and tumors (Karr 1981; Ney 1999).

With the establishment of the Clean Water Act in 1972, protocols were developed to detect deteriorating stream conditions as well as restore and maintain the physical integrity of these waterways (USEPA 1990). Water chemical analyses, bioassessments, and habitat assessments have been designed and implemented throughout the United States in order to identify these possible stressors, while playing a critical role in allowing the evaluation of the overall quality of aquatic communities and habitat (Winger 1981; Karr 1991; Talmage et al. 2002; Wang et al. 2006).

Chemical analyses provide empirical and legal validity to the assessment of stream condition while utilizing a wide range of analyses to detect stressors originating from industrial, residential, and agricultural point and non-point sources (Karr 2000; USEPA 2002; Conrad and Warner 2005). Often these analyses are used to monitor long-term trends in water composition and quality (Lawrence et al. 2004; Adams and Kochenderfer 2007). However, these analyses are often costly and time intensive,

difficult to correlate to impacts on aquatic communities, and provide only a “snap shot” of water quality for a given reach at a given point in time (Karr 1991; Karr 2000; Conrad and Warner 2005).

Stream bioassessment procedures supplement chemical sampling to provide the most accurate means of detecting and measuring overall stream water quality (Plafkin et al. 1989; Sheehan and Rasmussen 1999; OEPA 1994). Bioassessments can also be used to determine the cumulative impacts of pollution and habitat degradation on aquatic communities, determine water quality trends over time, and help to narrow down sources of local water quality degradation (Karr et al. 1983; Karr 1991). One such assessment, the Index of Biotic Integrity (IBI), was originally developed by Karr et al. (1983) and has since seen several revisions (Simon and Dufour 1997; IDNR 2000). This index provides a sensitive and reproducible measure of the integrity of aquatic communities using biological criteria to assess the health of the aquatic system (Karr and Dudley 1981; Karr 1983; Karr 1991). Total IBI scores are used to assign a narrative description to the biological integrity of an aquatic community within the sampled stream reach (Karr and Dudley 1981; Karr 1983; Simon and Dufour 1997). This type of assessment has been applied to several aquatic communities including fish, macroinvertebrates, and algae (Simon and Dufour 1997; Weigel et al. 2002; Rasmussen 2007).

Stream habitat assessments are used to guide stream enhancement and restoration projects with overall goals to improve water quality and the biological integrity of river systems while creating the ability to monitor these projects for changes brought about by habitat improvements (Maddock 1999; Montgomery and MacDonald 2002). These assessments also provide a method to identify limiting factors that influence the

biological potential, or health of a stream reach (Allan 1997; Wang et al. 1998; Maddock 1999). The Qualitative Habitat Evaluation Index (QHEI) is one assessment procedure currently utilized in Illinois to determine habitat quality by evaluating channel, instream cover, riparian zone, and floodplain structure (Rankin 1989). This index is designed to be a relatively rapid assessment used to determine habitat quality, identify limiting habitat factors, and monitor stream improvements (Hannaforde et al. 1997). Habitat variables estimated by the QHEI generally correspond to physical factors that directly influence aquatic community parameters such as species presence, absence, and composition of communities within a stream segment (Rankin 1989).

In addition to influencing aquatic community composition and structure, stream habitat quality limits the total "living space" available for these communities (Maddock 1999). Therefore, habitat indices can determine and evaluate the biological potential of a stream (Barbour et al. 1999). Since aquatic communities rarely exceed the quality of the habitat, comparing habitat assessments and bioassessments often reveal underlying causes of impairment to biological communities (Plafkin et al. 1989; Simon 1999; Conrad and Warner 2005). For example, reaches with severely altered habitat would be expected to have a relatively poor quality community and vice versa. Alternatively, high quality habitat with poor biological communities may be an indication of chemical pollution (Plafkin et al. 1989; Simon 1999). The ability to combine information on the chemical, biological, and physical integrity of a sample reach allows managers to accurately quantify stream quality and effectively manage water resources.

Given the total number of streams in Illinois, managers lack the time and the resources necessary to visit each stream and determine its ecological status (Seelbach et

al. 2005; Wang et al. 2006). To date, the Illinois Department of Natural Resources (IDNR) has determined which stream reaches to sample for overall stream quality based on historical data, reach accessibility, proximity to point source pollution, relative location to tributaries, relative position within the watershed, and whether or not the reach is representative of the entire watershed (IDNR 2002). However, determining if these sample reaches are representative of conditions throughout the stream is often difficult, due to the inherent spatial variability of streams and the subjective criteria used to select potential sample reaches (Seelbach and Wiley 1997; Seelbach et al. 1997; Palmer and Poff 1997; Montgomery and MacDonald 2002; Kilgour and Stanfield 2006).

In addition to selecting a representative sampling location, it is also difficult to determine the appropriate spatial scale for management objectives (Lammert and Allan 1999; Meader and Goldstein 2003; Wang et al. 2006; Brenden et al. in press). In the past, stream managers have conducted field assessments at reach lengths between 100 and 1,000 meters (Wang et al. 2006). However, these reaches may not be of sufficient length to extrapolate this information to larger river segments. For example, it may not be appropriate to estimate instream cover at one reach and then extrapolate this information to a larger spatial scale such as a sub-watershed level. When the appropriate spatial scale is not utilized, problems can arise with extrapolating data to larger spatial scales, leading to the misrepresentation of overall stream quality within that larger stream segment (Bryce and Clarke 1996). Since the majority of sampling efforts take place at relatively small spatial scales, stream spatial units representative of the entire stream reach should be developed to allow managers to effectively select, sample, and extrapolate information from representative sampling locations (Frissell et al. 1986; Bryce and Clarke 1996;

Seelbach et al. 1997). Once these representative sampling locations are identified, then it is possible to extrapolate reach-specific information to larger spatial scales (Bryce and Clarke 1996).

To develop a more consistent and unbiased procedure for choosing representative reaches to sample for stream quality, the IDNR and Illinois Natural History Survey (INHS) have proposed a technique by which watersheds are divided into relatively homogeneous stream segments, called valley segments. Valley segments within a watershed are determined by ArcGIS 9.2 Geographic Information System (GIS) software (ESRI, Redlands, California) parameters such as surficial geology, predicted flow class, predicted thermal class, slope, drainage area, and link number (Seelbach et al. 1997; ESRI 2006; Brenden et al., in press). Surficial geology defines the valley character of a stream, encompassing characteristics such as channel confinement and incision (Seelbach et al. 1997). The predicted flow class model was developed based on the daily flow data from gauging stations throughout the state of Illinois (Seelbach et al. 1997). The predicted thermal class model considers summer temperature means and diurnal fluctuations (Seelbach et al. 1997). Slope is the gradient (ratio of drop in a stream per unit distance) of the stream (USEPA 2004), while drainage area includes the total land upstream from that point in the stream where water flows back into the stream (Allan and Castillo 2007). Lastly, the model considers link number, another size surrogate that measures the sum of all first order streams upstream from a specific point within a stream (Seelbach et al. 1997).

The valley segment model assumes that 1) stream quality within the entire valley segment is fairly homogeneous, 2) stream segments with similar GIS parameter criteria

will be grouped into one valley segment, creating larger valley segments, 3) valley segment boundaries occur at a stream confluence, and 4) overall stream quality within each valley segment is more similar than the stream quality between adjacent valley segments (Seelbach et al. 1997; Seelbach and Wiley 1997). With these assumptions the model utilizes a clustering algorithm called the valley segment affinity search technique (VAST), which merges and separates valley segments at pre-specified thresholds for each of these GIS parameters (Brenden et al. in press). Adjacent stream arcs with similar GIS parameter values are combined into a multiple arc valley segment (MAVS), while stream arcs with differing parameter values are separated into two single arc valley segments (SAVS).

Utilizing the valley segment model, managers can select representative sampling locations. For example, the length of each valley segment is similar to the spatial scale at which stream managers sample stream habitat and biological communities and should account for the natural longitudinal variability within each stream (Vannote et al. 1980; Seelbach et al. 1997; Wang et al. 2006; Brenden et al. 2007). Valley segments are also at a similar scale to which physical processes and aquatic organisms function (Frissell et al. 1986; Seelbach et al. 1997). Segments contain multiple habitats required for stream fish during their life cycles, which may allow investigation into population dynamics of aquatic communities, since smaller spatial scales (i.e. reach, channel unit) are often inadequate to encompass population dynamics (Schlosser 1991; Frissell et al. 1997; Seelbach 1997; Seelbach et al. 1997; Lammert and Allan 1999). While the valley segment model encompasses temporal and spatial scales, it integrates both structure and function into a relatively low cost procedure that creates consistent understanding among

stream managers (Seelbach et al. 1997). Given these benefits, valley segments may be an appropriate system for evaluation, management, and extrapolation of stream condition.

To date no research has been conducted to test the major assumptions of the valley segment model, those being that habitat within valley segments are homogeneous and whether different valley segments have varying stream habitat quality variables. Therefore, another set of parameters, stream habitat variables, were selected in order to test the effectiveness of the valley segment model. Habitat measurements were collected for channel, instream cover, bank, riparian, and floodplain structure as previous stream classification systems and research have indicated the importance of variables within these habitat categories (Platts 1974; Gorman and Karr 1978; Schlosser 1982; Angermeier and Karr 1984; Bohn 1986; Frissell et al. 1986; Rankin 1989; Gregory et al. 1991; Seelbach et al. 1997; Hayano et al. 2002; Montgomery and MacDonald 2002; Talmage et al. 2002; Meader and Goldstein 2003).

Thus, the purpose of this research project was to assess stream valley segments as determinants of stream habitat quality with the intention to help the IDNR, INHS, and Illinois Environmental Protection Agency (IEPA) determine if valley segments are a valid way for selecting stream locations for investigating habitat quality. Information synthesized from this project will be used to develop the first protocols for sampling valley segments and create a habitat quality data set that can then be used by professionals to make informed management decisions related to overall stream quality. Thus, the objectives of this project were to 1) detail the methodology used to test the valley segment model, 2) determine if arcs within a MAVS have similar habitat variables,

- 3) determine if the stream habitat within a valley segment is homogeneous, and 4) determine if different valley segments have varying habitat variables.

If the assumptions of the model hold true, stream valley segments determined by grouping similar GIS parameter values will be at a fine enough spatial scale to determine habitat differences between reaches, thus we expect valley segments to be true habitat divisions within a stream. Therefore, reaches within each valley segments should have relatively similar habitat compared to habitat within other valley segments. This classification system can then be used to determine representative sample locations and allow for accurate and practical state-wide stream restoration planning while still retaining natural ecological variation among streams (Seelbach and Wiley 1997; Seelbach et al. 1997; Hawkins et al. 2000; Seelbach et al. 2005; Seelbach et al. 2006).

Methods

Stream Selection

All streams were located in the upper Embarras River watershed in east-central Illinois (Figure 1). The watershed has a drainage area of approximately 6,212 km². Approximately 75% of streams within the watershed are disturbed, with 73% of this disturbance coming from agriculture practices and 2% from urbanization while only 12% of streams in the watershed have been identified as least impacted by anthropogenic disturbance (IDNR 1996). To test the valley segment model, sampled streams were dispersed throughout the upper portion of the watershed in order to 1) ensure adequate coverage of the upper portion of the watershed and 2) test the model utilizing a variety of stream types.

Valley Segment Selection

All streams within the Embarras River watershed had previously been separated into different valley segments types by VAST using GIS parameters including surficial geology, predicted flow class, predicted thermal class, slope, drainage area, and link number. When considering which valley segment types to sample, segment accessibility (i.e. number of bridges crossing a specific valley segment) was considered first. Secondly, valley segments less than 1,600 meters in length were not selected as smaller valley segments would not allow proper spacing of replicate reaches, possibly resulting in reduced variability due to the close proximity of reaches.

Thirteen streams were identified and sampled within the Embarras watershed between May and August of 2007. Each stream consisted of two adjacent valley segment types for a total of twenty-six valley segments. There were two different types of valley

Figure 1. The geographic location of the Embarras River watershed is designated within the map of Illinois. Thirteen streams, indicated by the circles on the inset map, were selected within the upper portion of the watershed in order to test the valley segment model. The streams selected within this watershed included Brushy Fork (BF), Cottonwood Creek (CC), Hurricane Creek (HC), Little Embarras River (LER), Lost Creek (LC), Lower Kickapoo Creek (LKC), McNary Branch (MB), Polecat Creek (PC), Ranger Creek (RaC), Riley Creek (RiC), Scattering Fork (SF), Upper Kickapoo Creek (UKC), and Whetstone Creek (WC).

segments sampled: multiple arc valley segments and single arc valley segments, with a stream arc defined as a stream segment extending from one confluence to the next downstream confluence (Seelbach et al. 1997; Figure 2).

Seven stream locations consisted of one upstream MAVS and an adjacent downstream SAVS (Table 1). MAVS were selected because these valley segments are true tests of the stream arc merging routine that managers have used to delineate valley segments. The remaining six stream locations were composed of two adjacent SAVS (Table 1). Adjacent SAVS with similar drainage areas were selected to determine if these valley segments should have been combined into a MAVS. The assumption is that these similar sized stream segments should have similar habitat structure and quality (Vannote et al. 1980).

Reach Selection

Using Arc GIS, a valley segment from each stream was selected and separated into representative stream arcs. Each arc was divided into 400 meter increments (Figure 3). This distance was selected in order to 1) ensure that reaches sampled were not in close proximity to each other and 2) to ensure adequate coverage within each valley segment arc. Three reaches were randomly selected and sampled within each valley segment arc to minimize bias. In the field, reaches were not sampled within 200 meters of a bridge, tributary, or beaver dam as research has documented that these factors may influence habitat quality and aquatic community structure (Minshall et al. 1985; Barbour et al. 1999; Montgomery and MacDonald 2002).

Adjacent valley segments downstream of each initial valley segment were identified. Within these valley segments, reaches were randomly selected and sampled

Figure 2. Utilizing Arc GIS, streams within the Embarras River watershed were divided into spatial units called stream arcs. Adjacent stream arcs with similar values were grouped into multiple arc valley segments (MAVS), while arcs with dissimilar values were separated into different valley segment types. Lastly, stream arcs that were not similar to any other adjacent arc were classified as single arc valley segments (SAVS).

Table 1. Summary characteristics for adjacent valley segments sampled within the thirteen streams of the upper Embarras River watershed. Valley segment location was designated as either above (initially selected valley segment) or below (adjacent downstream valley segment).

Stream	Location (Above/Below)	Segment Type	Total Length (m)	Drainage Area (km ²)
BF	Above	SAVS	10,474.37	186.33
	Below	SAVS	3,709.55	203.93
CC	Above	SAVS	3,426.52	67.82
	Below	SAVS	5,952.15	71.97
HC	Above	MAVS	14,908.44	87.86-98.90
	Below	SAVS	12,598.74	108.57
LER	Above	MAVS	31,191.27	183.04-277.47
	Below	SAVS	10,569.66	288.84
LC	Above	MAVS	18,027.82	27.90-37.45
	Below	SAVS	8,889.61	43.98
LKC	Above	SAVS	7,435.76	69.50
	Below	SAVS	3,980.30	77.44
MB	Above	SAVS	6,411.40	119.02
	Below	SAVS	3,978.78	157.77
PC	Above	SAVS	3,687.43	73.55
	Below	SAVS	6,874.27	75.16
RaC	Above	SAVS	9,618.04	99.06
	Below	SAVS	7,508.16	109.06
RiC	Above	MAVS	28,722.69	6.77-103.78
	Below	SAVS	6,965.19	166.02
SF	Above	MAVS	31,397.12	9.88-27.50
	Below	SAVS	11,768.84	35.86
UKC	Above	MAVS	18,364.34	17.93-25.86
	Below	SAVS	5,621.39	35.52
WC	Above	MAVS	18,776.94	1.90-21.98
	Below	SAVS	9,629.61	29.21

Figure 3. Reach selection procedure implemented to identify reaches within valley segments. Every stream arc within a valley segment was divided into 400 meter increments, with each increment being a potential sampling location. Three reaches per stream arc were then randomly selected for sampling.

within each stream arc based on the same protocols used to identify reaches in the upstream valley segments. All downstream valley segments were SAVS. Within each stream, all reaches were sampled within one week in order to minimize temporal variability of habitat variables between reaches.

Stream Habitat Sampling

A modified USEPA Wadeable Streams Assessment (WSA) transect method was used to quantify habitat characteristics at a reach scale (USEPA 2004). This assessment utilizes a random, systematic sampling methodology to minimize bias in the placement and positioning of habitat measurements (USEPA 2004). A quantitative sampling procedure was selected in order to obtain direct measurements of stream habitat variables. Qualitative sampling was not implemented because of its subjectivity which could result in potential loss of resolution when performing statistical analyses and may hinder the ability to detect statistically significant differences between habitat variables, sample sites, and adjacent valley segments (Roper and Scarnecchia 1995; Wang et al. 1996; Hannaford et al. 1997).

The WSA delineates reach length based on wetted width, and systematically spaces each transect at one-tenth the total reach length (USEPA 2004). For this project, reach length was reduced to a set distance of 100 meters to provide more consistent comparisons among all reaches within a valley segment. While transects were still spaced at the interval outlined in the WSA, the number of total transects was reduced to ten due to the reduction in reach length. In addition, stream bank structure and riparian vegetation variables are typically measured five meters upstream and downstream of each

transect. However, our protocol was modified from five to three meters due to the previously mentioned modifications to reach length.

A Garmin ® Global Positioning System (GPS) was used to locate each sample reach within the valley segment. All streams were sampled at baseflow conditions. Each reach began at the top of the nearest channel unit and progressed 100 meters upstream. Every ten meters, transect points were systematically marked using flagging tape over the entire reach length, and at each transect, channel, bank, riparian, floodplain, and instream cover characteristics were collected.

Channel characteristics measured included wetted width, thalweg (flow path of the deepest water in the stream channel), water depth, and substrate. Wetted width, thalweg, and water depth were measured to the nearest centimeter. Organic and inorganic substrate was collected by a modified Wolman pebble count and assigned a categorical value (Wolman 1954). In order to perform statistical analyses on substrate, categorical scores were then assigned numeric values based on the size of substrate (Table 2).

Bank characteristics measured included bank angle, bank undercut, bank erosion, and dominant vegetation type. All four variables were determined for both left and right banks. Bank angle was measured with a clinometer to the nearest degree. Bank undercut distance was measured to the nearest 0.01 meter. Bank erosion was categorized into one of four categories (Table 2). The abundance of each stream bank vegetation type (herbaceous, woody-shrubs, trees, bare, and bedrock bank) was determined and a ranking system was employed (Table 2). For statistical purposes, bank angle, erosion, and dominant vegetation type for both left and right banks were averaged together for each

Table 2. Rank category values associated with habitat variables collected during quantitative habitat assessment of each sample reach.

Rank	Substrate	Bank Erosion	Bank Vegetation	Floodplain Quality	Instream Cover	Dominant Riparian Type
0			0-25%	No disturbance	0%	
1	Clay	0-10%	26-50%	Old field	1-10%	Herbaceous
2	Silt	11-25%	51-75%	Cow pasture	11-40%	Woody-shrub
3	Sand	26-50%	>75%	Agriculture	41-75%	Tree
4	Detritus	>50%			>75%	Mix of types
5	Fine gravel					
6	Gravel					
7	Cobble					
8	Boulder					
9	Rip rap					
10	Bedrock					

transect.

Riparian and floodplain characteristics measured included width of riparian zone, dominant riparian vegetation type, percent canopy cover, and immediate floodplain quality. All variables were determined for both left and right sides of the stream. Riparian zone width was estimated to the nearest ten meters, and the maximum riparian zone value that could be recorded was 100 meters. Total riparian zone width was determined by adding together both banks at each transect. Canopy cover was measured at the midpoint of each major transect with a convex spherical densiometer model C (Lemmon 1956). Dominant riparian type was categorized into one of four categories that best represented the riparian zone at the transect point: herbaceous, woody shrubs, trees, or mixed (combination of herbaceous, woody shrubs, and trees). Both riparian vegetation scores were then combined for a total score. Landuse within 30 meters of the riparian zone was recorded in order to determine floodplain quality, and a ranking system was then used based on increasing levels of floodplain disturbance (Table 2).

Instream cover characteristics measured included aquatic macrophytes, root mats, undercut banks, artificial substrate, submerged terrestrial vegetation, rootwads, overhanging vegetation, boulders, small woody debris, and large woody debris. Instream cover was visually estimated and ranked (Table 2). To be counted, overhanging vegetation had to be within one meter of the surface of the water. Small woody debris was counted if it had a minimum diameter of 5 centimeters and a minimum length of 0.75 meters. Large woody debris was counted if it was within the wetted width of the channel or at least partially within the bankfull stage of the channel. Large woody debris had to

have a minimum diameter of 10 centimeters and a minimum length of 1.5 meters (USEPA 2004).

Sinuosity and bankfull width were measured at each reach scale. Sinuosity was measured by using a rangefinder to determine the straight line distance and a surveyor's rope to determine the thalweg distance of each reach. Thalweg distance was then divided by the straight line distance to determine sinuosity. Bankfull width was defined as the channel that is filled every one to two years by a moderate sized flood event (USEPA 2004; USDA 2005). Bankfull width was recorded at three major transects within the sample reach based on the procedures outlined by the USDA (2005). Perturbations such as removal of natural vegetation, channel modification, bank erosion, and presence of tile drains were noted for each reach.

Statistical Analyses

Using statistical analysis software (SAS), a principle components analysis (PCA) of all stream habitat variables was used to determine if similarities existed within and between stream arcs. If two habitat variables had a correlation greater than 0.60, then one of these variables was removed from the PCA. The number of habitat variables collected at each reach was greater than the number of reaches sampled within each valley segment. A large sample size improves the consistency of PCA axes, and it has been suggested that there should be approximately five sample units for each observed variable (Tabachnik and Fidell 1989; McCune and Grace 2002). Therefore, transects within each reach were used as replicates instead of the reaches in order to increase the sample size for each PCA. By increasing the sample size, there was a more appropriate ratio between observed variables and sample size. Analyzing habitat data using transects was also

beneficial in that reach variability was able to be compared within valley segments while arc variability was able to be compared between valley segments.

Multiple arc valley segments were first analyzed to determine the effectiveness of the stream arc merging routine at combining similar stream arcs. A PCA was performed for each MAVS for a total of seven PCAs. Each principle component axis was selected only if it had an eigenvalue greater than one (McCune and Grace 2002). Based on the amount of variation explained by each PCA axis, a set number of axes were selected as the measure of environmental variability within and between valley segment arcs. In other words, it was not the focus of this project to determine how a single variable (e.g. wetted width) varied within and between stream arcs. Instead, it was more important to focus on how a suite of habitat variables described the environmental variation within and between stream arcs. Since each principle component axis is an aggregate of all environmental variables, each axis was analyzed using a multivariate analysis of variance (MANOVA).

Two MANOVAs were performed since the objectives of the project were to investigate the valley segment model at two different spatial scales. The first spatial scale focused on the scale of the stream reach (i.e. were there differences between reaches within a stream arc). The second spatial scale focused on the scale of the stream arc (i.e. were there differences between arcs within a MAVS). A Pillai's trace was selected to determine the p-value as it is more robust against violations of multivariate normality, and significant levels were accepted at a p-value less than 0.05 (Gotelli and Ellison 2004).

Comparisons between adjacent valley segments within all thirteen streams were then conducted to determine if the valley segment model was properly separating stream arcs into different valley segment types. For streams containing MAVS, all arcs within each MAVS were combined and all habitat data within these valley segments were compared to the downstream SAVS. Streams containing adjacent SAVS were also compared. A PCA was conducted for each stream to determine if similarities exist within valley segments and differences exist between adjacent valley segments. Based on the amount of variation explained by each PCA axis, a set number of axes were selected as a measure of environmental variability. Two MANOVAs were then used on these principle component axes to determine if significant differences occurred between reaches within each valley segment and between adjacent valley segments. A Pillai's trace was selected to calculate statistical significance, and significant levels were accepted at a p-value less than 0.05.

Results

Sample Stream Variability

Even with *a priori* stream selection criteria, a diversity of streams was selected within the upper Embarras River watershed to test the valley segment model (Figure 4). Channel, bank, riparian, and floodplain characteristics varied across all streams (Table 3). These characteristics also varied between reaches within each valley segment. For example, all reaches within Brushy Fork and Scattering Fork were modified such that the channel was straightened, stream bank vegetation was predominantly herbaceous, and the riparian zone was removed. The remaining streams had relatively few reaches with evidence of channel, bank, and riparian modifications. Streams like Polecat and Whetstone Creek had a relatively undisturbed, sinuous channel, stream banks with a mixture of vegetation types, and riparian zones that were intact but varying in width.

Arcs within each valley segment also varied based by valley segment model parameters and landscape parameters not included within the valley segment model (Tables 4 and 5, respectively). For example, model variables such as drainage area and slope ranged from 1.90 to 288.84 km² and 0.0076 to 1.6379 m/m, respectively. Agriculture was the dominant land use surrounding these streams (59.59% to 95.92%), while a relatively small portion of remaining land was forested (0% to 23.95%).

Excluded Stream Habitat Variables

Several habitat variables were not included in the statistical analyses. Instream cover variables were not included because these variables were unique to the meso-habitat spatial scale; thus, these variables may not be comparable at larger spatial scales. In addition, reach scale characteristics such as bankfull width and sinuosity were not

Figure 4. Graph depicting stream variability along principle component axes 1-3. All transect habitat data was summarized for each stream. On PCA axis one, the strongest loading positive habitat variables included substrate (0.3401), percent bare bank (0.3356), dominant riparian type (0.3333), canopy cover (0.3205), percent tree bank (0.2953), and percent bank erosion (0.2680), while strongest negative variables included percent herbaceous bank (-0.3413) and water depth (-0.2758). On PCA axis two, wetted width (0.4683), thalweg (0.4069), undercut bank (0.3376), percent bedrock bank (0.2907), and water depth (0.2772) were the strongest loading positive variables while floodplain quality (-0.3433) and bank angle (-0.3251) were the strongest negative variables. On the third PCA axis, bank erosion (0.3652), percent tree bank (0.3034), undercut bank (0.2534), and bank angle (0.2500) were the strongest loading positive variables while percent bedrock bank (-0.5564) and width of riparian zone (-0.3716) were the strongest negative variables.

Table 3. Mean values for channel, bank, riparian, and floodplain structure for each stream
($\pm$ one standard error is denoted in parentheses).

Stream	Wetted Width	Thalweg	Water Depth	Substrate	Bank Angle	Undercut Bank	%Woody-Shrub Bank	%Tree Bank
BF	10.30 (± 0.229)	0.58 (± 0.034)	0.43 (± 0.027)	3.33 (± 0.226)	48.15 (± 1.106)	0.00 (± 0.000)	0.30 (± 0.065)	0.04 (± 0.025)
CC	5.74 (± 0.250)	0.39 (± 0.030)	0.26 (± 0.021)	4.95 (± 0.068)	40.32 (± 1.345)	0.02 (± 0.011)	0.27 (± 0.047)	0.76 (± 0.092)
HC	7.73 (± 0.315)	0.44 (± 0.029)	0.24 (± 0.015)	5.75 (± 0.152)	34.18 (± 1.083)	0.02 (± 0.007)	1.07 (± 0.081)	0.36 (± 0.051)
LER	8.03 (± 0.252)	0.49 (± 0.021)	0.33 (± 0.016)	5.15 (± 0.040)	40.61 (± 0.966)	0.03 (± 0.008)	0.36 (± 0.039)	0.73 (± 0.062)
LC	3.25 (± 0.162)	0.31 (± 0.016)	0.19 (± 0.011)	4.96 (± 0.059)	41.07 (± 1.094)	0.00 (± 0.001)	0.60 (± 0.062)	0.63 (± 0.063)
LKC	6.07 (± 0.252)	0.39 (± 0.021)	0.23 (± 0.013)	5.25 (± 0.050)	38.74 (± 1.104)	0.04 (± 0.011)	0.56 (± 0.074)	0.63 (± 0.084)
MB	7.47 (± 0.301)	0.38 (± 0.027)	0.23 (± 0.018)	5.19 (± 0.054)	43.44 (± 1.277)	0.04 (± 0.014)	0.54 (± 0.071)	0.34 (± 0.066)
PC	6.10 (± 0.305)	0.32 (± 0.022)	0.19 (± 0.014)	5.81 (± 0.155)	39.04 (± 1.609)	0.01 (± 0.005)	0.74 (± 0.080)	0.50 (± 0.078)
RaC	5.01 (± 0.337)	0.35 (± 0.031)	0.23 (± 0.021)	4.94 (± 0.041)	37.03 (± 1.270)	0.00 (± 0.003)	0.43 (± 0.067)	0.58 (± 0.082)

Table 3 Continued.

Stream	Wetted Width	Thalweg	Water Depth	Substrate	Bank Angle	Undercut Bank	%Woody-Shrub Bank	%Tree Bank
RiC	6.35 (± 0.395)	0.39 (± 0.026)	0.26 (± 0.018)	5.00 (± 0.091)	50.81 (± 1.300)	0.01 (± 0.004)	0.88 (± 0.073)	0.49 (± 0.063)
SF	3.62 (± 0.131)	0.37 (± 0.008)	0.28 (± 0.008)	3.26 (± 0.088)	46.51 (± 1.006)	0.02 (± 0.004)	0.06 (± 0.021)	0.00 (± 0.000)
UKC	6.43 (± 0.184)	0.46 (± 0.021)	0.29 (± 0.015)	5.07 (± 0.090)	51.63 (± 0.988)	0.01 (± 0.007)	1.12 (± 0.087)	0.86 (± 0.061)
WC	3.82 (± 0.253)	0.18 (± 0.014)	0.10 (± 0.008)	5.17 (± 0.078)	51.32 (± 1.010)	0.01 (± 0.003)	1.03 (± 0.059)	0.72 (± 0.061)

Table 3 Continued.

Stream	%Bedrock Bank	%Herbaceous Bank	%Bare Bank	Bank Erosion	Canopy Cover	Floodplain Quality	Riparian Zone Width	Dominant Riparian Veg.
BF	0.00 (± 0.000)	2.87 (± 0.037)	0.42 (± 0.069)	0.33 (± 0.056)	0.76 (± 0.562)	3.00 (± 0.000)	7.35 (± 1.424)	3.00 (± 0.184)
CC	0.00 (± 0.000)	1.97 (± 0.090)	1.42 (± 0.091)	0.86 (± 0.085)	77.06 (± 3.495)	3.00 (± 0.000)	35.68 (± 2.259)	6.40 (± 0.205)
HC	0.09 (± 0.033)	1.88 (± 0.065)	1.29 (± 0.081)	0.72 (± 0.060)	47.61 (± 3.711)	3.00 (± 0.000)	59.37 (± 3.260)	7.07 (± 0.147)
LER	0.00 (± 0.000)	2.33 (± 0.040)	0.94 (± 0.047)	0.64 (± 0.044)	75.13 (± 2.031)	3.00 (± 0.000)	81.63 (± 4.194)	6.79 (± 0.123)
LC	0.00 (± 0.000)	2.09 (± 0.062)	1.50 (± 0.072)	0.59 (± 0.057)	85.89 (± 1.896)	2.47 (± 0.094)	94.64 (± 4.760)	7.48 (± 0.100)
LKC	0.00 (± 0.000)	2.16 (± 0.082)	1.17 (± 0.094)	0.80 (± 0.083)	45.53 (± 4.816)	3.00 (± 0.000)	48.80 (± 4.281)	7.02 (± 0.198)
MB	0.00 (± 0.000)	2.33 (± 0.094)	1.03 (± 0.093)	0.68 (± 0.072)	37.49 (± 4.726)	3.00 (± 0.000)	67.95 (± 4.841)	7.33 (± 0.148)
PC	0.10 (± 0.031)	2.21 (± 0.077)	0.96 (± 0.094)	0.50 (± 0.064)	71.90 (± 3.709)	2.67 (± 0.097)	175.67 (± 4.680)	7.93 (± 0.047)
RaC	0.00 (± 0.000)	2.34 (± 0.077)	1.14 (± 0.095)	0.45 (± 0.061)	70.53 (± 4.343)	1.90 (± 0.130)	25.05 (± 2.302)	6.68 (± 0.211)

Table 3 Continued.

Stream	%Bedrock Bank	%Herbaceous Bank	%Bare Bank	Bank Erosion	Canopy Cover	Floodplain Quality	Riparian Zone Width	Dominant Riparian Veg.
RiC	0.00 (± 0.000)	2.26 (± 0.056)	1.17 (± 0.072)	0.75 (± 0.057)	52.98 (± 3.918)	2.56 (± 0.088)	41.87 (± 3.073)	6.16 (± 0.235)
SF	0.00 (± 0.000)	2.99 (± 0.006)	0.28 (± 0.034)	0.29 (± 0.400)	1.42 (± 0.695)	3.00 (± 0.000)	1.11 (± 0.393)	2.20 (± 0.062)
UKC	0.00 (± 0.000)	2.07 (± 0.063)	1.32 (± 0.054)	0.69 (± 0.054)	78.21 (± 2.496)	2.78 (± 0.067)	63.40 (± 6.480)	6.02 (± 0.200)
WC	0.00 (± 0.000)	2.14 (± 0.059)	1.43 (± 0.061)	0.68 (± 0.064)	75.10 (± 2.707)	1.52 (± 0.103)	83.51 (± 5.611)	6.26 (± 0.203)

Table 4. Data provided by the IDNR and INHS detailing the valley segment model parameter values for each valley segment sampled within the upper Embarras River watershed. There are five categories for predicted flow class: "-99" indicates the model could not predict flow for this stream arc, "1" indicates low yield flow (<5 cm/year), "2" indicates moderate yield flow (5-10 cm/year), "3" indicates high yield flow (10-20 cm/year), and "4" indicates very high yield flow (>20 cm/year). Mean July temperature has four categories with "1" indicating cold water (<19°C), "2" indicating cool water (19-22°C), warm water (22-30°C), and very warm water (>30°C). Categories for link number were determined by looking at the relationship between link number and drainage area. There are five categories for this parameter with "1" indicating mostly first and second order streams (i.e. headwaters), "2" indicating mostly third, fourth, and some wadeable fifth order streams, "3" indicating large fourth order segments and medium sized fifth order segments that are marginally wadeable, "4" contains the remaining large fifth order and medium sized sixth order segments, and "5" contains the largest sixth order segments and all seventh and eighth order segments. Surficial geology (bedrock/colloviium and dune/lacustrine) considers the percentage of the total catchment area upstream of that arc with that type of geology.

Stream	Valley Seg.	Predicted Flow	Drainage Area	Slope (m/m)	Mean Temp.	Link #	Bedrock/ Colluvium	Dune/ Lacustrine
BF	Above	1	186.33	0.3357	3	1	0.0000	0.00
	Below	-99	203.93	0.3354	3	5	0.0000	0.00
CC	Above	3	67.82	0.4041	3	1	0.0000	0.00
	Below	1	71.97	0.4304	3	1	0.0000	0.00
HC	Above	2	87.86	0.6625	3	1	0.0000	0.00
	Above	2	98.90	0.7187	3	1	0.0000	0.00
	Below	2	108.57	0.7207	3	4	0.0000	0.00
LER	Above	2	81.24	0.2189	3	1	0.0000	0.00
	Above	2	183.04	0.2526	3	1	0.0000	0.00
	Above	2	194.24	0.2561	3	1	0.0000	0.00
	Above	2	277.47	0.2528	3	1	0.0000	0.00
	Below	2	288.84	0.2660	3	2	0.0000	0.00
LC	Above	1	27.90	0.3909	3	1	0.0000	0.00
	Above	2	37.45	0.5704	3	1	0.0000	0.00
	Below	1	43.98	0.6163	3	1	0.0000	0.00
LKC	Above	-99	69.50	0.7600	3	5	0.0000	0.00
	Below	2	77.44	0.8231	3	2	0.0000	0.00
MB	Above	2	119.02	0.8284	3	1	0.0000	0.00
	Below	1	157.77	0.8916	3	1	0.0000	0.00
PC	Above	1	73.55	0.6336	3	1	1.0000	0.00
	Below	2	75.16	0.6855	3	1	0.8824	0.00
RaC	Above	3	99.06	0.4983	3	1	0.0000	0.00
	Below	2	109.34	0.5284	3	1	0.0000	0.00
RiC	Above	1	6.77	0.5223	3	1	0.0000	0.00
	Above	1	103.78	0.2947	2	1	0.0000	0.00
	Below	2	166.02	0.3527	3	1	0.0000	0.00
SF	Above	1	9.88	0.0076	3	1	0.0000	0.00
	Above	1	27.50	0.0437	3	1	0.0000	0.56
	Below	2	35.86	0.0471	3	2	0.0000	1.00
UKC	Above	2	17.93	0.1444	3	1	0.0000	0.00
	Above	2	25.86	0.2723	3	1	0.0000	0.00
	Below	3	35.52	0.3513	2	1	0.0000	0.00
WC	Above	1	1.90	1.3527	3	1	0.0000	0.00
	Above	1	21.98	1.0424	3	1	0.0000	0.00
	Below	2	29.21	1.6379	3	1	0.0000	0.00

Table 5. Data provided by the IDNR and INHS detailing landuse characteristics for each stream arc sampled within each valley segment.

Stream	Valley Segment	Sinuosity	Percent watershed landuse				
			Agriculture	Urban	Forest	Undisturbed	Disturbed
BF	Above	1.14	93.31	0.73	0.09	0.09	94.04
	Below	1.10	92.98	1.08	0.01	0.01	94.06
CC	Above	1.15	76.30	1.48	6.22	6.19	77.78
	Below	1.04	75.10	1.79	9.94	6.90	76.89
HC	Above	1.52	75.07	0.14	10.30	10.18	75.21
	Above	1.14	74.11	0.12	10.82	10.71	74.23
	Below	1.16	73.40	0.12	11.39	11.27	73.52
LER	Above	1.51	91.59	0.99	1.77	1.67	92.59
	Above	1.23	90.13	0.87	1.78	1.77	91.01
	Above	1.15	89.50	0.89	1.91	1.90	90.34
	Above	1.20	89.76	0.92	1.94	1.92	90.68
	Below	1.31	89.29	0.89	2.20	2.18	90.19
LC	Above	1.13	81.00	0.00	7.00	7.00	81.00
	Above	1.08	77.00	0.00	11.00	11.00	77.00
	Below	1.28	73.00	0.00	13.00	13.00	73.00
LKC	Above	1.24	63.06	10.78	10.68	10.54	73.84
	Below	1.29	62.84	9.68	11.13	10.99	72.52
MB	Above	1.09	77.74	0.93	9.89	9.85	78.68
	Below	1.06	73.75	0.71	13.15	13.08	74.45
PC	Above	1.64	82.21	2.14	6.98	6.94	84.36
	Below	1.86	80.91	2.10	8.04	8.00	83.00
RaC	Above	1.12	71.19	1.55	13.09	12.91	72.74
	Below	1.18	70.44	1.41	13.95	13.78	71.84
RiC	Above	1.35	59.59	17.73	4.83	4.79	77.32
	Above	1.52	77.18	7.71	3.05	2.85	85.07
	Below	1.25	75.67	9.55	3.96	3.85	85.33
SF	Above	1.23	95.92	0.04	0.00	0.00	95.96
	Above	1.12	86.92	7.29	0.50	0.05	94.20
	Below	1.40	87.53	6.21	0.38	0.04	93.75
UKC	Above	1.12	63.50	28.47	0.99	0.97	91.97
	Above	1.36	63.35	25.51	1.99	1.91	88.86
	Below	1.40	67.00	19.70	3.17	3.01	86.69
WC	Above	1.12	66.38	0.09	12.72	12.10	66.47
	Above	1.68	74.73	0.07	14.24	13.84	74.81
	Below	1.45	63.81	0.05	23.95	23.61	63.86

included because these variables were not estimated at each transect. These variables were removed from the PCA, since missing values for each transect would result in those transects being removed from the PCA. In addition, only variables that have a complement at a larger spatial scale were included in these analyses (Palmer and Poff 1997). For example, habitat variables such as wetted width and water depth were not only collected at each transect spatial scale but also relate back to the size and position of a stream; thus these variables have complements to larger spatial scales.

Multiple Arc Valley Segments

Comparisons were made within and between stream arcs of each MAVS; these arcs were determined to be similar by the model arc merging routine and combined into one valley segment. Comparing the PCA for each MAVS, 47-62% of the variation within the habitat data was explained by the first three principle component axes (Table 6). These three axes were then used in two MANOVAs to compare environmental variability within and between stream arcs.

Within each stream arc, reaches were significantly different from each other ($2.64 \leq F \leq 18.07$; $p \leq 0.0203$; Table 7), indicating a relatively heterogeneous habitat quality within these arcs. An example of habitat variability observed between reaches within a stream arc is the upper stream arc of valley segment A in Hurricane Creek (Figure 5). While these reaches were within the same stream arc, there was a great deal of variability between these reaches, with the greatest differences in stream habitat between variables such as thalweg, substrate, percent bedrock bank, canopy cover, riparian zone width, and dominant riparian type. The influence of each habitat variable fluctuated across streams, such that different habitat variables may have been responsible for the separation

Table 6. Eigenvalues for PCA axes 1-3 for each MAVS. Data for each PCA was analyzed at the transect spatial scale. The cumulative variation explained by PCA axes 1-3 was between 47-62 % for each stream.

Stream	PCA Axis	Eigenvalue	Cumulative % Variance Explained
HC	1	3.572	23.81
	2	2.829	42.67
	3	2.173	57.16
LER	1	3.111	20.74
	2	2.431	36.94
	3	1.516	47.05
LC	1	3.186	21.24
	2	2.761	39.64
	3	1.775	51.47
RiC	1	4.598	30.66
	2	3.018	50.77
	3	1.636	61.68
SF	1	3.043	23.41
	2	2.670	43.95
	3	1.658	56.70
UKC	1	3.258	21.72
	2	2.593	39.00
	3	1.551	49.34
WC	1	3.381	22.54
	2	2.611	39.94
	3	2.026	53.45

Table 7. MANOVA results for both stream arc and stream reach spatial scales for each of the seven streams with MAVS present. Overall significance was determined at a p-value less than 0.05.

Stream	Type	Pillai's Trace	F	NumDF	DenDF	P
HC	Stream arc	0.3831231	10.35	3	50	2.078E-05
	Reach	0.97243091	16.09	6	102	6.009E-13
LER	Stream arc	0.46520981	6.61	9	324	1.117E-08
	Reach	0.50445909	12.03	6	214	1.233E-11
LC	Stream arc	0.58122576	24.06	3	52	6.679E-10
	Reach	0.35873641	3.86	6	106	0.0016
RiC	Stream arc	0.69685719	39.85	3	52	1.635E-13
	Reach	0.87609098	13.77	6	106	1.623E-11
SF	Stream arc	0.32570204	8.37	3	52	1.228E-04
	Reach	1.01132997	18.07	6	106	2.431E-14
UKC	Stream arc	0.59665904	25.15	3	51	3.976E-10
	Reach	0.9430189	15.46	6	104	1.324E-12
WC	Stream arc	0.7420349	46.02	3	48	3.674E-14
	Reach	0.27860398	2.64	6	98	0.0203

Figure 5. Graph depicting reach variability along principle components analysis axes 1-3 within the upper stream arc of the above valley segment of Hurricane Creek. On PCA axis one, the strongest loading positive habitat variables included water depth (0.4481) and thalweg (0.4442), while strongest negative variables included riparian zone width (-0.4377), substrate (-0.3885), and percent bedrock bank (-0.2945). On PCA axis two, percent bare bank (0.4513), percent bank erosion (0.3763), dominant riparian vegetation (0.3006), canopy cover (0.3002), and percent bedrock bank (0.2714) were the strongest loading positive variables while percent woody-shrub bank (-0.4266) and percent herbaceous bank (-0.2549) were the strongest negative variables. On the third PCA axis, percent herbaceous bank (0.4634), dominant riparian vegetation (0.3957), canopy cover (0.3868), and percent tree bank (0.3763) were the strongest loading positive variables while percent bank erosion (-0.4309) and percent bare bank (-0.2871) were the strongest negative variables.

Figure 6. Graph depicting habitat variability along principle components analysis axes 1-3 for the MAVS sampled within Riley Creek. On PCA axis one, the strongest loading positive habitat variables included percent woody-shrub bank (0.3284), percent bare bank (0.3282), dominant riparian type (0.2601), and thalweg (0.2541), while strongest negative variables were floodplain quality (-0.4319), percent herbaceous bank (-0.3213), and substrate (-0.2527). On PCA axis two, riparian zone width (36.48), percent bare bank (0.3532), percent bank erosion (0.3147), dominant riparian type (0.3109), and substrate (0.2511) were the strongest loading positive variables while water depth (-0.4698) and thalweg (-0.4406) were the strongest negative variables. On the third PCA axis, percent bank erosion (0.4028), wetted width (0.3929), thalweg (0.2662), and water depth (0.2587) were the strongest loading positive variables, while percent tree bank (-0.3971), percent woody-shrub bank (-0.3430), and dominant riparian type (-0.2887) were the strongest negative variables.

Figure 7. Graph depicting habitat variability along principle components analysis axes 1-3 for the MAVS sampled within Upper Kickapoo Creek. On PCA axis one, the strongest loading positive habitat variables included dominant riparian type (0.4176), riparian zone width (0.3826), percent bare bank (0.3542), canopy cover (0.3372), percent bank erosion (0.3019), and percent woody-shrub bank (0.2706), while strongest negative variable was percent herbaceous bank (-0.3342). On PCA axis two, thalweg (0.4923), water depth (0.4721), and wetted width (0.4012) were the strongest loading positive variables while canopy cover (-0.2715) and undercut bank (-0.2588) were the strongest negative variables. On the third PCA axis, undercut bank (0.4617), bank angle (0.4492), water depth (0.3023), and thalweg (0.2927) were the strongest loading positive variables while substrate (-0.4803) was the strongest negative variable.

- Stream Arc A
- ▽ Stream Arc B

of adjacent valley segments.

In all seven MAVS, stream arcs were significantly different from each other ($6.61 \leq F \leq 46.02$; $p \leq 1.23E-04$; Table 7). Two streams, Riley Creek and Upper Kickapoo Creek, best illustrate differences between stream arcs (Figures 6 and 7, respectively). Stream arc differences within the MAVS of Riley Creek have the greatest differences among variables such as wetted width, water depth, percent bank erosion, and canopy cover, while stream arcs within the MAVS of Kickapoo Creek Upper showed the greatest differences among habitat variables such as substrate, riparian zone width, and floodplain quality.

Adjacent Valley Segments

Since a great deal of habitat heterogeneity is present within MAVS, there is concern for whether or not the model arc merging routine is effectively merging similar stream arcs. Therefore, it is necessary to determine if the combined habitat variability within a MAVS is significantly different than the combined variability within the adjacent downstream SAVS. Despite the apparent habitat heterogeneity within MAVS, all stream arcs within a MAVS were combined and compared to the downstream SAVS within the same stream. Streams with two adjacent SAVS were also compared at this point.

When comparing the PCA of adjacent valley segments for each stream, 49-73% of the variation within the habitat data was explained by the first three principle component axes (Table 8). These three axes were then used to compare environmental variability within and between valley segments. Within all valley segments, reaches significantly differed from each other ($3.52 \leq F \leq 19.69$; $p \leq 0.0003$; Table 9),

Table 8. Eigenvalues for PCA axes 1-3 for each stream. Data for each PCA was analyzed at the transect spatial scale. The cumulative variation explained by PCA axes 1-3 was between 49-73% for each stream.

Stream	PCA Axis	Eigenvalue	Cumulative % Variance Explained
BF	1	5.819	41.56
	2	3.211	64.50
	3	1.317	73.91
CC	1	2.854	19.02
	2	2.692	36.97
	3	2.089	50.89
HC	1	3.631	24.20
	2	2.635	41.77
	3	2.154	56.13
LER	1	3.057	20.38
	2	2.429	36.57
	3	1.524	46.73
LC	1	2.741	18.28
	2	2.606	35.65
	3	1.632	47.03
LKC	1	2.578	18.42
	2	2.419	35.70
	3	2.307	52.18
MB	1	3.394	22.63
	2	2.942	42.24
	3	2.110	56.30
PC	1	3.022	18.89
	2	2.502	34.53
	3	1.983	46.92
RaC	1	3.075	20.50
	2	2.421	36.64
	3	1.948	49.62
RiC	1	3.668	24.46
	2	2.621	41.93
	3	1.934	54.82
SF	1	3.041	23.40
	2	2.370	41.63
	3	1.499	53.16
UKC	1	2.861	19.07
	2	2.609	36.46
	3	1.603	47.15
WC	1	2.933	19.55
	2	2.537	36.47
	3	2.122	50.62

Table 9. MANOVA results for both valley segment and reach spatial scales for each stream in the upper Embarras River watershed. Overall significance was determined at a p-value less than 0.05.

Stream	Type	Pillai's Trace	F	NumDF	DenDF	P
BF	Valley segment	0.93395942	245.13	3	52	1.166E-30
	Reach	1.0541013	19.69	6	106	2.475E-15
CC	Valley segment	0.4693152	15.03	3	51	3.903E-07
	Reach	0.7051424	9.44	6	104	2.894E-08
HC	Valley segment	0.4577553	21.39	3	76	3.814E-10
	Reach	1.2927842	11.81	15	234	1.886E-21
LER	Valley segment	0.1459291	7.52	3	132	1.110E-04
	Reach	1.1243696	7.30	33	402	5.216E-25
LC	Valley segment	0.19813255	6.51	2	79	0.0005
	Reach	0.53512681	3.52	15	243	1.733E-05
LKC	Valley segment	0.0554737	1.02	3	52	0.3923
	Reach	0.4179149	4.67	6	106	0.0003
MB	Valley segment	0.2365331	5.37	3	52	0.0027
	Reach	0.5070595	6.00	6	106	1.981E-05
PC	Valley segment	0.8037138	70.97	3	52	2.168E-18
	Reach	0.8154113	12.16	6	106	2.295E-10
RaC	Valley segment	0.428246	12.98	3	52	1.908E-06
	Reach	0.501906	5.92	6	106	2.326E-05
RiC	Valley segment	0.6553495	50.07	3	79	3.111E-18
	Reach	1.5539152	17.41	15	243	1.042E-30
SF	Valley segment	0.577842	36.04	3	79	8.853E-15
	Reach	1.18225793	10.54	15	243	2.125E-19
UKC	Valley segment	0.620696	42.00	3	77	3.470E-16
	Reach	1.3069405	12.20	15	237	3.415E-22
WC	Valley segment	0.7334337	68.79	3	75	1.759E-21
	Reach	0.9115372	6.72	15	231	5.691E-12

suggesting a great deal of habitat heterogeneity exists within each valley segment across all streams. This corresponded to observations made during data collection. An example of the reach habitat variability observed is the above valley segment of Polecat Creek (Figure 8). Reaches within this valley segment were the most variable with regard to wetted width, substrate, percent woody-shrub bank, and percent bank erosion.

In 12 of 13 streams, adjacent valley segments significantly differed from each other ($5.37 \leq F \leq 245.13$; $p \leq 0.002$; Table 9). Brushy Fork and Cottonwood Creek were used to illustrate the differences observed between adjacent valley segments (Figures 9 and 10, respectively). Valley segments within Brushy Fork had observed differences between wetted width, substrate, riparian zone width, dominant riparian vegetation, and percent herbaceous bank, while variables that were different between valley segments in Cottonwood Creek were wetted width, percent bank erosion, canopy cover, and riparian zone width. Kickapoo Creek Lower was the only stream in which adjacent valley segments were not significantly different.

Potential Longitudinal Pattern

The physical characteristics of a stream change across its longitudinal profile. Once we observed differences between valley segments, it was important to determine if the valley segment model was delineating these segments due to differences in model parameter values or if this separation was a factor of changes along the longitudinal profile of the stream. Reaches within each stream were ranked with lower values representing upstream reaches and higher values representing downstream reaches. Reaches were ranked regardless of valley segment, and an analysis of variance

Figure 8. Graph depicting reach variability along principle component axes 1-3 for reaches within the above valley segment of Polecat Creek. On PCA axis one, the strongest loading positive habitat variables included floodplain quality (0.4250), thalweg (0.3576), water depth (0.3342), and canopy cover (0.3008) while strongest negative variables were riparian zone width (-0.4297), dominant riparian vegetation (-0.3092), and substrate (-0.3053). On PCA axis two, percent bare bank (0.5365) and percent bank erosion (0.4873) were the strongest loading positive variables while percent herbaceous bank (-0.5409) was the strongest negative variable. On the third PCA axis, water depth (0.5246), wetted width (0.4987), and thalweg (0.4697) were the strongest loading positive variables, and there were no strong loading negative variables on this axis.

- Reach 1
- ▽ Reach 2
- Reach 3

Figure 9. Graph depicting habitat variability along principle component axes 1-3 for adjacent valley segments sampled within Brushy Fork. On PCA axis one, the strongest loading positive habitat variables included percent bare bank (0.3737), percent bank erosion (0.3594), dominant riparian type (0.3202), riparian zone width (0.3135), percent woody-shrub bank (0.3045), and wetted width (0.2587), while strongest negative variables were substrate (-0.3507) and percent herbaceous bank (-0.2996). On PCA axis two, water depth (0.4566), thalweg (0.4517), wetted width (0.3895), and bank angle (0.2586) were the strongest loading positive variables while floodplain quality (-0.3960) and riparian zone width (-0.2526) were the strongest negative variables. On the third PCA axis, canopy cover (0.6316) and bank angle (0.4923) were the strongest loading positive variables while percent herbaceous bank (-0.4456) was the strongest negative variable

- Valley Segment A
- ▽ Valley Segment B

Figure 10. Graph depicting habitat variability along principle component axes 1-3 for adjacent valley segments sampled within Cottonwood Creek. On PCA axis one, the strongest loading positive habitat variables included thalweg (0.4994), water depth (0.4985), wetted width (0.3933), and percent bank erosion (0.2930), while strongest negative variable was substrate (-0.3659). On PCA axis two, percent herbaceous bank (0.5157) and bank angle (0.3486) were the strongest loading positive variables while percent bare bank (-0.4425), canopy cover (-0.3070), percent tree bank (-0.2754), and percent bank erosion (-0.2723) were the strongest negative variables. On the third PCA axis, riparian zone width (0.5249) and dominant riparian type (0.5226) were the strongest loading positive variables while percent woody-shrub bank (-0.3926) and percent tree bank (-0.2968) were the strongest negative variables.

- Valley Segment A
- ▽ Valley Segment B

(ANOVA) was performed on wetted width and thalweg, two variables predicted to increase longitudinally across these valley segments.

While most reaches within a stream had significant differences between wetted width and thalweg ($p \leq 0.0367$ and $p \leq 0.0330$, respectively), Duncan post hoc tests indicated reach variability without a longitudinal pattern. For example, at Scattering Fork we expected wetted width rank to increase sequentially as the stream progressively drained more land. However, when compared to expected ranks (1, 2, 3, 4, 5, 6, 7, 8, 9), observed wetted width fluctuated unpredictably (8, 6, 2, 9, 5, 1, 3, 4, 7).

Discussion

The Illinois stream valley segment model was designed to provide stream managers with a consistent and unbiased procedure for selecting representative sampling locations as well as the ability to extrapolate reach specific information to a larger spatial context. Based on the assumptions of this model, we expected that reaches within a stream arc would have similar channel morphology and hydrology (Seelbach et al. 1997). In addition, adjacent arcs with similar morphology and hydrology are grouped together into MAVS while arcs with dissimilar characteristics are separated into different valley segment types (Seelbach et al. 1997; Brenden et al. in press). Model parameters should influence stream habitat; thus, we would expect that habitat follows a similar pattern within and between valleys segments (Montgomery 1999; Inoue and Masanori 2002; Walters et al. 2003). This project was designed to test the assumptions of the valley segment model, with the objectives to 1) detail the methodology used to test the valley segment model, 2) determine if arcs within a MAVS have similar habitat variables, 3) determine if the stream habitat within a valley segment is homogeneous, and 4) determine if different valley segments have varying habitat variables.

Sampling Methodology

The sampling methodology utilized to test the Illinois valley segment model was intended to provide a basic procedure to test differences within and between valley segment types. This methodology allows managers to randomly select sampling locations while properly spacing reaches across each valley segment. Instead of selecting individual habitat variables which do not adequately portray environmental variation within and between valley segments, a PCA was used to combine a wide range of stream

habitat variables including channel, bank, riparian, and floodplain structure into one aggregate variable (i.e. principle components axis) in order to incorporate environmental variability when testing this model. This methodology also allows us to test this model at several different spatial scales and determine how many samples are necessary for each spatial scale. Ultimately, this methodology could be used to test future refinements to the valley segment model as well as test similar models developed in other states.

Multiple Arc Valley Segments

The effectiveness of the valley segment arc merging routine was evaluated by looking at seven MAVS. Within each stream arc, a great deal of habitat variability was observed between reaches, indicating that these arcs are relatively heterogeneous in regards to stream habitat. At the reach spatial scale, this contradicts one assumption of the valley segment model that arcs are relatively homogeneous in regard to habitat quality. However, this heterogeneity could be explained by several factors including natural variation, spatial and temporal watershed land use patterns, and issues of spatial scale.

Streams are dynamic systems that are constantly changing across a landscape (Minshall 1988; Fisher 1997; Montgomery and Buffington 1998; Frothingham et al. 2002). While stream habitat characteristics tend to be generalized across a large spatial scale such as an ecoregion, stream habitat variability is present within a smaller context such as a stream arc, and stream variability at this scale is influenced by factors such as local climate, geology, hydrology, and topography (Leopold et al. 1964; Bryce and Clarke 1996; Meixler 1999; Montgomery 1999; Gomi et al. 2002; Bisson et al. 2006). Rosgen (1994) states that channel morphology can change over a relatively short scale

due to changes in geology and tributaries; this parallels what we have observed within each stream arc. Additionally, Frothingham et al. (2002) observed that reach variability existed at the link scale, with a link having a similar definition to that of a stream arc (i.e. stream section between two confluences).

While natural stream variation influences habitat heterogeneity, land draining into these systems can also influence stream heterogeneity (Schlosser 1991; Roth et al. 1996; Allan 2004). However, landuse characteristics were not selected as parameters for the valley segment model. While agricultural practices continue to dominate landuse in the Embarras River watershed, landuse adjacent to each stream was variable creating patches across the longitudinal profile of the stream (White and Harrod 1997). We hypothesize that landuse variability may be contributing to habitat heterogeneity between reaches within each stream arc. These observations parallel other studies which have documented that land use is often more variable at the reach spatial scale as compared to larger spatial scales, leading to greater habitat heterogeneity at the reach scale (Roth et al. 1996; Allan 2004).

The temporal scale at which the landscape has been altered can often influence stream habitat heterogeneity (Harding et al. 1998; Frothingham et al. 2002). For this project, reaches within each stream were sampled within a short time period, and it was unlikely that drastic landscape alterations would have occurred during this time period. However, long term temporal variation across the landscape may be contributing to habitat heterogeneity. This is particularly true in east-central Illinois where landuse has been altered gradually over time (Frothingham et al. 2002). A recently disturbed landscape may influence stream habitat structure differently compared historical

landscape disturbance (Harding et al. 1998; Allan 2004). For example, stream sections channelized fifty years ago may influence channel structure and habitat complexity differently compared to stream sections more recently channelized. Allan (2004) terms this "legacy effect," in which past stream modifications continue to affect channel structure and hydrologic variability and contribute to stream habitat heterogeneity across valley segments.

The spatial scale at which the valley segment model operates may also explain observed habitat heterogeneity. The model assigned parameter values for each stream arc, but these values may not be homogenous at a smaller spatial scale such as reaches within these arcs. For example, slope is calculated for the entire stream arc, but slope may fluctuate across various portions of this arc influencing flow regime and riffle-run-pool development at the reach scale. Therefore, valley segment model parameters cannot predict reach-scale characteristics of just one reach; however, when all reaches are pulled together, combined habitat characteristics should approximate the valley segment model parameter values.

Despite habitat heterogeneity between reaches, habitat heterogeneity was greater between stream arcs within each MAVS. At the stream arc scale, this contradicts a main assumption of the valley segment merging routine, since only similar stream arcs should be combined. While differences among arcs can be attributed to habitat variability within each arc, another explanation could be the spatial proximity of reaches within each stream arc. Reaches within each stream arc were closer in proximity to each other compared to reaches within an adjacent stream arc; thus, these reaches may be more

similar in stream habitat compared to reaches within a downstream arc (Fortin and Dale 2005).

It is clear that stream arcs are adequately defining stream environmental variation, since habitat variability within each stream arc was less than habitat variability between stream arcs. At the arc spatial scale, stream arcs are relatively homogeneous in regards to habitat quality. Thus, model assumptions of habitat homogeneity can still be substantiated at the arc spatial scale. However, these arcs are at a smaller spatial scale than valley segments, and to accurately determine whether or not these arcs were successfully combined, we need to compare them to downstream valley segments to determine whether or not this heterogeneity exists at a larger spatial scale.

Adjacent Valley Segments

A consistent pattern with regards to stream habitat within and between adjacent valley segments was observed in spite of the variety of streams used to test the valley segment model. Within each valley segment, reaches differed from each other, contradicting a main assumption of the valley segment model that stream quality is homogenous within each valley segment. Habitat heterogeneity was already expected in the MAVS, which were already observed to be relatively heterogeneous. However, it was interesting to note that SAVS were also heterogeneous with regards to habitat structure. This indicates that there is a similar pattern among all stream arcs such that stream habitat is relatively heterogeneous within the reach spatial scale. While stream habitat is heterogeneous at the reach scale, the valley segment model was never designed to predict habitat quality at this spatial scale, instead it was supposed to predict habitat quality at the valley segment spatial scale. Model parameters were calculated at the level

of stream arc, and while stream arcs show the ability to define stream variation appropriately, predictions of habitat homogeneity can only be made when comparing valley segment types.

Another pattern observed in most streams was that adjacent valley segments differed from each other, indicating that environmental variation within valley segments was less than the environmental variation among valley segments. For all streams, no predominant habitat variable was responsible for separating adjacent valley segments, rather the importance of each habitat variable fluctuated between streams. For example, water depth was the most influential habitat variable for the Little Embarras River while riparian zone width was most influential for Whetstone Creek. These observations support that a combination of all habitat variables were responsible for the separation of valley segments rather than individual habitat variables.

Lower Kickapoo Creek was the only stream in which adjacent valley segments were not different from each other. While there is no way to definitively determine why these valley segments did not differ from each other, we can speculate on some possible explanations. These adjacent valley segments had similar drainage areas; thus, one possibility is that these arcs should have been merged together into a MAVS because these segments should have similar habitat characteristics (Leopold et al. 1964; Vannote et al. 1980). However, this is an unlikely explanation since valley segments within Cottonwood Creek and Polecat Creek had similar drainage areas but still differed from each other. A more likely explanation is that environmental variation within each valley segment was so large that it did not allow for any differentiation between adjacent valley segments.

Based on the environmental variability observed within a MAVS, we expected that valley segments defined by the valley segment model would not differ from downstream SAVS. However, environmental variability within downstream SAVS was still different compared to MAVS. At the valley segment scale, the stream arc merging routine effectively combined similar stream arcs such that habitat heterogeneity within MAVS was different from other valley segment types.

Not only were MAVS different from adjacent SAVS, most adjacent SAVS were also dissimilar, indicating that environmental variability was defined appropriately for each valley segment type. Therefore, at the valley segment spatial scale we can conclude that these units have relatively homogenous habitat compared to other valley segment types. Because we see differences among valley segments, we can sample one reach within each valley segments and treat it as representative of the entire segment. The model was created as a management tool for the valley segment spatial scale, and this research validates its effectiveness at the designed spatial scale.

Potential Longitudinal Pattern

It is important to determine whether or not the valley segment model is delineating valley segments based on model parameters or if the valley segment spatial unit is sufficiently large that it encompasses natural changes in the streams thus explaining why valley segments types differ from each other. Several studies have documented that the physical characteristics of a stream change as the stream progresses from a headwater stream to a larger river (Leopold 1964; Vannote 1980). For example, drainage area increases from headwater to large river resulting in a larger volume of water at a given position within the stream (Allan and Castillo 2007).

While a longitudinal pattern may be a practical explanation for why valley segments differed from each other, this explanation is countered when wetted width and thalweg depth, two surrogates for stream size, were investigated. As we move downstream across adjacent valley segments, we expected to observe each reach get progressively wider and deeper. While there were differences in regards to these variables between reaches in most streams, these variables did not increase in the predictable longitudinal pattern across adjacent valley segments. The inability to detect a longitudinal pattern may be due to the fact that these valley segments were sampled over a relatively small portion of the total stream length, and that over this smaller spatial scale, stream width and depth fluctuate in an unpredictable fashion. Models like the river continuum concept predict these types of changes in habitat structure over a much larger spatial scale (Vannote et al. 1980). Frothingham et al. (2002) states that this longitudinal pattern operates at larger scales such as a watershed compared to smaller spatial scales. Therefore, we can conclude that a longitudinal pattern was not responsible for the differences between valley segments and that a combination of the model parameters was responsible for successfully delineating valley segments.

Management Implications

Management objectives are often geared towards several different spatial scales, depending upon the objectives of the project (Lewis et al. 1996; Bailey et al. 2003; Rieman et al. 2006). Therefore, the utility of valley segments will depend on the proposed project objectives. Three examples are given to demonstrate the versatility and appropriateness of the valley segment model.

Evaluating watershed quality is a common management practice. Watersheds are evaluated in order to identify high quality areas, monitor long term trends in stream quality, and target potential water quality problems (Miller et al. 2006). Typically a manager will sample a set number of reaches within a watershed in order to determine the overall quality of that watershed. Due to the spatial constraints of valley segments, one reach cannot be identified as a representative reach for an entire watershed. However, one reach can be representative of a valley segments, and several valley segments sampled across a watershed would provide an idea of the habitat diversity present within the entire watershed. Utilizing the valley segment model could reduce the number of reaches typically sampled within a watershed, thus saving valuable time and resources.

In addition, valley segments of high quality habitat can be identified and protected in order to preserve habitat diversity; thus allowing for the identification of stream references segments and target valley segments in need of restoration. In the past, reference reach locations were selected because it was determined that these reaches were minimally impacted by water quality stressors (Bailey et al. 2003). After these locations were identified, managers would compare these locations to other locations of unknown habitat quality. Typically this is done to determine stream segments with degraded habitat to focus on for restoration projects. Utilizing the model at the valley segment scale, one reach can be sampled within a reference valley segment while another reach can be sampled within a valley segment of unknown quality. Then the two reaches can be compared to each other to determine the habitat quality of the unknown valley segment.

At the valley segment spatial scale, managers may also need to reconsider reach length. Traditionally, the IDNR and INHS sample reaches between 100-300 meters in total length (IDNR 2002). However, it will be important to determine if the traditional reach length is sufficient within the context of the valley segment model. A larger reach may incorporate more of the existing habitat heterogeneity within each valley segment; thus making these reaches more "representative" of the entire valley segment.

Once a relatively poor quality valley segment has been identified, a biologist may want to determine the range of habitat condition within a valley segment and identify regions for possible habitat restoration. According to the reach spatial scale, the amount of habitat heterogeneity present within each valley segment requires that multiple reaches would need to be sampled within this valley segment in order to identify possible restoration reaches. This would be particularly beneficial when working with MAVS, which are composed of several stream arcs, and it may be important to identify which arc is in most need of stream restoration. This may be particularly useful if a stream manager was looking at a habitat-specific species such as darters, where it may be important to determine where riffles are present throughout the segment and identify areas to place artificial riffles.

The valley segment model is appropriate for several stream projects and at several different spatial scales. However, the utility of this model declines when the appropriate spatial scale is not considered in conjunction with the management objectives of the project. It is important to determine which spatial scale is appropriate for a project and then determine the number of reaches necessary for that spatial scale. Failure to sample a

sufficient number of reaches could result in the misuse of the valley segment model and lead to the misrepresentation of reach-scale data to a larger spatial context.

Future Research

It has been documented that stream habitat influences aquatic community structure such that with habitat complexity, aquatic community diversity tends to increase (Gorman and Karr 1978; Schlosser 1982; Beisel et al 1998). With this project, a consistent pattern of habitat variability within and between adjacent valley segments was observed. While it is known that habitat influences the type and quality of aquatic communities within a particular reach, it will be important to determine how aquatic communities operate within valley segments. Habitat heterogeneity across a valley segment scale is less likely to create fish community heterogeneity since fish operate at a more regional spatial scale (Angermeier and Winston 1998). Therefore, a hypothesis would be that fish communities are relatively homogenous within each valley segment. Conversely, macroinvertebrate communities tend to operate at a more local spatial scale (Richards et al. 1997; Nerbonne and Vondracek 2001). With macroinvertebrate communities, the expectation might be that there is some degree of community heterogeneity within valley segments. Research still needs to be conducted to determine whether or not aquatic communities respond in a predictable pattern similar to habitat heterogeneity within and between valley segments.

Stream habitat variables have been documented to change seasonally (Gorman and Karr 1978; Capone and Kushlan 1991). Water depth changes seasonally from spring, where streams are periodically inundated by precipitation events to summer, where streams are typically losing water as they replace groundwater. Therefore, it may be

pertinent to revisit the valleys segment model parameters to determine their effectiveness at defining valleys segments over a temporal scale. Future research projects will contribute to the continual refinement of the Illinois valley segment model.

Conclusions

Our examination of the valley segment model revealed a consistent pattern of environmental variation within and between valley segments. We found that this model performed well at grouping arcs into distinct valley segment types despite habitat heterogeneity between reaches within stream arcs. This was due to the fact that there was a greater environmental variability among valley segments compared to environmental variability within valley segments. Thus, we can conclude that this model encompasses both spatial and temporal scales while integrating both structure and function into a relatively low cost procedure that creates consistent understanding among stream managers (Seelbach et al. 1997). The valley segment model is an effective management tool because it 1) retains the stream's natural ecological variation, 2) allows for modeling and extrapolation of stream systems, and 3) provides both accurate and practical state-wide planning of representative sites and identification of stream impairments. Given the benefits of this model in conjunction with our observations that this model is effectively partitioning stream habitat variation, we can conclude that valley segments are an appropriate system for evaluating, managing, and extrapolating data on stream condition.

References

- Adams, M.B., and J.N. Kochenderfer. 2007. Long-term stream chemistry monitoring on the fernow experiment forest: implications for sustainable management of hardwood forests. Pages 97 to 106 in D.S. Buckley and W.K. Clatterbuck, editors. Proceedings of the 15th central hardwood forest conference. U.S. Department of Agriculture, Forest Service, Southern Research Station, Asheville, North Carolina.
- Allan, J.D. 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. *Annual Reviews of Ecology, Evolution, and Systematics* 35: 257-284.
- Allan, J.D., D.L. Erickson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. *Freshwater Biology* 37: 149-161.
- Allan, J.D., and M.M. Castillo. 2007. Stream ecology: structure and function of running waters, 2nd Edition. Springer, Dordrecht, The Netherlands.
- Angermeier, P.L., and J.R. Karr. 1984. Relationships between woody debris and fish habitat in a small warmwater stream. *Transactions of the American Fisheries Society* 113: 716-726.
- Angermeier, P.L., and M.R. Winston. 1998. Local versus regional influences on local diversity in stream fish communities of Virginia. *Ecology* 79: 911-927.
- Bailey, R.C., R.H. Norris, and T.B. Reynoldson. 2003. Bioassessment of freshwater ecosystems: using the reference condition approach. Springer, New York.
- Barbour, M.T., T.J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates, and fish, 2nd edition. United States Environmental Protection Agency, Office of Water, EPA 841-B-99-002. Washington, D.C.
- Bartholow, J.M. 2000. Estimating cumulative effects of clearcutting on stream temperatures. *River* 7: 284-297.
- Berkman, H.E., and C.F. Rabeni. 1987. Effect of siltation on stream fish communities. *Environmental Biology of Fishes* 18: 285-294.
- Bisson, P.A., D.R. Montgomery, J.M. Buffington. 2006. Valley segments, stream reaches, and channel units. Pages 23-49 in F.R. Hauer and J.M. Buffington, editors. *Methods in stream ecology*, 2nd edition. Academic Press, San Diego, California.
- Bohn, C. 1986. Biological importance of stream bank stability. *Rangelands* 8: 55-56.

- Brenden, T.O., L. Wang, P.W. Seelbach, and R.D. Clark Jr. In press. A spatially-constrained clustering program for river valley segment delineation from GIS river networks. *Environmental Modeling and Software*.
- Brenden, T.O., R.D. Clark, A.R. Cooper, P.W. Seelbach, L. Wang, S.S. Aichele, E.G. Bissell, and J.S. Stewart. 2006. A GIS framework for collecting, managing, and analyzing multi-scale landscape variables across large regions for river conservation and management. Pages 49-74 in R.M. Hughes, L. Wang, and P.W. Seelbach, editors. *Landscape influences on stream habitats and biological assemblages*. American Fisheries Society, Symposium 48, Bethesda, Maryland.
- Bryce, S.A., and S.E. Clarke. 1996. Landscape-level ecological regions: linking state-level ecoregion frameworks with stream habitat classifications. *Environmental Management* 20: 297-311.
- Capone, T.A., and J.A. Kushlan. 1991. Fish community structure in dry-season stream pools. *Ecology* 72: 983-992.
- Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley, and V.H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* 8: 559-568.
- Conrad, R.C., and S.S. Warner. 2005. Fish community report: west fork of the White River and selected tributaries. Muncie Sanitary District's Bureau of Water Quality. Muncie, Indiana.
- Crunkilton, R., J. Kleist, J. Ramcheck, W. DeVita, and D. Villeneuve. 1996. Assessment of the response of aquatic organisms to long-term *insitu* responses to urban runoff. Pages 95-111 in L.A. Roesner, editor. *Effects of watershed development and management on aquatic ecosystems*. American Society of Civil Engineers, New York.
- Diana, M., J.D. Allan, and D. Infante. 2006. The influence of physical habitat and land use on stream fish assemblages in Southeastern Michigan. Pages 359-374 in R.M. Hughes, L. Wang, and P.W. Seelbach, editors. *Landscape influences on stream habitats and biological assemblages*. American Fisheries Society, Symposium 48, Bethesda, Maryland.
- ESRI. 2006. Arc/Info user's manual. Environmental Systems Research Institute, Redlands, California.
- Figuerola-Nieves, D., T.V. Royer, and M.B. David. 2006. Controls on chlorophyll-a in nutrient-rich agricultural streams in Illinois, USA. *Hydrobiologia* 568: 287-298.

- Fisher, S.G. 1997. Creativity, idea generation, and the functional morphology of streams. *Journal of the North American Benthological Society* 16: 305-318.
- Fitzpatrick, F.A., M.A. Harris, T.L. Arnold, and K.D. Richards. 2005. Urbanization influences on aquatic communities in northeastern Illinois streams. *Journal of the American Water Resources Association* 40: 461-475.
- Fortin, M, and M. Dale. 2005. *Spatial analysis: a guide for ecologists*. Cambridge University Press, Cambridge, United Kingdom.
- Frissell, C.A., W.J. Liss, C.E. Warren, and M.D. Hurley. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. *Environmental Management* 10: 199-214.
- Frothingham, K.M., B.L. Rhoads, and E.E. Herricks. 2002. A multi-scale conceptual framework for integrated ecogeomorphological research to support stream naturalization in the agricultural Midwest. *Environmental Management* 29: 16-33.
- Gomi, T., R.C. Sidle, and J.S. Richardson. 2002. Understanding processes and downstream linkages of headwater systems. *BioScience* 52: 905-916.
- Gorman, O.T., and J.R. Karr. 1978. Habitat structure and stream fish communities. *Ecology* 59: 507-515.
- Gotelli, N.J., and A.M. Ellison. 2004. *A primer of ecological statistics*. Sinauer Associates, Inc, Sunderland, Massachusetts.
- Gregory, S.V., F.J. Swanson, W.A. McKee, and K.W. Cummins. 1991. An ecosystem perspective of riparian zones. *Bioscience* 41: 540-551.
- Hannaford, M.J., M.T. Barbour, and V.H. Resh. 1997. Training reduces observer variability in visual- based assessments of stream habitat. *Journal of the North American Benthological Society* 16: 853-860.
- Harding, J.S., E.F. Benfield, P.V. Bolstad, G.S. Helfman, and E.B.D Jones, III. 1998. Stream biodiversity: the ghost of land use past. *Proceedings of the National Academy of Science* 95: 14843-14847.
- Harper, D. 1992. *Eutrophication of freshwaters: principles, problems, and restoration*. Chapman and Hall, New York, New York.
- Hawkins, C.P., R.H. Norris, J. Gerritsen, R.M. Hughes, S.K. Jackson, R.K. Johnson, and R.J. Stevenson. 2000. Evaluation of the use of landscape classifications for the prediction of freshwater biota: synthesis and recommendations. *Journal of the North American Benthological Society* 19: 541-556.

- Hayano, M.Y., O.H. Nagata, and I.R. Irvine. 2002. Importance of instream cover for young masu salmon, *Oncorhynchus masou*, in autumn and winter. *Fisheries Management and Ecology* 9: 217-223.
- IDNR (Illinois Department of Natural Resources). 1996. Inventory of resource rich areas in Illinois: an evaluation of ecological resources. IDNR, Center for Wildlife Ecology, Springfield, Illinois.
- IDNR (Illinois Department of Natural Resources). 2000. Draft manual for calculating index of biotic integrity scores for streams in Illinois. IDNR, Division of Fisheries, Springfield, Illinois.
- IDNR (Illinois Department of Natural Resources). 2002. Fisheries stream sampling guidelines. IDNR, Division of Fisheries, Springfield, Illinois.
- Infante, D.M., M.J. Wiley, and P.W. Seelbach. 2006. Relationship among channel shape, catchment characteristics, and fish in lower Michigan streams. *American Fisheries Society Symposium* 48: 339-357.
- Inoue, N, and Masanori M. 2002. Effects of longitudinal variation in stream habitat structure on fish abundance: an analysis based on subunit-scale habitat classification. *Freshwater Biology* 47: 1594-1607.
- Johnson, L.B., C. Richards, G.E. Host, and J.W. Arthur. 1997. Landscape influences on water chemistry in Midwestern stream ecosystems. *Freshwater Biology* 37: 193-208.
- Karr, J.R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6: 21-27.
- Karr, J.R. 1991. Biological integrity: a long-neglected aspect of water resource management. *Ecological Applications* 1: 66-84.
- Karr, J.R., and D.R. Dudley. 1981. Ecological perspective on water quality goals. *Environmental Management* 5: 55-68.
- Karr, J.R., and E.W. Chu. 2000. Sustaining living rivers. *Hydrobiologia* 422/423: 1-14.
- Karr, J.R., L.A. Toth, and D.R. Dudley. 1985. Fish communities of Midwestern streams: a history of degradation. *BioScience* 35: 90-95.
- Karr, J.R., P.L. Angermeier, and I.J. Schlosser. 1983. Habitat structure and fish communities of warmwater streams. U.S. Environmental Protection Agency, EPA-600/D-83-094, Corvallis Oregon.

- Kilgour, B.W. and L.W. Stanfield. 2006. Hindcasting reference conditions in streams. Pages 623-639 in R.M. Hughes, L. Wang, and P.W. Seelbach, editors. Landscape influences on stream habitats and biological assemblages. American Fisheries Society, Symposium 48, Bethesda, Maryland.
- Lammert, H., and J.D. Allan. 1999. Assessing the biotic integrity of streams: effects of scale in measuring the influence of land use/cover and habitat structure on fish and macroinvertebrates. *Environmental Management* 23: 257-270.
- Larson, L.L., and S.L. Larson. 1996. Riparian shade and stream temperature: a perspective. *Rangelands* 18: 149-152.
- Lau, J.K., T.E. Lauer, and M.L. Weinman. 2006. Impacts of channelization on stream habitats and associated fish assemblages in east central Indiana. *American Midland Naturalist* 156: 319-330.
- Lawrence, G.B., B. Momen, and K.M. Roy. 2004. Use of stream chemistry for monitoring acidic deposition effects in the Adirondack region of New York. *Journal of Environmental Quality* 33: 1002-1009.
- Lemmon, P.E. 1956. A spherical densiometer for estimating forest understory density. *Forest Science* 2: 314-320.
- Leopold, L.B., M.G. Wolman, and J.P. Miller. 1964. Fluvial processes in geomorphology. Freeman, San Francisco, California.
- Lewis, C.A., N.P. Lester, A.D. Bradshaw, J.E. Fitzgibbon, K. Fuller, L. Hakanson, and C. Richards. 1996. Considerations of scale in habitat conservation and restoration. *Canadian Journal of Aquatic Science* 53 (S1): 440-445.
- Maddock, I. 1999. The importance of physical habitat assessment for evaluating river health. *Freshwater Biology* 41: 373-391.
- McCune, B., and J.B. Grace. 2002. Analysis of ecological communities. MjM Software Design, Gleneden Beach, Oregon.
- Meador, M.R., and R.M. Goldstein. 2003. Assessing water quality at large geographic scales: relations among land use, water physicochemistry, riparian condition, and fish community structure. *Environmental Management* 31: 504-517.
- Meixler, M.S. 1999. Regional setting. Pages 11-24 in M.B. Bain and N.J. Stevenson, editors. Aquatic habitat assessment: common methods. American Fisheries Society, Bethesda, Maryland.
- Miller, M.A., A.C.C. Colby, P. Kanehl. 2006. Report on the regional monitoring and assessment program study of wadeable streams in the driftless area ecoregion of

western Wisconsin. U.S. Environmental Protection Agency, Office of Research and Development, EPA 600/R-06/165, Washington, D.C.

- Minshall, G.W. 1988. Stream ecosystem theory: a global perspective. *Journal of the North American Benthological Society* 7: 263-288.
- Minshall, G.W., K.W. Cummins, R.C. Peterson, C.E. Cushing, D.A. Bruns, J.R. Sedell, and R.L. Vannote. 1985. Developments in stream ecosystem theory. *Canadian Journal of Fisheries and Aquatic Sciences* 42: 1045-1055.
- Mitner, R.J., and E.T. Rankin. 1998. Primary nutrients and the biotic integrity of rivers and streams. *Freshwater Biology* 40: 145-158.
- Montgomery, D.R. 1999. Process domains and the river continuum. *Journal of the American Water Resources Association* 35: 397-410.
- Montgomery, D.R., and J.M. Buffington. 1998. Channel processes, classification, and response. Pages 13-42 in R.J. Naiman and R.E. Bilby, editors. *River ecology and management: lessons learned from the Pacific coastal region*. Springer, New York.
- Montgomery, D.R., and L.H. MacDonald. 2002. Diagnostic approach to stream channel assessment and monitoring. *Journal of the American Water Resources Association* 38: 1-16.
- Morgan, A.M., T.V. Royer, M.B. David, and L.E. Gentry. 2006. Relationships among nutrients, chlorophyll-a, and dissolved oxygen in agricultural streams in Illinois. *Journal of Environmental Quality* 35: 1110-1117.
- Nerbonne, B.A., and B. Vondracek. 2001. Effects of local land use on physical habitat, benthic macroinvertebrates, and fish in the Whitewater River, Minnesota, USA. *Environmental Management* 28: 87-99.
- Ney, J.J. 1999. Practical use of biological statistics. Pages 167-191 in C.C. Kohler and W.A. Hubert, editors. *Inland fisheries management in North America*, 2nd edition. American Fisheries Society, Bethesda, Maryland.
- OEPA (Ohio Environmental Protection Agency). 1994. Ohio water resources inventory. OHEPA, Division of Water Quality Monitoring and Assessment, Report 305(b), Columbus, Ohio.
- Oscos, J., P.M. Leunda, R. Miranda, C. Garcia-Fresca, F. Campos, and M.C. Escala. 2005. River channelization effects on fish population structure in the Larraun River (Northern Spain). *Hydrobiologia* 543: 191-198.

- Palmer, M.A., and N.L. Poff. 1997. The influence of environmental heterogeneity on patterns and processes in streams. *Journal of the North American Benthological Society* 16: 169-173.
- Paul, M.J. and J.L. Meyer. 2001. Streams in the urban landscape. *Annual Review of Ecology and Systematics* 32: 333-365.
- Plafkin, J.L., M.T. Barbour, K.D. Porter, S.K. Gross, and R.M. Hughes. 1989. Rapid bioassessment protocols for use in streams and rivers, benthic macroinvertebrates and fish. U.S. Environmental Protection Agency, EPA 440-4-89-001, Washington, D.C.
- Platts, W.S. 1974. Geomorphic and aquatic conditions influencing salmonids and stream classification—with application to ecosystem classification. US Forest Service, Inter-mountain Forest and Range Experiment Station, Boise, Idaho.
- Rankin, E.T. 1989. The qualitative habitat evaluation index (QHED): rationale, methods, and application. Ohio Environmental Protection Agency, Division of Water Quality Planning and Assessment. Columbus, OH.
- Rabeni, C.F., and R.B. Jacobson. 1999. Warmwater streams. Pages 505-525 in C.C. Kohler and W.A. Hubert, editors. *Inland fisheries management in North America*, 2nd edition. American Fisheries Society, Bethesda, Maryland.
- Rasmussen, J.A. 2007. Classification of Illinois lakes and reservoirs and evaluation of the potential use of phytoplankton as biocriteria. Master's Thesis. Eastern Illinois University, Charleston, Illinois.
- Rhoads, B.L. and E.E. Herricks. 1996. Naturalization of headwaters streams in Illinois: challenges and possibilities. Pages 331-367 in A. Brooks and F.D. Shields Jr., editors. *River channel restoration: guiding principles for sustainable projects*. John Wiley and Sons. New York.
- Richards, C., R. Haro, L. Johnson, and G. Host. 1997. Catchment and reach-scale properties as indicators of macroinvertebrate species traits. *Freshwater Biology* 37: 219-230.
- Rieman, B., J. Dunham, and J. Clayton. 2006. Emerging concepts for management of river ecosystems and challenges to applied integration of physical and biological sciences in the Pacific Northwest, USA. *International Journal of River Basin Management* 4: 85-97.
- Rogers, C.E., D.J. Brabander, M.T. Barbour, and H.F. Hemond. 2002. Use of physical, chemical, and biological indices to assess impacts of contaminants and physical habitat alteration in urban streams. *Environmental Toxicology and Chemistry* 21: 1156-1167.

- Roper, B.B., and D.L. Scarnecchia. 1995. Observer variability in classifying habitat types in stream surveys. *North American Journal of Fisheries Management* 15: 46-53.
- Rosgen, D.L. 1994. A classification of natural rivers. *Catena* 22: 169-199.
- Roth, N.E., J.D. Allan, and D.L. Erickson. 1996. Landscape influences on stream biotic integrity assessed at multiple spatial scales. *Landscape Ecology* 11: 141-156.
- Royer, T.V., M.B. David, and L.E. Gentry. 2006. Timing of riverine export of nitrate and phosphorus from agricultural watersheds in Illinois: implications for reducing nutrient loading to the Mississippi River. *Environmental Science and Technology* 40: 4126-4131.
- Schlosser, I.J. 1982. Fish community structure and function along two habitat gradients in a headwater stream. *Ecological Monographs* 52: 395-414.
- Schlosser, I.J. 1991. Stream fish ecology: a landscape perspective. *BioScience* 41: 704-712.
- Seelbach, P.W., L. Wang, R.D. Clark, T. Brenden, A. Cooper, M.J. Wiley, D. Allan, L. Hinz, K.E. Wehrly, T. Zorn, E. Baker, R.J. Stevenson, B. Pijanowski, J. Lyons, M. Mitro, A. Holtrop, D. Day, and M. DePhilip. 2005. Ecological classification for environmental assessment: demonstration, validation, and application to regional assessment across Illinois, Michigan, and Wisconsin. United States Environmental Protection Agency Grant Number R-83059601-0. Ann Arbor, Michigan.
- Seelbach, P.W., and M.J. Wiley. 1997. Overview of the Michigan rivers inventory (MRI) project. Michigan Department of Natural Resources, Fisheries Division. Fisheries Technical Report 97-3, Lansing, Michigan.
- Seelbach, P.W., M.J. Wiley, J.C. Kotanchik, and M.E. Baker. 1997. A landscape-based ecological classification system for river valley segments in lower Michigan (MI-SEV Version 1.0). Michigan Department of Natural Resources, Research Report 2036, Lansing, Michigan.
- Seelbach P.W., M.J. Wiley, M.E. Baker, and K.E. Wehrly. 2006. Initial classification of river valley segments across Michigan's lower peninsula. Pages 25-48 in R.M. Hughes, L. Wang, and P.W. Seelbach, editors. *Landscape influences on stream habitats and Biological assemblages*. American Fisheries Society, Symposium 48, Bethesda, Maryland.
- Sheehan, R.J. and J.L. Rasmussen. 1999. Large Rivers. Pages 431-454 in C.C. Kohler and W.A. Hubert, editors. *Inland fisheries management in North America*, 2nd edition. American Fisheries Society, Bethesda, Maryland.

- Shields, F.D., A.J. Bowie, and C.M. Cooper. 1995. Control of streambank erosion due to bed degradation with vegetation and structure. *Journal of the American Water Resources Association* 31: 475-489.
- Simon, T.P. 1999. Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, New York.
- Simon, T.P. and R. Dufour. 1997. Development of index of biotic integrity expectations for the core regions of Indiana. V. U.S. Environmental Protection Agency, Region V Water Division, Watershed and Nonpoint Source Branch, EPA 905/R-96/002, Chicago, Illinois.
- Tabachnik, B.G., and L.S. Fidell. 1989. Using multivariate statistics, 2nd edition. Harper and Row, New York.
- Talmage, P.J., J.A. Perry, and R.M. Goldstein. 2002. Relation of instream habitat and physical conditions to fish communities of agricultural streams in the northern Midwest. *North American Journal of Fisheries Management* 22: 825-833.
- USDA (United States Department of Agriculture). 2005. Guide to identification of bankfull stage in the Northeastern United States. USDA, General Technical Report RMRS-GTR-133-CD, Washington, D.C.
- USEPA (United States Environmental Protection Agency). 2004. Wadeable stream assessment: field operations manual. USEPA, Office of Water and Office of Research and Development, EPA841-B-04-004, Washington, D.C.
- USEPA (United States Environmental Protection Agency). 2002. National water quality inventory: report to congress, 2002 reporting cycle. USEPA, Office of Water, EPA-841-R-07-001, Washington, D.C.
- USEPA (United States Environmental Protection Agency). 1990. Biological criteria: national program for guidance for surface waters. USEPA, Office of Water Regulations and Standards, EPA-440/5-90-004, Washington, D.C.
- Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* 37: 130-137.
- Walters, D.M., D.S. Leigh, M.C. Freeman, B.J. Freeman, and C.M. Pringle. 2003. Geomorphology and fish assemblages in a Piedmont river basin, USA. *Freshwater Biology* 48: 1950-1970.

- Wang, L., J. Lyons, and P. Kanehl. 2002. Effects of watershed best management practices on habitat and fish in Wisconsin streams. *Journal of the American Water Resources Association* 38: 663-680.
- Wang, L., J. Lyons, and P. Kanehl. 1998. Development and evaluation of a habitat rating system for low-gradient Wisconsin streams. *North American Journal of Fisheries Management* 18: 775-785.
- Wang, L., P.W. Seelbach, and R.M. Hughes. 2006. Introduction to landscape influences on stream habitats and biological assemblages. Pages 1-23 in R.M. Hughes, L. Wang, and P.W. Seelbach, editors. *Landscape influences on stream habitats and biological assemblages*. American Fisheries Society, Symposium 48, Bethesda, Maryland.
- Wang, L.J., T.D. Simonson, and J. Lyons. 1996. Accuracy and precision of selected stream habitat estimates. *North American Journal of Fisheries Management* 16: 340-347.
- Weigel, B.M., L.J. Henne, and L.M. Martinez-Rivera. 2002. Macroinvertebrate-based index of biotic integrity for protection of streams in West-Central Mexico. *Journal of the North American Benthological Society* 21: 686-700.
- Wetzel, R.G. 2001. *Limnology: lake and river ecosystem*, 3rd edition. Academic Press, New York.
- White, P.S., and J. Harrod. 1997. Disturbance and diversity in a landscape context. Pages 128-159 in J.A. Bissonette, editor. *Wildlife and landscape ecology: effects of pattern and scale*. Springer, New York.
- Winger P.V. 1981. Physical and chemical characteristics of warmwater streams: a review. American Fisheries Society, Warmwater Streams Symposium: 32-44.
- Wolman, M.G. 1954. Method of sampling coarse level substrate. *Transactions of the American Geophysical Union (EOS)* 35: 951-956.
- Zwick, P. 1992. Stream habitat fragmentation: a threat to biodiversity. *Biodiversity and Conservation* 1: 80-97.