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Landscape influences on stream biotic integrity assessed at multiple spatial scales

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Abstract

The biological integrity of stream ecosystems depends critically on human activities that affect land use/cover along stream margins and possibly throughout the catchment. We evaluated stream condition using an Index of Biotic Integrity (IBI) and a habitat index (HI), and compared these measures to landscape and riparian conditions assessed at different spatial scales in a largely agricultural Midwestern watershed. Our goal was to determine whether land use/cover was an effective predictor of stream integrity, and if so, at what spatial scale. Twenty-three sites in first-through third-order headwater streams were surveyed by electrofishing and site IBIs were calculated based on ten metrics of the fish collection. Habitat features were characterized through field observation, and site HIs calculated from nine instream and bank metrics. Field surveys, aerial photograph interpretation, and geographic information system (GIS) analyses provided assessments of forested land and other vegetation covers at the local, reach, and regional (catchment) scales.

The range of conditions among the 23 sites varied from poor to very good based on IBI and HI scores, and habitat and fish assemblage measures were highly correlated. Stream biotic integrity and habitat quality were negatively correlated with the extent of agriculture and positively correlated with extent of wetlands and forest. Correlations were strongest at the catchment scale (IBI with % area as agriculture, $r^2 = 0.50$, HI with agriculture, $r^2 = 0.76$), and tended to become weak and non-significant at local scales. Local riparian vegetation was a weak secondary predictor of stream integrity. In this watershed, regional land use is the primary determinant of stream conditions, able to overwhelm the ability of local site vegetation to support high-quality habitat and biotic communities.

1. Introduction

Stream ecosystems often are adversely affected by human alteration of surrounding lands. When naturally vegetated landscapes are converted to urban or agricultural uses, physical and biological relationships with adjacent streams are affected, usually resulting in habitat degradation and negative impacts on stream biota (Karr and Schlosser 1978; Schlosser 1991). A naturally vegetated catchment, or at least a protected riparian zone, is widely viewed as critical to the biological integrity of river ecosystems (Gregory *et al.* 1991; Naiman 1992; Sweeney 1992). Riparian vegetation serves many functions important to the maintenance of natural stream processes (Karr and Schlosser 1978; Petersen *et al.* 1987). Inputs of organic litter and large woody debris to streams with natural riparian vegetation are of great importance to ecosystem function (Gregory *et al.* 1991). Woody debris provides cover for organisms and influences the development of channel morphology (Karr and Schlosser 1978; Naiman *et al.* 1993). Leaf litter and other organic detritus supply energy subsidies to the aquatic food web (Vannote *et al.* 1980). Streamside vegetation provides shade necessary for natural temperature regimes, thus preventing excessive summer warming (Barton *et al.* 1985). By reducing

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overland flow of water to stream channels, riparian vegetation also regulates sediment transport (Osborne and Kovacic 1993) and moderates terrestrial inputs of nutrients from agricultural sources (Lowrance *et al.* 1984). Natural vegetation can moderate streamflow extremes during normal flood/drought cycles.

Landscape alteration and the removal of riparian vegetation potentially can affect the species diversity and assemblage composition of fish communities through a number of adverse changes to the stream system. The removal of streamside vegetation may cause a shift from allochthonous (terrestrial) to autochthonous (algal) production within low-order streams (Minshall *et al.* 1985; Delong and Brusven 1993). The absence of woody debris, as in many agricultural streams, causes a reduction in heterogeneity of depth, substrate, and current velocity, resulting in wide, shallow streams with little structural complexity and affording poor habitat for many aquatic species (Schlosser 1991). Along with the installation of artificial tile drainage systems, the removal of riparian vegetation alters runoff patterns and creates "flashy" streams with more extreme high and low flows, increased scouring, and greater streambank erosion (Schlosser 1991). Removal of native vegetation also increases the potential for overland and channel erosion, causing increased siltation of stream bottoms and obliterating the clean gravel surfaces required as spawning habitat by many species (Berkman and Rabeni 1987). Habitat alteration is cited as a contributing cause in the decline of 73% of fish species extinctions in North America (Miller *et al.* 1989), and declining fish populations in many midwestern rivers have been paralleled by landscape changes and habitat degradation during this century (Karr *et al.* 1985; Trautman 1981).

Rivers are hierarchial systems, having characteristics evident over a range of spatial scales from the microhabitat to the entire stream network (Frissell *et al.* 1986; Sedell *et al.* 1990). The functional benefits of intact vegetation within the watershed are likely to be scale-dependent. At the local scale, of a few meters to a few hundreds of meters, riparian vegetation can influence instream habitat by providing shade, inputs of organic matter and woody debris, and perhaps aid in maintaining local bank and channel stability. However, the extent of riparian cover over some much larger spatial scale must be of greater importance to overall stream nutrient and sediment inputs, water temperature, energy sources, and flow regime. For example, the fraction of forested streambank within 2.5 km upstream of sites in southern Ontario streams was an effective predictor of weekly maximum water temperatures and hence the presence of brook trout (Barton *et al.* 1985).

A full understanding of the adverse effects of the loss of riparian vegetation requires scale-appropriate analysis (O'Neill *et al.* 1989). At the local scale, several studies have used semi-quantitative or categorical methods to characterize riparian features such as vegetation type, height, slope, riparian zone width, and channel features (Delong and Brusven 1991; Petersen 1992; Platts *et al.* 1983). By mapping instream woody debris within a series of 45 to 70 m stream reaches in British Columbia, Fausch and Northcote (1992) demonstrated that instream woody debris correlated with streamside forest composition adjacent to sites. Nutrient concentrations, monitored at various intervals between a stream and the edge of an agricultural field, have exhibited a marked decrease in traversing grass and forested vegetated buffer strips (Osborne and Kovacic 1993; Lowrance *et al.* 1984; Peterjohn and Correll 1984).

At the broader scale of the watershed or entire stream, several studies have used maps or a geographic information system (GIS) to relate land cover patterns to instream conditions. Steedman (1988) reported that biotic integrity (IBI, Karr *et al.* 1986) of Ontario streams was related to the proportion of stream channel length with riparian forest coverage. In a nationwide study of the influence of upland and streamside land uses on stream nutrient concentrations, Omernik *et al.* (1981) found that stream nitrate and phosphorus concentrations were strongly related to watershed land uses, but not to land use near the stream margins. Utilizing the buffer and overlay functions of a **GIS** to isolate riparian buffer areas, Osborne and Wiley (1988) concluded that riparian land uses nearest to the stream were most closely related to instream nutrient concentrations.

Our study examined 23 stream sites within an agricultural and urbanizing landscape in southeastern Michigan to investigate stream condition as a function of land use/cover. We used the IBI based on the species richness and composition of stream fish assemblages as a biological assessment tool (Karr *et al.* 1986; Karr 1991), and evaluated habitat using an index (HI) of habitat quality and diversity (MDNR 1991). Land use/cover was quantified at several spatial scales to determine if measurement scale influenced the effectiveness of vegetation measures as predictors of instream ecological status.

2. Methods

2.1. Study sites

The River Raisin, with a mainstem length of 216 km, drains an upland region of glacial moraine and till plain geography with a diverse mixture of forested land, wetlands and small farms, and an extensive lake plain of clay soils dominated by corn and soybean agriculture. Historically a region of wetlands and hardwood forest, the 2776 km^2 watershed is now dominated by agricultural land uses and supports a human population of 130,000 (1990 census), fairly well dispersed throughout its area. Twenty-three stream locations were chosen as study sites, located on wadeable first- through third-order tributaries representative of the range of land uses in the upper half of the River Raisin drainage basin (Fig. 1; Appendix). To minimize differences due to site location, all sampling was conducted with the Eastern Cornbelt Plains ecoregion. To facilitate comparisons of riparian conditions with IBI and habitat variables, sites were chosen that ranged from little or no natural vegetation, to narrow strips of vegetation adjoining an agricultural field, to wider floodplain forests and woodlots.

2.2. Land use/cover

Land use/cover was quantified at local, reach, and catchment scales (Fig. 2). The local scale consisted of individual stream sites including at least one riffle-pool sequence (150 m long), assessed by field surveys; the reach scale examined a 1500 m stream length, interpreted from aerial photos at 1:5000 scale; and the catchment scale examined both the riparian corridor and the entire drainage area upstream of a site, using a GIS data base at 1:24000 scale (Table 1).

Local riparian vegetation was assessed by field transect surveys at all sites between 2 June and 30 October 1992. Four transects were established on each bank at 50 m intervals along a 150 m stream reach, for a total of 8 transects. Beginning at the estimated level of bankful discharge on the stream bank and extending 30 m perpendicular to the stream, vegetation cover was classified into one of ten categories, with transitions between categories noted to the nearest 0.5 m. Categories included canopy forest, shrub, meadow, marsh, marsh with sparse trees, mixed herbaceous vegetation, herbaceous filter strip, agricultural cropland, mown grass, or other. No data were gathered beyond the 30 m distance.

Color aerial slides of field sites obtained from county Agricultural Stabilization and Conservation Service (ASCS) offices were used to analyze vegetation over a 1500 m stream reach. These photographs, taken annually throughout agricultural areas of Michigan by low-altitude flyovers, are used by ASCS staff to assess land cover for federal crop subsidy programs. Each slide covers one section (1 square mile or 2.59 km^2) of area. Flyover dates were between 11 June and 21 July 1992. Riparian vegetation was traced by projecting aerial photo images onto 7.5 minute topographic maps enlarged to a scale of 1:5000. Map features such as road crossings were aligned with slide images for positioning and to control the scale of projection. The maximum distortion between projected image and map features was estimated as 50 m over a length of 1 km ground distance, for approximately a 5% error. Riparian vegetation width was measured and recorded at 50 m ground distance intervals for 1000 m upstream and 500 m downstream of the study site origin, on each side of the stream, for a total of 62 measurements per stream reach. We employed the same ten vegetation categories as in field transects, but with difficulty because of the coarser scale. Grass filter strips could not be reliably distinguished from cropland, and delineation among pastures, dry mixed herbaceous areas, and marshes was difficult. However, forest and row crops were easily distinguishable, and areas of

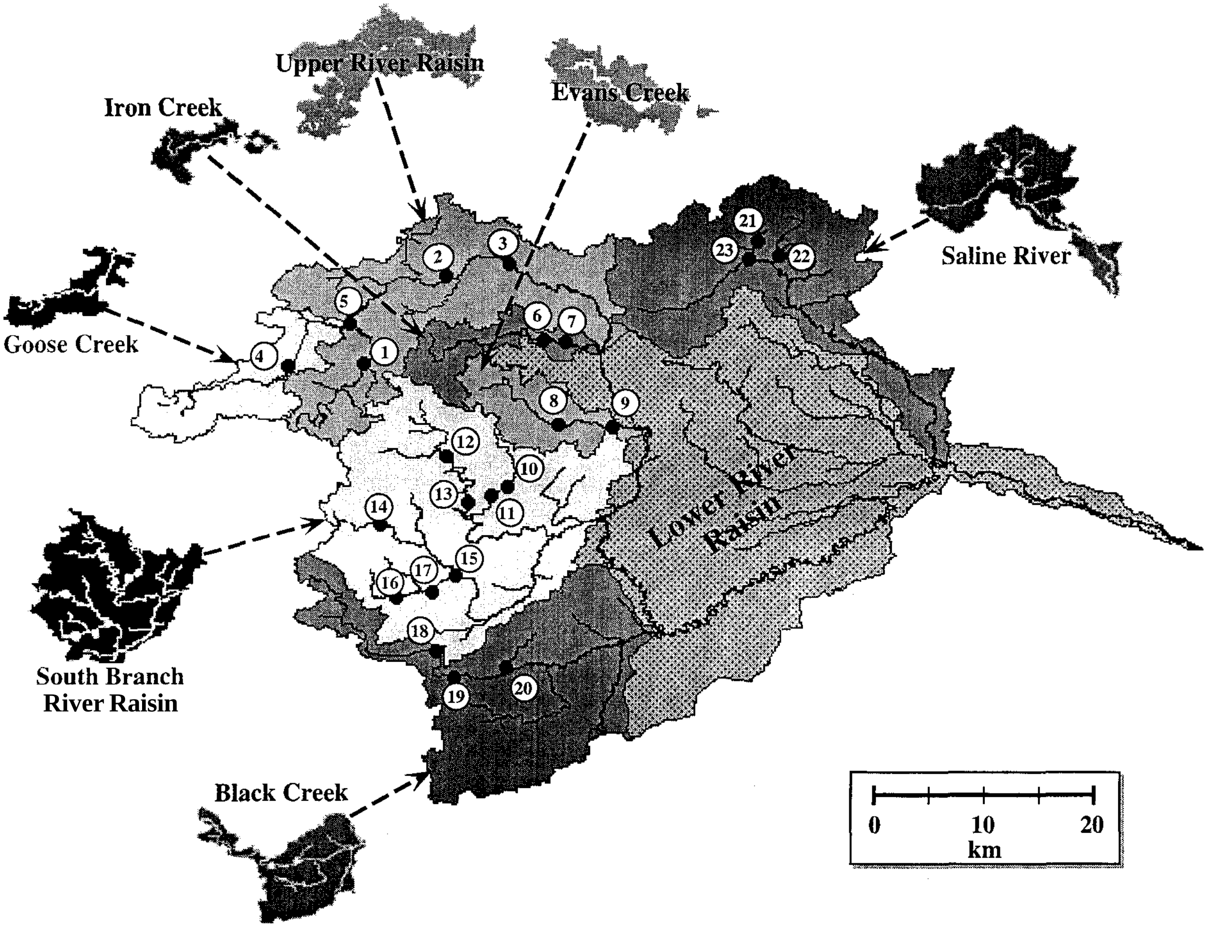

Fig. 1. Location **of** study sites in the glacial till terrain of the upper and middle River Raisin watershed. Note that sites are located within seven small catchments. Further detail is provided in Appendix.

mixed herbaceous vegetation and occasional woody plants could be distinguished from cropland/filter strips. Vegetation boundaries were recorded to the nearest 2.5 m ground distance. Transects which doubled back across or ran parallel to the stream due to river meanders were excluded. Summary riparian width values for each 1500 m reach included the mean and median. We also estimated "stream reach coverage", defined as the fraction of the 1500 m stream reach having a riparian land cover of some specified minimum width, using riparian zone widths of 15, 30, 50, and 125 m. These four riparian widths were used to evaluate the influence of the width designation on summary statistic values.

Several GIS analyses were performed to examine land use and vegetation cover for the stream corridor and catchment upstream of each site. Land use/cover data were taken from an existing digital database, the Michigan Resource Information **Sys**tem (MIRIS), derived from 1979–85 aerial photographs digitized at 1:24000. These data are considered sufficiently accurate to include landscape units of 1–2 ha or larger (MDNR 1990). Land classifications followed a hierarchical system of land cover/use categories, which can be aggregated into eight broad categories of urban, agricultural, nonforested, forested, surface water, wetland, barren, or other (Anderson ***et*** *al.* 1976). "Wetlands" included areas of wooded, shrub/scrub, and emergent vegetation, along with aquatic beds and flats. Land use/cover was calculated as area and percent cover

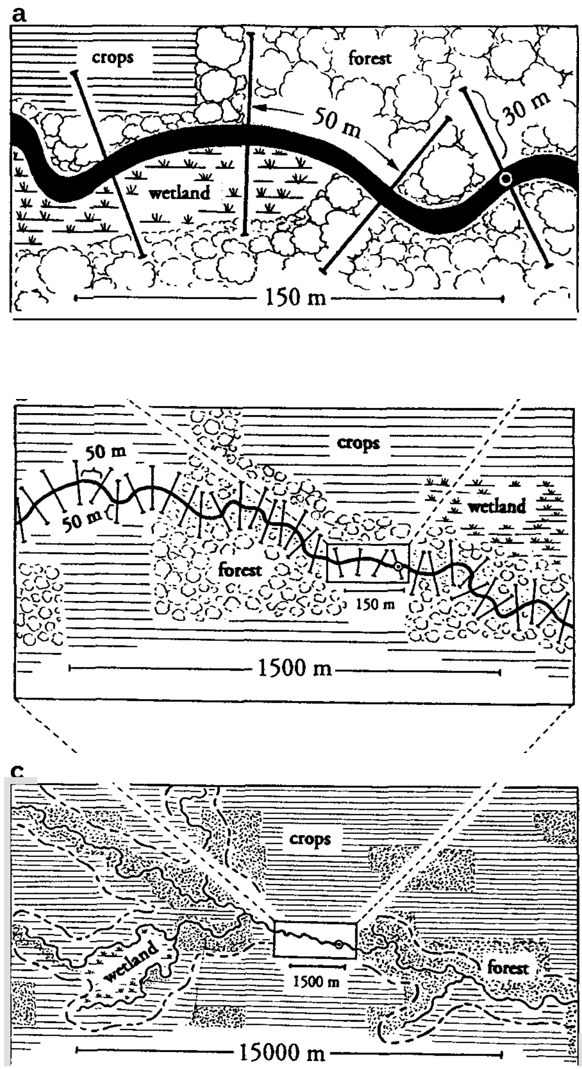

Fig. **2.** Analysis of riparian vegetation at multiple spatial scales. (a - site; b - reach, c - region). Vegetation was quantified (a) at the local scale by running 30 m transects laterally at 50 m intervals within a 150 m stream length, (b) at the reach scale using aerial photographs to measure vegetation at 50 m intervals over a 1500 m stream distance, and (c) using a **GIS** to quantify vegetation covers within riparian buffers and throughout the catchment upstream of a field sampling point.

for each of seven catchments within the River Raisin watershed and for the catchment of each of the 23 study sites.

The GIS programs C-MAP and ARCinfo were used to isolate riparian zones around each river segment and to overlay these river buffers with land uses. First, for an entire stream length, a "buffer" was created isolating an area of a specified width on each side of the river. This buffer was then overlain with a land-use layer, to crate a new data layer consisting of land use/cover within the riparian zone. Within this buffer zone, we determined the coverage of each major land-cover type, both as area and percent. Fifty meters was the smallest buffer width considered, given the resolution of the MIRIS data. This procedure was repeated using widths of 125 and 250 m, to construct zones of 100, 250, and 500 m total width (both sides of the stream). Because we typically sampled more than one site on a particular tributary of the Raisin watershed (Fig. 1), riparian buffers were constructed for a total of seven separate streams.

We used regression analyses to determine whether land-use variables were correlated across the spatial scales of measurement, and between categories of land use/cover. Clearly these are not independent variables: smaller spatial units are encompassed by larger spatial units, and an increase in one cover category necessarily requires a decrease in another cover category. Our use of regression here is intended to be descriptive, and correlation coefficients simply are indicators of relative strength of relationships.

2.3. *Index of Biotic Integrity*

Electrofishing surveys were conducted between 8 June and 16 July 1992, with two final sites surveyed on 5 and 7 September 1992. At nineteen sites fish were collected with an ABP-3 or Badger-1 backpack electroshocking unit. A boat-mounted electrofishing unit was used at four larger sites. Before electrofishing, two consecutive 50 m stream reaches were blocked with seine nets. A single electrofishing pass was made in an upstream direction within each section. We attempted to maintain consistent fishing effort at all sites, and to fish thoroughly any habitat structures encountered, such as pools, riffles, runs, fallen trees, and overhanging vegetation.

For each sampling site, an Index of Biotic Integrity (IBI) was calculated following the methods outlined by Karr *et al.* (1986), with some modifications (MDNR 1991; Roth 1994). A second-order site (site 1, Appendix 1) on the upper Raisin

Table 1. Three scales of riparian vegetation analysis employed at each of 23 locations in the River Raisin watershed.

Scale	Method	Assessed stream length	Assessed buffer width	Summary measures
site (1:1)	field transects	150 m	to 30 m	total vegetation width (m) woody vegetation width (m)
reach (1:5000)	aerial photographs	1500 m	to edge of vegetation	median vegetation width (m)
			15, 30, 50 or 125 m on each side of stream	stream reach coverage (%) for each buffer width
regional (1:24000)	GIS	entire length upstream of site (> 2 km)	50, 125 or 250 m on each side of stream	% area within each riparian buffer under a specified land use

Table 2. Stream habitat variables incorporated in the Habitat Index (HI). Assigned scores are given in parentheses. See MDNR (1991) and Roth (1994) for additional detail.

Category	Description	Range of scores
1. stable substrate/ suitable cover	coarse gravels, submerged wood, undercut banks and other stable habitats	0 – 20
2. embeddedness	area of substrate covered with fine substrates	0 – 20
3. velocity:depth variability	range of v:d conditions defined by $v >$ or $< 0.5\ m^3/s$; $d >$ or < 0.3 m	0 – 20
4. flow stability	natural and continuous flows vs. flashy or ephemeral, contribution of point discharges	0 – 15
5. bottom deposition	extent of siltation and bottom deposition filling of pools and around obstructions	0 – 15
6. pool-riffles-runs-bends	habitat variety, deep pools and riffles vs. channelized streams with uniform habitat	0 – 15
7. bank stability	stable, minimal erosion or bank failure vs. steep, eroded banks with potential for slumping	0 – 10
8. bank vegetative stability	extent of streambank surface covered with vegetation or rubble, protection from erosion	0 – 10
9. streamside cover	fraction of streambank/nearbank area vegetated, shrubs and trees vs. grass or unvegetated	0 – 10

was selected as a reference site based on the diversity of its fish assemblage and its high degree of habitat heterogeneity. Fish species were classified into throphic guilds and three categories of silt tolerance (see Roth 1994) based on literature descriptions (Scott and Crossman 1973; Trautman 1981; Becker 1983). Classifications were generally consistent with previous classification efforts (Detenbeck *et al.* 1992; Ohio EPA 1989; Whittier *et al.* 1987). For trophic groupings, herbivore-detritivores were defined as fishes whose diet consists of greater than 25% plant material while omnivores were those with a diet of 0–25% plant material (Schlosser 1982). Fishes that are piscivorous as adults but insectivorous as juveniles were classified as piscivores, regardless of the size of specimens captured. Species preying primarily upon large invertebrates (*e.g.*, crayfish) were also included in the piscivore category, to denote their status as high-level carnivores.

2.4. *Habitat assessment*

Instream habitat was assessed by field observation using a protocol developed by the Michigan DNR

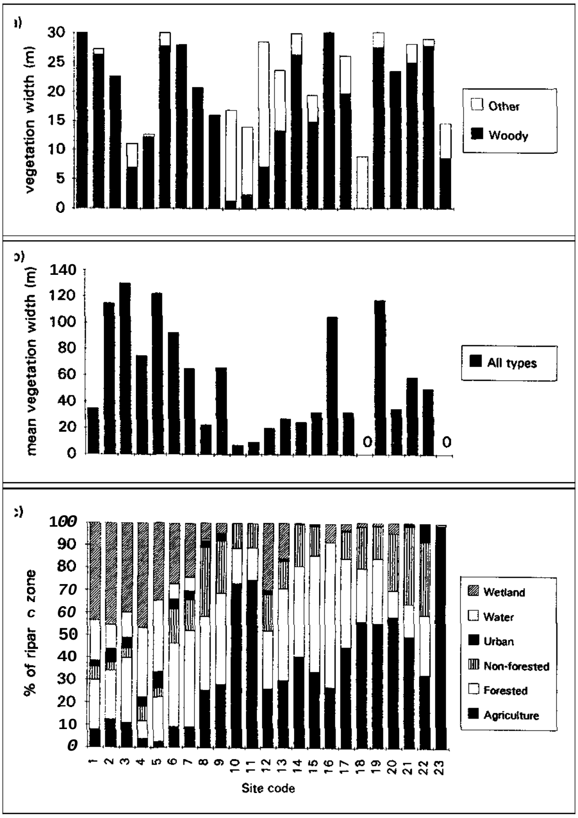

Fig. 3. Examples of results from riparian vegetation analyses at three spatial scales. (a) Field transect data showed a range of vegetated cover, to a maximum measured width of 30 m. "Woody vegetation" includes canopy forest and shrubs. "Other natural vegetation" includes mixed herbaceous cover and planted filter strips, but does not include lawns, pasture, or crops. **(b)** Median width of vegetation, measured from 1:5000 scale aerial photographs along a 1500 m stream reach. Vegetation (woody and herbaceous) widths were measured from the stream to the outer edge of natural vegetation adjacent to the stream. **(c)** Land use/cover as a percentage of area within a 50 m riparian buffer upstream of each site, determined by GIS.

(MDNR 1991). Each of nine factors related to habitat condition was first classified and then assigned a score according to a standard verbal description (Table 2). Scores for individual factors were summed to give a habitat index (HI) on a scale of 0 to 135. In order to reduce observer bias, assessments at most sites were made by the same two individuals, after visiting multiple sites to observe the range of available conditions. A 200+ m stream reach was examined to observe the complete range of habitats available.

3. Results

3.1. Land use/cover

*1. **Local scale.*** Field surveys at the 23 sites revealed that local riparian conditions ranged from little or no vegetation to complete forest cover (Figure 3a). The mean width of local woody vegetation (canopy forest and shrub) ranged from 0 to 30 m, the maximum possible value by this method. Width of combined woody (canopy forest and shrub) and herbaceous (meadow, marsh, mixed, and filter strip) vegetation ranged from 8.8 to 30 m. Over the 23 sites, woody and total vegetation widths corresponded closely ($r = 0.823$, $p < 0.01$). A few sites with very narrow strips of woody vegetation in the Black Creek and South Branch catchments had moderate coverage (8.8 to 16.9 m) when both vegetation types were included.

2. ***Reach scale.*** Median vegetation width along 1500 m stream reaches ranged from 0 to 130 m across the 23 locations, based on aerial photo measurements (Fig. 3b). Stream reach coverage based on a 30 m wide buffer varied from 0 to 100%. Stream reach coverage was high, > 60% at 18 of 23 sites, when a buffer width of 15 m was selected, but declined as buffer width was increased (Fig. 4). When a 125 m buffer designation was employed, no sites had this extent of vegetation coverage, and a majority of sites had <20% coverage. Using a 30 or 50 m buffer resulted in a more even distribution of estimates of percent coverage across the 23 sites, and so probably is most useful for distinguishing among sites.

Ranking of sites according to extent of stream reach coverage were not greatly affected by the choice of riparian width designation. Analyses using Spearman's rank correlation (r,) and Kendall's Tau showed that sites were ranked in a similar manner, regardless of the buffer width selected (Table 3). As expected, the greatest discrepancies occurred in comparing rankings between the narrowest (15 m) and widest (125 m) stream reach coverage, but even these values were significantly correlated ($r_s = 0.752$, $p < 0.001$; Tau $= 0.567$, $p < 0.005$; $n = 23$). Median riparian vegetation width yielded rankings similar to those using stream reach coverage ($r_s = 0.845$ to 0.981, Tau $=$ 0.691 to 0.906). Rankings of median width of

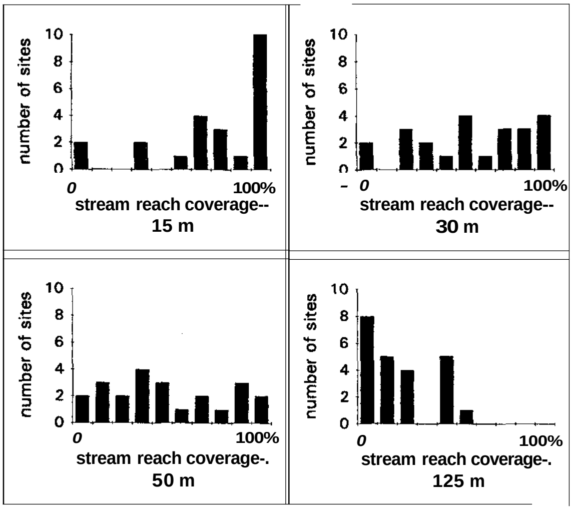

Fig. 4. The influence of designated buffer width on strea reach coverage estimated from aerial photographs. "Stream reach coverage" refers to the percent of a 1500 m stream reach that is vegetated with woody and herbaceous plants within a buffer of designated width of 15, 30, 50 or 125 m (each bank). Note that most sites have high vegetation coverage based on 15-m buffers, and most sites have low coverage based on 125-m wide buffers. Maximal site separation was achieved using 30-m and 50-m buffers.

riparian vegetation were most highly correlated with estimates of stream reach coverage when a 30 m buffer was employed in calculating stream reach coverage.

3a. Regional Scale – Riparian. Land use/cover in 50-m riparian buffers upstream of each site varied greatly among sites and also among the seven catchments (Fig. 3c). Less than 11% of the riparian zone was agriculture in the Upper Raisin, Goose Creek, and Iron Creek catchments. At the other extreme, 51.3% of the 50-m wide riparian zone of Black Creek was in agricultural use. The extent of riparian wetlands also differed greatly among the seven tributaries examined. The Upper Raisin had the greatest portion of the 50-m riparian zone as wetland (36.5%), followed by Goose Creek (34.7%) and Iron Creek (19.8%). The remaining catchments had only 0.6 to 3.7% of the riparian zone as wetland.

Other land-use categories also varied among the seven tributaries, although to a lesser extent than did agriculture and wetlands. Forested land comprised 19.9% of the 50-m riparian zone for Goose Creek, 32.0% for the Upper Raisin, and 39.5 to 46.7% for the remaining subbasins. Urban land use made up 0.2% (Black Creek) to 11.9% (Evans Creek) of riparian zone lands.

For all seven catchments, the percent of land in agricultural use increased as the riparian width was expanded from 50 to 125 to 250 m. These increases were paralleled by corresponding decreases in proportion of forested land and wetlands at greater distances from the stream edge (Roth 1994). However, altering the designation of riparian width did not greatly change the relative comparisons of riparian land uses across the seven catchments.

3b. Regional scale – Catchment. GIS analysis of land use/cover within the catchment upstream of each study site revealed a wide variation in use/cover at this broadest scale of analysis (Roth 1994). The extent of agriculture within seven stream catchments ranged from 35.7% (Iron Creek) to 84.2% (Black Creek). Forest cover varied less dramatically, from 9.6% (Black Creek) to 25.0% forest (Iron Creek). Urban lands made up only 1.4–13.1% of catchment land use. Goose Creek (9.2%), Upper Raisin (8.7%) and Iron Creek (5.6%) had the greatest representation of wetlands. All other areas had minimal wetland coverage (0.5–2.2%). Across all catchments and sites, land use/cover at the catchment level showed some correspondence to land use/cover within 50-m riparian buffers.

***4.** Comparison of methods and scales* To determine the similarity of riparian measures obtained at local, reach, and catchment scales using three different methods, correlations between field, aerial photograph, and GIS estimates of riparian land use/cover were examined. Reach-scale (aerial photo) measures of median vegetation width corresponded more closely with field estimates of widths of woody vegetation alone ($r = 0.521$, $p = 0.011$) than with field-assessed widths that included filter strips and other herbaceous vegetation ($r = 0.275$, $p = 0.204$). The stronger correlation for woody vegetation is to be expected, because aerial photo interpretation adequately detected woody vegetation, but less clearly distinguished herbaceous cover from agricultural land use. Some discrepancy between site- and reach-scale results is expected due to real differences in vegetation between the 150 m stream length assessed by field

Table 3. Rank correlations between stream reach coverage for four different buffer widths, listed as Spearman's r. In parentheses are values for Kendall's Tau (**n** = 23). Stream reach coverage is the percent of a 1500 m stream reach having a vegetated buffer of a designated minimum width (15, 30, 50 or 125 m), as determined from aerial photograph interpretation.

	15 m buffer	30 m buffer	50 m buffer	125 m buffer
15 m buffer	1			
30 m buffer	0.946 (0.822)	1		
50 m buffer	0.880 (0.727)	0.972 (0.884)	1	
125 m buffer	0.752 (0.567)	0.855 (0.701)	0.910 (0.749)	1

survey and the 1500 m reach assessed by photographs. When median widths of woody and shrub vegetation in aerial photos were recalculated to consider only the 150 m length of stream that was surveyed, correlations between reach and site data increased (wooded vegetation alone: $r = 0.618$, $p = 0.002$; total vegetation width: $r = 0.381$, $p = 0.073$).

Regional (GIS) and local-scale (field) measures of riparian conditions corresponded poorly. Local vegetation widths measured in field surveys showed a weak negative correlation with the percent of the upstream riparian zone as agriculture (total vegetation width: $r = -0.270$, $p = 0.213$; wooded vegetation alone: $r = -0.429$, $p = 0.041$). Field-estimated vegetation widths showed a weak positive correlation with GIS estimates of percent forest (total vegetation width: $r = 0.429$, $p = 0.041$; wooded vegetation alone: $r = 0.431$, $p = 0.040$).

The correspondence between regional and reach estimates was fairly high, perhaps because measures over a 1500 m reach were more reflective of broader-scale land-use patterns than were the local (field survey) measures. The strongest relationship obtained between GIS and aerial photograph measures was the negative correlation between percent agriculture and median vegetation width ($r = -0.603$, $p = 0.002$).

3.2. *Index of Biotic Integrity*

A total of 43 fish species were collected at the 23 River Raisin sites. The number of species per site ranged from 5 to 21, and the number of individuals per site from 45 to 532. Sites varied extensively in species composition. Estimates of the IBI ranged from a low score of 22 to a high of 46, out of a maximum possible score of 50 points (median = 34, mean = 33.7, s.d. = 7.2). Highest values were clustered in the upper tributary catchments. The relative contributions of each metric to the IBI were evaluated by correlation analysis and regression of total IBIs on combinations of the individual metric scores. The number of intolerant species had the highest individual rank correlation with the total IBI (Spearman's rank, $r_s = 0.862$, $p < 0.001$), followed by the total number of species ($r_s = 0.693$, $p < 0.001$). **A** "two-metric IBI" containing only these scores was highly correlated with IBI ($r^2 = 0.923$, $p < 0.001$).

Regressions of land use and riparian variables against the IBI showed that land uses at larger spatial scales, whether the entire catchment upstream of a site or the entire riparian corridor upstream of a site, were the most effective predictors of site to site variation in IBI scores. Measures of local riparian vegetation at site and reach scales typically were ineffective predictors. Regression of IBI estimates against the percent of upstream catchment area as agriculture explained 49.6% of the variation in IBI scores ($p < 0.001$, Fig. 5). While other catchments land uses, including the percent of land use as forest ($r^2 = 0.479$, $p < 0.001$) and percent as wetland ($r^2 = 0.397$, $p < 0.001$), explained a high degree of IBI variation, these variables were themselves highly correlated with the percent of upstream catchment area as agriculture ($r^2 = 0.762$, $p < 0.001$; $r^2 = 0.543$, $p < 0.001$ respectively). To address the loss of some independence inherent in catchment-scale land use data for sites along a single tributary stream, further analysis of catchment data was conducted. In this analysis, only land uses in catchment immediately above a site but below the next upstream site were considered. Relationships between IBI and catchment data were not substantially altered by this further analysis.

Riparian land uses along the entire length of a stream above a sampling site, as determined

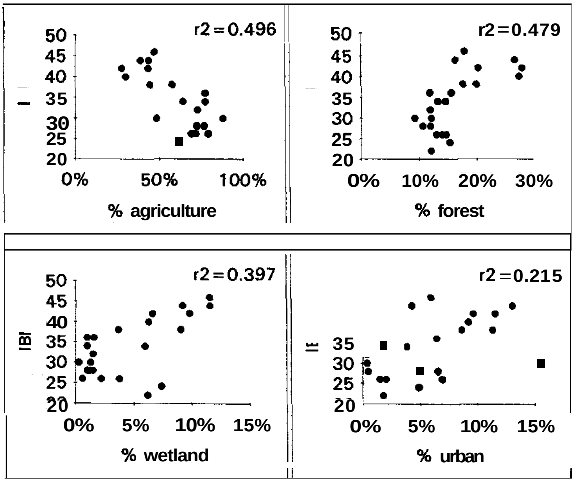

Fig. **5.** Relationships between IBI scores and four categories of land use throughout the catchment area upstream of each site. The IBI was inversely correlated with percent agriculture ($p < 0.001$), and positively correlated with percent forest ($p < 0.001$), wetland ($p = 0.001$) and urban ($p = 0.026$).

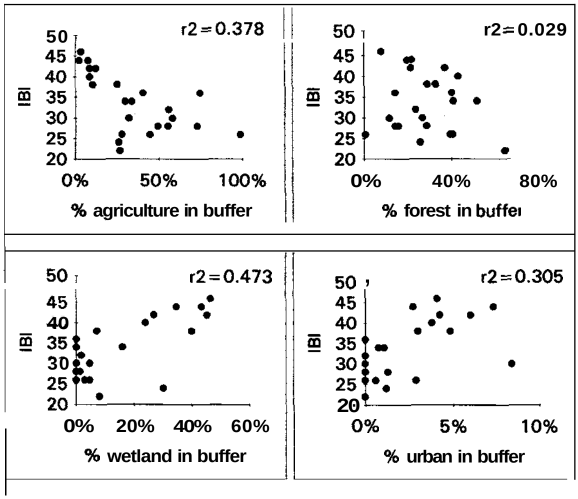

Fig. 6. Relationships between IBI scores and four categories of riparian land use, estimated from GIS buffer analysis. The IBI was inversely correlated with the extent of agriculture within 50-m riparian buffers ($p = 0.002$) and positively correlated with the extent of wetlands ($p < 0.001$).

through GIS analysis, also were highly correlated with the IBI (Fig. 6). Land uses were expressed as percent cover within a 50-m wide riparian buffer zone upstream of sites. In particular, the percent of the riparian zone as agriculture explained 37.8% of the variance in IBI ($p = 0.002$), while the percent as wetland explained 47.3% ($p < 0.001$). Interestingly, the extent of forested land (not counting wooded and shrub/scrub wetland) within 50-m riparian buffers was not an effective predictor of IBI values.

Riparian land use measured at smaller spatial scales was unable to explain significant amounts of site-to-site variation in IBI values. Median vegetation width along a 1500 m stream reach, assessed from aerial photographs, accounted for 8.8% of variation in the IBI ($p = 0.17$), and stream reach coverage using a 30-m buffer accounted for only 7.3% of IBI variation ($p = 0.21$). Field measures of local riparian vegetation proved even less successful. Total vegetation width and woody vegetation width from 30-m transects measured at sites explained only 1–2% of variation in IBI estimates (Roth 1994).

Because various measures of vegetation cover were to some extent correlated with one another, we wished to determine whether a model with two or more land cover variables would be a significant improvement. Because the extent of site-specific catchment area as agriculture provided the best single regression against the IBI, we evaluated the residuals of this regression against remaining landscape variables using scatterplots and regression. Local vegetation width explained the greatest amount of remaining variation, accounting for 13.0% of residual scatter ($p = 0.090$). No other factor explained more than 6% of the residual variation.

Two measures of fish community integrity were considered as alternatives to the IBI for comparison with landscape variables. These alternatives were the number of species **(S)** and proportional representation by sensitive (intolerant) species (Fausch ***et al.*** 1990). In comparisons with land use and riparian factors, these measures showed trends similar to the IBI. Species richness **(S)** was higher at sites whose upstream catchments were predominantly wetlands, and lower at sites with a greater extent of agricultural lands (Roth 1994). The number of species was also correlated with land uses within upstream riparian zones. However, in general, the values of correlation coefficients were lower in models with S than with the IBI. **As** with the IBI, local riparian vegetation widths were not related to **S.**

Comparisons of landscape variables with the representation of sensitive species yielded similar results. The percent of individuals as intolerant fishes was higher at sites with greater percent of wetlands, in either the upstream catchment or within the riparian zone. Scores were depressed in areas with extensive agriculture, at either the catchment or riparian level. The percent intolerants was low at sites where more than 35% of the riparian zone was in agriculture, indicating that these species may be vulnerable to the removal of riparian vegetation at the catchment level.

3.3. Habitat assessment

Habitat index scores for the 23 sites ranged from 11 to 111 (median = 48, mean = 55.0, s.d. = 31.2), of a maximum possible score of 135. In general, as was seen for the IBI, highest scores were obtained in the northwestern portion of the Raisin watershed, at sites within Goose Creek, the Upper Raisin, and Iron Creek catchments. Lowest scores were reported at sites within the Saline River and Black Creek catchments (Appendix).

Habitat index values were good predictors of IBI scores ($r^2 = 0.334$, $p = 0.004$, Fig. 7). We also found that intolerant fish species were scarce at sites that had HI values below 60, and that they made up an increasing fraction of the assemblage with increasing values of HI above this threshold (Roth 1994). Moreover, landscape variables were effective predictors of HI scores, and as with the IBI, relationships were strongest when land cover was measured at the largest spatial scales (Fig. 8).

Among single parameters considered in regression analyses, the percent of catchment above a site as agriculture explained by far the greatest amount of variability in habitat scores ($r^2 = 0.758$, $p < 0.001$). The percent agriculture within the 50-m riparian zone upstream of the site also was a significant predictor of HI scores ($r^2 = 0.533$, $p < 0.001$), as was the percent of riparian wetlands ($r^2 = 0.454$, $p < 0.001$). Many of the other catchment or riparian land use factors were highly correlated with habitat scores, but also were collinear with the percent of the catchment as agriculture.

Riparian measures from aerial photographs did not correlate strongly with HI scores. For example, the stream reach coverage along 1500 m of stream (percent of reach having at least a 30 m vegetated buffer) was borderline significant ($p = 0.059$), explaining only 16.0% of the variability in HI. Habitat index scores were not significantly correlated with field-assessed riparian widths of total vegetation ($r^2 = 0.115$, $p = 0.11$). The relationship between HI and local woody vegetation was borderline significant ($r^2 = 0.164$, $p = 0.06$) although of low predictive power. However, plots of HI against each of these local measures suggest there may be a threshold of vegetation needed to maintain high habitat scores. Among sites with an average vegetation width at or near 30 m, habitat index values varied widely. In contrast, high HI scores were rarely attained at sites with vegetation widths less than 15 m (Fig. 8c).

In analysis of residuals, the single factor explaining the greatest amount of remaining variation beyond that explained by a regression of HI on percent agriculture was local vegetation width, which accounted for 13.6% of the residual variation ($p = 0.08$). No other factor explained a significant portion of the residual variation, although the width of local woody vegetation explained the next highest proportion of variation ($r^2 = 0.093$, $p = 0.16$).

4. Discussion

*4.1. **Ecological** assessment of the River Raisin watershed*

The range of estimates for the Index of Biotic Integrity for fish assemblages of the River Raisin indicates that sites vary from poor to very good. Measures of instream habitat and of land use/cover reveal a similarly wide range of conditions. This is consistent with other evidence that the Raisin is neither extremely degraded nor pristine, but exists at some intermediate state of anthropogenic disturbance. Tributaries in the northwest region of the watershed, where the extent of wetlands and forest fragments is greatest and agriculture is less dominant, generally show highest values of the IBI and HI.

The biological integrity of the fish assemblage, as quantified by the IBI, was directly correlated with habitat quality (Fig. 7), consistent with a large literature relating stream fish assemblages to instream habitat conditions (Matthews 1987;

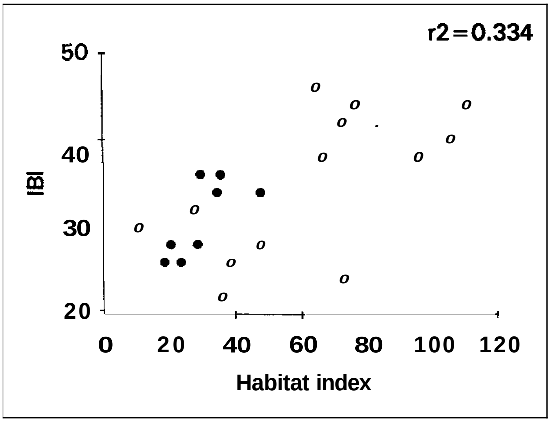

Fig. **7.** For the 23 River Raisin sites, the IBI was positively correlated with Habitat Index scores ($p = 0.004$). The IBI is a composite of ten metrics obtained from fish collections; the Habitat Index is a composite of nine metrics of local stream and bank habitat.

Schlosser 1982). Habitat characteristics assessed by the HI included the variety of velocities and depths within a reach, riffle and pool structure, substrate embeddedness, cover, and bank stability (Table 2). The HI did not explain a significant amount of IBI variation beyond that explained by the extent of catchment land use as agriculture, probably because habitat index scores were themselves highly correlated with land use factors. Catchment land uses and riparian vegetation likely play a strong role in structuring habitat features, which in turn influence the composition of the fish community.

There is ample reason to believe that habitat quality has declined in the River Raisin system due to human alterations to the landscape, and that fish species have experienced a corresponding decline. Similar patterns of habitat degradation and corresponding declines of fish species have been noted in a number of Midwestern watersheds (Trautman 1981; Karr *et al.* 1985). Landscape modification within catchments, particularly affecting wetland drainage and stream channelization, have contributed to the destruction of spawning habitat in Ohio's Maumee River, a basin adjoining the River Raisin, where a large percentage of headwater species have declined in the past century (Karr *et al.* 1985). In the Huron River of southeastern Michigan, Yant and Humphries (1980) noted shifts in fish species composition between 1938 and

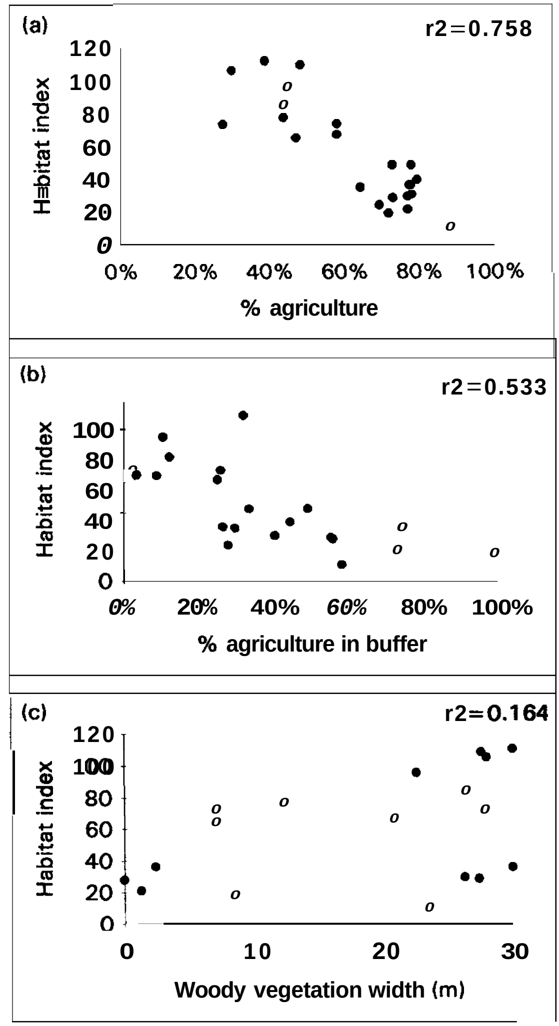

Fig. 8. Scores of the Habitat Index compared with landscape variables. The Habitat Index showed a strong negative correlation with (a) the percent of the catchment above a site as agriculture ($p < 0.001$) and (b) the percent of the upstream riparian buffer as agriculture ($p < 0.001$), but showed no significant linear relationship with (c) local vegetation extent, as measured in field surveys.

1977, including a decline in the number of silt-intolerant darter species and a concurrent increase in the abundance of silt-tolerant species. Examining fish data from 1940 and 1992, Gatz and Harig (1993) reported a decline in IBI in an Ohio stream of the Eastern Cornbelt Plains and cited habitat degradation as the likely cause.

In a previous study of the River Raisin, Smith *et al.* (1981) noted strong relationships between habitat characteristics and fish distributions. Species requiring particular critical habitats were restricted to sites in the Raisin watershed where human impacts were less evident. Silt- and pollution-tolerant species made up a greater portion of species assemblages in impacted areas. Siltation from agriculture was suspected as a dominant factor in determining the distribution of fishes with particular habitat needs, because siltation over clean rock and gravel substrates may affect spawning or feeding behaviors. Smith *et al.* (1981) observed that these impacts were diminished in areas having vegetative cover.

4.2. Land use and stream biotic integrity

The need for a hierarchical view of stream ecosystems is evident from inspection of their physical features and also from an analysis of human impacts. Large watersheds are comprised of tributaries and their catchments, tributaries contain multiple stream reaches, each reach potentially includes riffle, pool and other habitat units, and these habitat units each contain multiple microhabitats (Frissell *et al.* 1986; Sedell *et al.* 1990). We found that measures of land use and riparian vegetation at larger spatial scales were superior predictors of stream ecological integrity than were more local measures. Sites whose upstream catchments were dominated by agriculture ranked lowest by both the IBI and HI, whereas sites whose land areas contained higher percentage of naturally vegetated land, particularly wetlands, tended to rank higher. The IBI and the HI behaved remarkably similarly in this regard. Variation in both metrics within the watershed could be predicted well (50% of IBI variation, 76% for HI) from GIS estimates of land use within catchment areas upstream of field sites. Predictive power was less when land use in only the 50-m riparian buffer was used, and fell to marginal of non-significant levels for reach and local vegetation measures (*e.g.*, Fig. 8). While it might be argued that this pattern is indicative only of measurement accuracy, we suspect that the important controllers of stream ecological integrity are indeed operating on the larger spatial scale.

The poor sensitivity of IBI and HI scores to local and reach-scale riparian measures was surprising. However, our results provide two indications that local riparian vegetation may make some contribution to stream integrity. First, multiple regression models for IBI and HI scores each identified local vegetation as the second variable, after catchment area in agriculture, although in each instance the additional variation explained was small and not significant at the 5% level. Second, inspection of Fig. 8c suggests that sites lacking adequate riparian coverage at the local level tended to have low habitat scores, suggesting there might be a threshold of local vegetation needed to maintain high-quality instream habitat. These local patches of high-quality riparian vegetation may be necessary to prevent isolated stream reaches from suffering extreme degradation, but may not be sufficient to maintain highest quality habitat. Thus, the idea that riparian vegetation will support “oases” of high-quality stream habitat (Marsh and Luey 1982) needs to be re-evaluated in the context of the entire river ecosystem. Upstream processes may overwhelm the ability of local, isolated patches of riparian vegetation to support stable instream habitat.

4.3. Methods of riparian assessment

The choice of scale for assessing riparian vegetation involves tradeoffs between resolution and scope of data gathered. Of our three methods, the field surveys offered the most accurate, detailed, and current evaluation of riparian conditions in close proximity to stream study sites. Depending on the type of information required by the investigator, the transect method could be modified by incorporating assessments of riparian slope, canopy height, woody plant diversity, stem density, and other characterizations of the riparian community, as needed. However, the labor-intensive nature of field assessments suggests that only small sections of stream are likely to be evaluated using such a detailed protocol.

Aerial photograph interpretation is a simple, efficient method for quantifying local riparian conditions. Photographic images provide a “bigger picture” of local conditions than is visible from the field, and can provide a reasonable local assessment when field surveys are not possible. Where available, photographs from different time periods could be used to assess temporal changes in vegetation patterns. The disadvantages of using aerial photographs include the measurement errors described previously and some difficulty in differentiating vegetation types. Familiarity with typical riparian vegetation from ground surveys proved helpful in recognizing vegetation types in the aerial photographs.

Use of a GIS for the entire watershed made possible an evaluation of riparian land cover over entire stream lengths and within catchments upstream of sites. Although the accuracy of information did not approach that of field surveys, a much larger geographic area, representing "regional" conditions, could be assessed. Use of a previous digitized database allowed for a relatively easy analysis, but interpretations were limited to the accuracy of the existing and possibly outdated information. For the scope of this study, data based upon 1:24000 maps provided a satisfactory level of detail. While it is often difficult to obtain digital information for particular time periods, GIS potentially offers the ability to measure past changes in vegetation patterns and to model future scenarios.

Each scale of analysis allowed for the assessment of a different portion of the riparian ecosystem, from the local to regional level. Nonetheless, assessment of riparian conditions corresponded reasonably well between local and reach scales, and between reach and regional scales. One might expect to obtain the weakest correlation when comparing the smallest and largest scales of measurement, as we found. However, sites were intended to be representative of the stream where they were located, and in our judgement were indeed representative. In this light the correspondence between field surveys and GIS estimates was weaker than might be anticipated. The lack of strong correspondence between field and GIS measures indicated that local riparian condition was not wholly determined by landscape-scale patterns.

Assessments of riparian conditions in both aerial photograph and GIS analyses were not highly sensitive to the choice of riparian zone width. Within each method and scale, results using different buffer zone widths correlated well. However, the choice of buffer width strongly affected the ability of each analysis to distinguish among sites. It may often be useful to explore the influence of buffer width on site separation, within the constraints of the accuracy of the land-use/cover database. Selection of too wide or too narrow a buffer is likely to result in poor ability to discriminate among sites.

Rather than a single "best" scale or method for the measurement of riparian vegetation, there are a number of different options depending on the goals of the study and what underlying processes are deemed important. However, based on our measures of fish assemblages, habitat quality, and land use at multiple spatial scales, it appears that stream biotic integrity is more strongly influenced by landscape than by local land uses. Although the maintenance of vegetated riparian buffer zones can be expected to convey numerous advantages to stream ecosystems, our results cast doubt on the effectiveness of localized efforts. Our findings are consistent with the view that degradation of instream habitat is a principal cause of reduced biotic integrity as assessed by the IBI, and this habitat degradation likely results from altered flow regime, increased sediment inputs and decreased organic inputs over considerable distances upstream of a site. We believe it is an open question whether protection of stream margins alone, especially within localized stream sections, is a sufficient measure to offset human-induced changes to entire catchments.

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Appendix

Specific locations, drainage area and stream order of 23 sites in the River Raisin watershed. Sites are located on streams within one of the seven catchments listed (see also Fig. 1). Drainage areas were calculated using site-specific boundaries digitized into a GIS. Stream orders (Strahler 1957) were determined using 1:24000 topographic maps.

Site code	Stream name	Catchment	Drainage area (ha)	Stream order	IBI score	HI score
1	River Raisin	Upper Raisin	4887.0	2	44	111
2	River Raisin	Upper Raisin	19156.0	3	42	85
3	River Raisin	Upper Raisin	25087.0	3	38	96
4	Goose Creek	Goose	6891.6	1	46	65
5	Goose Creek	Goose	10422.5	2	44	77
6	Iron Creek	Iron	5493.6	1	42	72
7	Iron Creek	Iron	5830.2	1	40	106
8	Evans Creek	Evans	4088.1	1	38	67
9	Evans Creek	Evans	7820.0	1	26	24
10	Black Creek	South Branch	3027.3	1	28	21
11	Black Creek	South Branch	3096.7	1	36	36
12	Wolf Creek	South Branch	7787.0	2	24	73
13	Wolf Creek	South Branch	10209.4	2	34	35
14	Hazen Creek	South Branch	1388.5	1	36	30
15	Hazen Creek	South Branch	6228.4	2	34	48
16	South Branch R. Raisin	South Branch	2744.5	1	22	36
17	South Branch R. Raisin	South Branch	5738.4	2	26	39
18	Black Creek	Black	5032.3	1	32	28
19	Black Creek	Black	6324.1	1	28	29
20	Fairfield Drain	Black	2064.7	1	30	11
21	Unnamed drain	Saline	3583.2	2	26	48
22	Wood Outlet Drain	Saline	3799.9	2	30	109
23	Saline River	Saline	5733.0	1	26	19