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Research article

Influences of upland and riparian land use patterns on stream biotic integrity

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Abstract

We explored land use, fish assemblage structure, and stream habitat associations in 20 catchments in Opequon Creek watershed, West Virginia. The purpose was to determine the relative importance of urban and agriculture land use on stream biotic integrity, and to evaluate the spatial scale (i.e., whole-catchment vs riparian buffer) at which land use effects were most pronounced. We found that index of biological integrity (IBI) scores were strongly associated with extent of urban land use in individual catchments. Sites that received ratings of poor or very poor based on IBI scores had > 7% of urban land use in their respective catchments. Habitat correlations suggested that urban land use disrupted flow regime, reduced water quality, and altered stream channels. In contrast, we found no meaningful relationship between agricultural land use and IBI at either whole-catchment or riparian scales despite strong correlations between percent agriculture and several important stream habitat measures, including nitrate concentrations, proportion of fine sediments in riffles, and the abundance of fish cover. We also found that variation in gradient (channel slope) influenced responses of fish assemblages to land use. Urban land use was more disruptive to biological integrity in catchments with steeper channel slopes. Based on comparisons of our results in the topographically diverse Opequon Creek watershed with results from watersheds in flatter terrains, we hypothesize that the potential for riparian forests to mitigate effects of deleterious land uses in upland portions of the watershed is inversely related to gradient.

Introduction

The structure and function of stream ecosystems are inextricably linked to the status and condition of their surrounding watershed. The amount and source of primary production in streams are regulated by the amount of shading and quantity of leaf litter entering the stream from the surrounding forest (Wallace et al. 1999). Near-stream vegetation acts synergistically with geology and topography to influence channel form (Gregory 1992), instream habitat (Bisson et al. 1987), nutrient dynamics (Cummins 1992), and temperature and flow patterns (Risser 1990). As a result, the diversity and productivity of stream communities are strongly tied to the condition of the landscape (e.g., Hynes 1975; Vannote et al. 1980; Schlosser 1991), and maintaining some level of protection to streamside vegetation is believed to be integral to preserving the biological integrity of stream ecosystems (Gregory et al. 1991; Sweeney 1992; Naiman et al. 1993).

Since the enactment of the Clean Water Act in 1977, there have been significant improvements in water quality and stream health to many of America's river systems due largely to mitigation of the acute effects of point-source water pollution (Browne 1981; Osborne and Wiley 1988). However, in spite of these improvements, streams and rivers throughout America continue to degrade at alarming levels (United States Environmental Protection Agency 2000). Continued degradation of stream ecosystems is due mostly to nonpoint-source pollution and the cumulative impacts of changing land use on stream habitats and biological communities. Removal of upland and riparian vegetation through farming and urbanization disrupts land-water linkages leading to reductions in water quality (Perterjohn and Correll 1984; Osborne and Wiley 1988; Zampella 1994), simplification of stream channels due to siltation (Judy et al. 1984; Rabeni and Smale 1995), less stable thermal and flow regimes (Leopold 1968; Barton et al. 1985; Imhof et al. 1991), and ultimately, reduced biological integrity (Richards et al. 1996; Roth et al. 1996; Wang et al. 1997).

Numerous natural processes operating at multiple scales interact to control the form and development of watersheds and streams and ultimately the biological communities they support (e.g., Richards et al. 1996; Roth et al. 1996; Wang et al. 1997). Regional differences in climate, lithology, and natural disturbance regimes influence the transport of water, sediments, and wood, which ultimately determine stream habitat. These factors, along with the biogeography and migration potential of native populations, determine the structure of local fish assemblages. Even within a region, site-specific variation in catchment size, topography, and the spatial position of stream sites in the watershed may be expected to play a significant role in stream ecosystem structure. However, the relative importance of each of these processes in controlling ecosystem structure differs among locations, and the processes themselves may be sensitive to landscape alterations (McDonnell and Pickett 1990). As a result, there is substantial variation among regions and stream types in the extent to which land use changes result in significant reductions in ecosystem integrity, and in which physical and biological components of stream ecosystems are most sensitive to changing land use. Moreover, the scale at which land use influences are most pronounced varies as well. For example, in studies where scale influences were tested, whole-catchment land use patterns were found to be better predictors of stream biological integrity in some studies (e.g., Frissel et al. 1986; Poff and Ward 1990; Naiman et al. 1992), while others suggest riparian land use patterns were more influential (Davies and Nelson 1994; Lammert and Allan 1999; Stauffer et al. 2000). Since it is not feasible to experiment with landscapes, improving our understanding of land use effects will largely depend on relating the results of site-specific studies that use similar response measures and techniques to evaluate responses of stream habitat and communities along land use gradients.

In recent years, indices of stream health based on multiple structural and functional measures of local fish assemblages have gained wide acceptance among resource managers and aquatic biologists as stream monitoring and assessment tools. These indices of biotic integrity (IBI, Karr (1991)) integrate information from multiple levels of biological organization in order to provide a broad, ecologically sound tool with which to evaluate the biological condition of streams, and to assess human impacts on stream communities. In this study, we use two indices of biotic integrity along with measures of stream habitat to assess land use effects in a Ridge and Valley watershed. The specific objectives were to i) evaluate the relative effects of urban and agriculture land use on biotic integrity, and ii) determine the spatial scale (i.e., whole catchment vs riparian zone) in which land use is most strongly correlated with biotic integrity. We also compare the results of this study to other studies with similar goals and methods in an effort to derive hypotheses regarding potential factors influencing regional and site-specific differences in stream ecosystem responses to urban and agricultural land use.

Methods

Study area

Opequon Creek (Figure 1) drains 894 km^2 of the northern Shenandoah Valley in Virginia and West Virginia before emptying into the Potomac River on the USA east coast, near Washington, DC. The basin is located within the Central Appalachian Ridge and Valley physiographic province and is underlain by limestone and shale geology. The predominant land use in the basin has been agriculture for well over a century. Currently, 57% of the watershed is in agriculture, most of which is pasture, although some rowcrops and apple and peach orchards also occur. Two urban areas exist in the basin; Winchester, Virginia in the southern portion of the basin, and Martinsburg, West Virginia in the northern portion (Figure 1). In addition, numerous smaller municipalities and residential developments are scattered throughout the watershed and urban land use represents about 5% of the total watershed. In the last 30 years, the basin has experienced substantial suburban growth; between 1970

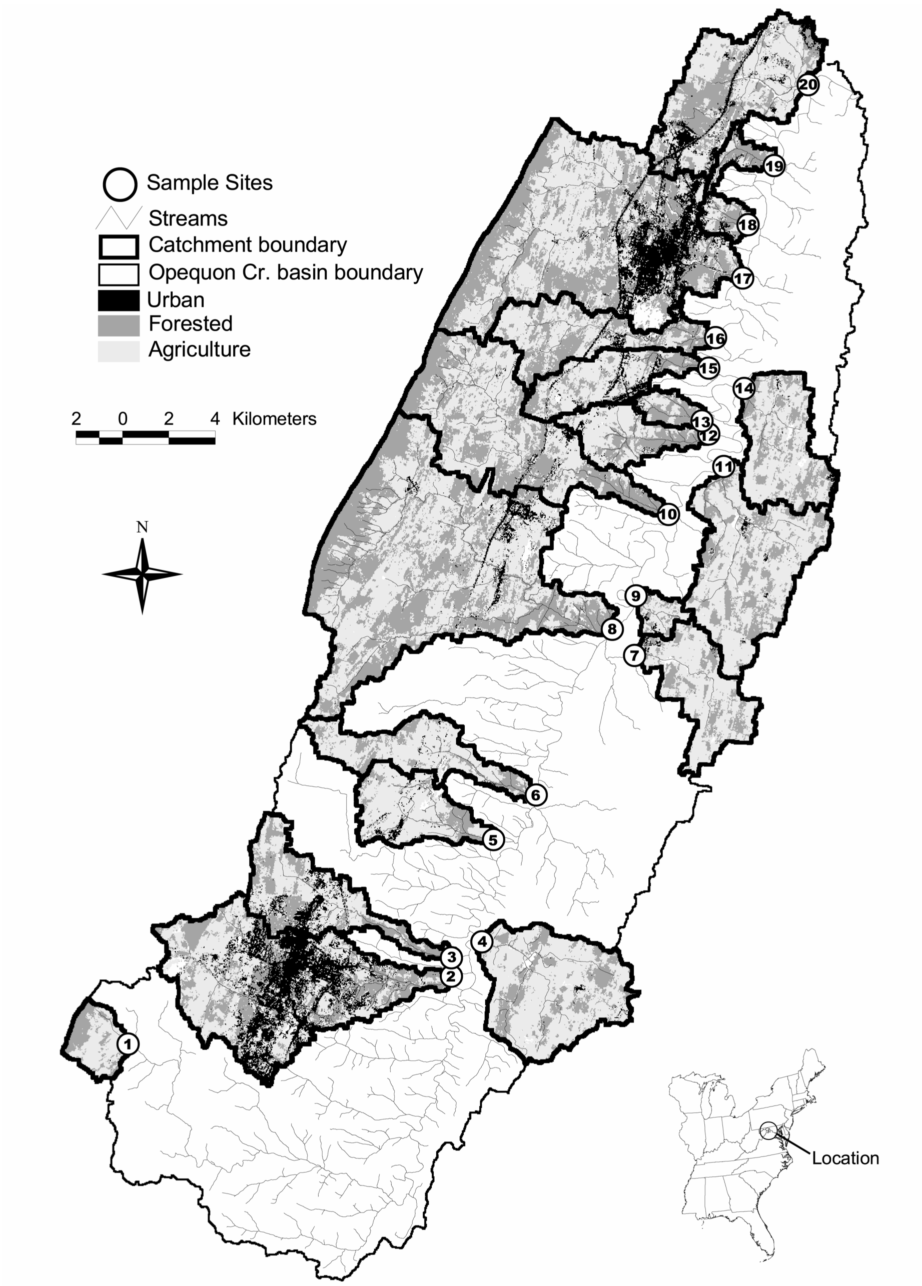

Figure 1. Map of Opequon Creek watershed depicting the 20 catchments sampled and land use/land cover. Fish and instream habitat assessments were made near the bottom of each catchment (numbered circles).

and 1990 the human population has increased 53% (derived from the Master Area Reference File of the U.S. Census Bureau). Currently, forest covers about 37% of the basin and the remaining area is mainly water (primarily farm ponds), barren (mostly limestone and shale mines), and a small amount of forested wetlands.

Study design

Twenty 2nd and 3rd order tributaries of Opequon Creek were sampled for fish and stream habitat, and land use was assessed in their associated catchments. Streams were selected to represent the geographical extent of the Opequon Creek watershed, and included only streams that had permanent flow all year and land use data available in their respective catchments. Ecological assessments (i.e., fish and stream habitat measurements) were conducted within stream reaches located near the bottom of each tributary. The bottom of each study reach was at least two riffle-pool sequences above the confluence with Opequon Creek to minimize hydrologic influences of the mainstem Opequon. We located sample sites near the bottom of tributaries so that ecological assessments would represent stream responses to the cumulative condition of the catchment. In addition, sampling single sites in replicate catchments rather than sampling multiple sites within a single catchment ensured statistical independence among sampling units. Moreover, by selecting catchments within a watershed for comparison, we have largely controlled for natural variation in climate and the zoogeography of fishes.

Fish and stream habitat were measured at sites defined as 40 times mean stream width. We chose this distance because at least two riffle-pool sequences were represented within these boundaries, and because it has been found sufficient to characterize fish diversity patterns in small streams (Lyons 1992; Angermeier and Smogor 1995). Thus, sample sites varied in length between 64 and 428 m in our study.

Landscape influences including land use and stream channel slope were summarized at three spatial scales: a whole-catchment scale which included the entire drainage area upstream of sample sites, a riparian-reach scale defined as a 120-m buffer area adjacent to each stream bank and 400 times mean stream-width in length, and a riparian-site scale defined as a 30-m buffer and 80 times mean stream-width in length. The bottom of both riparian buffers corresponded with the bottom of the reaches where fish and habitat sampling occurred. We used stream width to define the lengths of buffer areas because longitudinal boundaries with which streamside vegetation influences stream channel conditions are proportional to stream size (Schiemer and Zalewski 1992).

Analyses compared the strength of univariate relationships between percent land use and stream habitat and a fish index of biotic integrity (IBI) across spatial scales using correlation and regression analyses. Within each scale, the independent variables (i.e., percent of different land use types) are necessarily auto-correlated. Thus, our use of regression analyses in this study is meant to be descriptive, providing a measure of the relative strength of these relationships (Roth et al. 1996).

Landscape analyses

We mapped sample sites in the field using a Trimble Pathfinder Pro XL global positioning system (GPS). We collected and analyzed landscape information using a geographic information system (GIS) (ArcInfo, ESRI Inc., Redlands, CA) for the catchment and riparian zone immediately above the sample sites. Landscape information measured or calculated for this study included measures of stream length, topographic slope, catchment area, and land cover type. All map layers were projected into the Universal Transverse Mercator (UTM) projection system (zone 17), using the North American 1983 datum.

We digitized stream lines from US Geological Survey (USGS)1:24,000 scale topographic quadrangle maps. Seventeen USGS quadrangles were digitized individually and joined together for an Opequon watershed-wide stream map. USGS 1:24,000-scale digital elevation models, corresponding to these same topographic maps, were joined together using GIS to create a basin-wide map of topography. Sample site positions mapped using GPS were overlain on these layers to derive catchment boundaries.

Catchment boundaries were derived using watershed modeling programs available in the ArcInfo software package. The catchment delineation process consisted of processing the digital elevation models into models of flow accumulation and flow direction; these models (maps) were then used to determine all cells flowing into the sample site. The collection of such cells forms a catchment boundary map. Resulting catchment boundaries and sample site locations are depicted in Figure 1. Catchment boundaries were used in subsequent processing to summarize landscape attributes above sample sites.

Land use/land cover data were gleaned from three primary sources of varying spatial scale. For whole-catchment land use summaries, we used the USGS/EPA National Land Cover Database (formerly called the Multi-Resolution Land Cover project, or MRLC) for Federal Region 3 (Vogelmann et al. 1998a, 1998b). This dataset was derived from satellite imagery (c. 1991–93), and classifies land cover into 15 classes. We found through field validation surveys and examination of aerial photographs that the satellite images incorrectly classified numerous subcategories of urban (e.g., high density vs low density) and agriculture (e.g., grazing vs row-crop), especially in areas where relatively small patches of several subcategories were interspersed. Consequently, we collapsed the number of land cover classes to six broader categories: urban, agriculture, forest, wetlands, water, and barren, and we used this six-category convention for all three scales. The map scale of whole-catchment data is approximately 1:100,000 and has a minimum mapping unit of 900 m^2.

Riparian-reach-scale land use/land cover was mapped for 18 of 20 sampled catchments by manual interpretation of 1:12,000 digital orthophotographs and aerial photographs. Orthophotographs were obtained from the USGS, National Mapping Division and were created from c. 1991 aerial photography. Orthophotographs were not available for two of the sample sites. Riparian-reach-scale land cover was summarized at 25-m^2 minimum mapping units within the resulting polygonal area.

Riparian-site-scale land use/land cover information was assessed from field surveys at 16 of the 20 sites in the summer of 1996. At each site, eighty transects spaced one mean stream-width apart were established perpendicular to, and on both sides of the stream. Land cover was visually assessed in three zones along each transect: 1–2 meters, 2–10 meters, and 10–30 meters from the stream bank (i.e., estimated level of bankfull discharge). The width of all three zones was one mean stream-width and extended one-half the distance to adjacent transects on either side. For each transect, we classified land use/land cover into one of the six broad land cover categories. Because the three zones varied in size, the frequency of each land use/land cover was weighted by the size of the zones in which they occurred and summed to calculate overall percentages.

Stream slope was calculated to roughly correspond to the catchment, riparian-reach, and riparian-site scales used for landscape summaries. We measured stream slope at all scales by overlaying the stream map on a map of slope (degrees) calculated from the digital elevation model using ArcInfo. In this map, slope for each 900 m^2 cell is calculated by comparison with cells in a 9×9-cell neighborhood. Cells on the slope map falling directly under the stream layer were determined and averaged for all streams occurring above the sample site (catchment scale), for stream segments in a riparian-reach segment above the sample site, and for stream segments occurring within five times the reach length of the riparian-site surveys.

Stream assessments

Fish and stream habitat sampling was conducted between July and October 1994 which corresponded to base flow conditions. For fish sampling, stream reaches were stratified into a series of riffle, pool, and run mesohabitats. Beginning at the downstream end of the study reach, we blocked the upstream and downstream ends of each mesohabitat with block nets, and shocked each separately using the three-pass removal approach. A sample was the sum of all three passes. Fish were collected with a Smith-Root Model 12-B backpack shocker or 2.5 GPP shore-based system, identified to species, and released. Young-of-the-year fish were not considered in the analyses.

For assessments of biological integrity at study sites, we used two separate Indices of Biotic Integrity (IBI). One index was developed, tested, and validated for highland streams in the State of Maryland (USA) by Roth et al. (2000). This seven-metric index (MD-IBI) was derived from a larger list of 41 candidate metrics based on its ability to discriminate between reference and degraded stream sites as defined in Roth et al. (1998). Individual metrics were assigned a score of 1 (worst), 3, or 5 (best) depending on criteria outlined in Roth et al. (2000). Site IBI scores were calculated as the mean of seven individual metric scores and therefore also ranged between one and five. Sites with mean scores above 4.0 are considered to have ″good″ biotic integrity (i.e., on average, biological metrics fall within the upper 50% of reference site values); sites with mean scores between 3.0 and 3.9 are considered to have ″fair″ biotic integrity (i.e., on average, biological metrics fall between the 10^{th} and 50^{th} percentile of reference site values); sites with mean scores between 2.0 and 2.9 are considered to have ″poor″ biotic integrity (i.e., on average, biological metrics fall below the 10^{th} percentile of reference site values), and sites with mean scores below 1.9 are considered to have ″very poor″ biotic integrity (most or all biological metrics fall below the 10^{th} percentile of reference site values) (Roth et al. 2000).

The other IBI was developed and tested for streams in the mid-Atlantic Highlands region, U.S. (McCormick et al. 2001). This nine-metric index (MAH-IBI) was derived from a list of 58 candidate metrics. As with the MD-IBI, McCormick et al. (2001) evaluated metrics for their power to discriminate between reference and impacted sites, and metric combinations were tested for redundancy to maximize classification efficiency. Each of the nine

Table 1. Fish metrics used to calculate indices of biotic integrity for Opequon Creek tributaries. The seven-metric Maryland index (MD-IBI) was taken from Roth et al. (2000) and the nine-metric Mid-Atlantic Highland index (MAH-IBI) was taken from McCormick et al. (2001).

Individual metrics Category	MD-IBI	MAH-IBI	Response to degradation
Species richness	No. of benthic species[1]	No. of benthic species[2]	Decline
		No. of cyprinid species[2]	Decline
Indicator species	No. of intolerant species[1]	No. of intolerant species[2]	Decline
	% tolerant fish	% of tolerant fish	Increase
	% abundance of dominant species	% of fish in Family Cottidae	Decline
		% of non-indigenous fish	Increase
Trophic structure	% of fish as generalist feeders	% of macro-ominivores	Increase
	% of fish as insectivores	% of invertivore-piscivores	Decline
Repro. function	% of fish as lithophilic spawners	% of fish as gravel spawners	Decline

[1]Metric adusted for basin size using equations described in Roth et al. (2000).; [2] Metric adjusted for basin size using equations described in McCormick et al. (2001).

metrics was assigned a score between 0 and 10 based on criteria outlined in McCormick et al. (2001). The site score was the sum of the nine metrics times 1.11 so that site IBI ranged between 0 and 100. The terrain, geology, and fish fauna of Opequon Creek are comparable to that for the highland region of Maryland and the mid-Atlantic Highland Area and therefore these indices should be well-suited for use on Opequon Creek. Metrics used to calculate both indices are described in Table 1. A list of species and associated traits used in IBI scoring are found in Table 2. Trophic guild assignments, relative tolerances to habitat degradation, habitat preferences, and reproduction habits information were determined by regional literature references as outlined in Roth et al. (2000) for the MD-IBI, and McCormick et al. (2001) for the MAH-IB.

Instream habitat assessments were conducted at each site at least one day prior to fish sampling. We used a combination of systematic and stratified-random sampling approaches to assess stream habitat. Initially, a series of 80 equally spaced transects were established perpendicular to stream flow. Transects were spaced one half of the average stream-width apart. This distance ensured that data from at least two transects (usually more) were used for individual mesohabitats (e.g., riffles and pools). Subsequently, we took depth, water velocity, and substrate measurements at individual points along the transect. In order to maximize sampling efficiency, we used a stratified-random approach to select point locations along each transect. First, we visually classified channel unit types under each transect using protocols described by Hawkins et al. (1993). This hierarchical system classifies channel unit types according to water velocity, turbulence, depth patterns, pool-forming processes, and locations of habitat units within the stream channel. Subsequently, we took three depth and velocity (measured at 60% depth) measurements, and five substrate measurements within the lateral boundaries of each channel unit across the length of each transect. These methods improved efficiency by ensuring that all habitat types were sampled independent of their rarity and because a disproportionate amount of effort was not put into sampling relatively homogeneous habitats. In addition, it allowed us to summarize data by habitat type (e.g., maximum pool depths) and gave us an independent estimate of habitat diversity (i.e., number of channel unit types occurring in each stream reach). Although the channel unit classification method we used is admittedly subjective (i.e., based on visually-determined estimates of channel morphology and hydrology), the same team of investigators conducted these assessments at all sites and so any bias should be consistent among sites.

Water velocity (ft/sec or cm/sec) was measured using a Marsh-McBirney digital flow meter and depth was measured with a meter stick to the nearest cm. Substrate particles at each sample point were classified into one of 13 size categories based on the length of the intermediate axis of each particle. Substrate size categories were a modification of the Wentworth scale (Cummins 1962) and represented a geometric progression in sizes from silt (< 2 mm) to boulder (> 256 mm). Fish cover estimates including the amounts of large woody debris (% of reach area), overhanging vegetation (% of bank length), and undercut banks (%

Table 2. Fish species collected in Opequon Creek tributaries and associated life history traits used to calculate indices of biotic integrity. For the Maryland index (MD), Roth et al. (2000) classified fish as follows: For trophic guild, TP = top predator, GE = generalist, IV = invertivore, IS = insectivore, OM = omnivore, and AL = algivore; for tolerance, T = tolerant, I = intolerant, and – = no tolerance category assigned; for habitat preferences, BE = benthic, no assignment = other habitat preferences; for reproduction, LI – lithophilic spawners (i.e., require mineral substrates for breeding), no assignment = other reproductive modes; and for introduced, N = native to the Chesapeake Bay drainage, or I = introduced. For the Mid-Atlantic Highlands index (MAH), McCormick et al. (2001) classified fish as follows: For trophic guild, IN = invertivore, IP = Invertivore-Piscivore, and OH = Omnivore-Herbivore; for tolerance, T = Tolerant, I = Intolerant, and – = no tolerance category assigned; for habitat preferences, BE = benthic and CO = water column; for reproduction, AT = Egg attacher, BS = broadcast spawner, CG = clean gravel spawner, NA = nest associate, and NG = nest guarder; for introduced, N = native to the Potomac River basin, I = introduced to the Potomac.

	Trophic		Tolerance		Reproduction		Habitat		Introduced	
Family/species	MD	MAH	MD	MAH	MD	MAH	MD	MAH	MD	MAH
Anguillidae (freshwater eels)										
Anguilla rostrata	GE	IP	–	T		–		CO	N	N
Cyprinidae (minnows)										
Campostoma anomalum	AL	OH	I	–	LI	CG		BE	N	N
Clinostomus funduloides	IV	IN	I	–	LI	NA		CO	N	N
Cyprinella spiloptera	IV	IN	I	–		AT		CO	N	N
Luxilus cornutus	OM	IN	I	T	LI	NA		CO	N	N
Margariscus margarita	IV	IN	–	–	LI	BS		CO	N	N
Nocomis micropogon	OM	IN	I	–	LI	CG		CO	N	N
Notemigonus crysoleucas	OM	OH	T	T		BS		CO	N	N
Notropis amoenus	OM	IN	I	I	LI	NA		CO	N	N
Notropis hudsonius	OM	IN	I	–	LI	BS		CO	N	N
Notropis procne	IV	IN	I	–	LI	NA		CO	N	N
Notropis rubellus	IV	IN	–	–	LI	NA		CO	N	N
Pimephales notatus	OM	OH	T	T		AT		CO	N	N
Pimephales promelas	OM	OH	–	T		AT		CO	I	I
Rhinichthys atratulus	OM	OH	T	T		CG		BE	N	N
Rhinichthys cataractae	OM	IN	I	–		CG		BE	N	N
Semotilus atromaculatus	GE	IP	T	T	LI	NG		CO	N	N
Semotilus corporalis	GE	IP	I	–	LI	NG		CO	N	N
Catostomidae (suckers)										
Catastomus commersoni	OM	OH	T	T	LI	BS		BE	N	N
Erimyzon oblongus	IV	OH	–	–		BS		BE	N	N
Hypentelium nigricans	IV	IN	I	I	LI	CG		BE	N	N
Moxostoma erythrum	OM	IN	–	–	LI	CG		BE	N	I
Ictaluridae (bullhead catfish)										
Ameiurus natalis	OM	OH	–	T		NG	BE	BE	N	N
Noturus insignis	IV	IN	I	–		NG	BE	BE	N	N
Cottidae (sculpins)										
Cottus bairdi	IS	IN	I	I	LI	NG	BE	BE	N	N
Cottus cognathus	IS	IN	I	–	LI	NG	BE	BE	N	N
Cottus girardi	IS	IN	–	–	LI	NG	BE	BE	N	N

Table 2. Continued.

	Trophic		Tolerance		Reproduction		Habitat		Introduced	
Family/species	MD	MAH	MD	MAH	MD	MAH	MD	MAH	MD	MAH
Centrarchidae (sunfishes)										
Amboplites rupestris	GE	IP	–	–	LI	NG		CO	I	I
Lepomis auritus	GE	IP	I	–		NG		CO	N	N
Lepomis cyanellus	GE	IP	T	T		NG		CO	I	I
Lepomis gibbosus	IV	IN	T	–		NG		CO	N	N
Lepomis macrochirus	IV	IN	T	T		NG		CO	I	I
Micropterus dolomieui	TP	IP	–	–		NG		CO	I	I
Micropterus salmoides	TP	IP	T	–		NG		CO	I	I
Fundulidae (killifishes)										
Fundulus diaphanous	IV	IN	–	T		AT		CO	N	N
Percidae (perches)										
Etheostoma blennioides	IS	IN	–	–		AT	BE	BE	N	N
Etheostoma flabellaria	IS	IN	–	T	LI	NG	BE	BE	N	N
Etheostoma olmsteadi	IS	IN	T	T		NG	BE	BE	N	N
Salmonidae (trouts)										
Oncorhynchus mykiss	TP	IP	–	I	LI	CG		CO	I	I
Salvelinus fontinalis	GE	IP	I	I	LI	CG		CO	N	N

of bank length) were visually estimated to the nearest 10% for the study reach. Each estimate was made for each mesohabitat within the sampling reach, weighted by the relative amount of each mesohabitat in the sampling reach, and the weighted values summed to obtain reach-wide estimates. Large woody debris included wood > 0.3 m in diameter and at least 0.5 m in length. Undercut banks and overhanging vegetation estimates included portions of the stream bank that would provide cover for large (i.e., > 20-cm) fish. These estimates were admittedly subjective but were conducted by the same team of investigators at all sites, so any biases should be consistent across sites.

Water quality samples were taken and stream discharge measurements were made at the time of fish sampling to represent summer base-flow conditions, and in February 1995, during a period of prolonged rainfall, to represent high-flow conditions. Three measures of water quality were summarized from water chemistry measurements: total nitrate concentration, ammonia concentration, and turbidity (Table 3). Several other variables were measured but were either highly correlated with other selected measures (e.g., total suspended solids highly correlated with turbidity), showed little meaningful variation among sites (e.g., dissolved oxygen), or were believed to be unreliable for synoptic surveys (e.g., orthophosphate concentrations). Water chemistry data for the two seasons were highly correlated ($r > 0.63$ for all three measures) though summer measures were more variable, and the winter measures exhibited stronger associations with land use. Consequently, we report only winter high-flow measurements.

We summarized habitat data into a suite of 16 variables known to be indicative of stream degradation. They included measures of hydrology, channel morphology, habitat diversity, water quality, sediment size, and fish cover (Table 3). As with the fish metrics described above, some habitat measures are strongly related to basin area. Specifically, measures of mean stream width and average maximum pool depth (Table 3) were linearly related to the log of watershed area ($r^2 = 0.57$, $p = 0.0001$; and $r^2 = 0.62$, $p = 00004$, respectively). Consequently, we used the residuals of those relationships (i.e., remaining variation after accounting for the effect of basin area) as summary statistics in our analyses. We also tested habitat diversity for a basin-size relationship but found no relationship ($r^2 = 0.07$, $p = 0.25$).

Table 3. Description of instream habitat variables measured at, or calculated for, each study site.

Variable	Description
Channel characteristics	
Mean width	-Average wetted channel width (m) at base flow calculated from 40 transects and adjusted for basin area (see text).
Mean maximum pool depth	-Average of the maximum depths (cm) of all non-eddy pool habitats in reach adjusted for basin area (see text).
Microhabitat diversity	-Number of channel unit types (Hawkins et al. 1993) present in study reach.
CV Pool depth	-Coefficient of variation of all pool depth measurements excluding eddy pools.
CV riffle velocity	-Coefficient of variation of water velocity (cm/s) measurements taken in riffles.
Water quality	
Turbidity	-Measured with LeMotte Model 2008 turbidity meter (NTU) in well-mixed areas (riffles).
Total nitrates	-Total nitrate as N (mg/l) concentrations measured on grab samples according to Standard Methods for the Examination of Water and Wastewater (1992).
Ammonia	-Ammonia as N (mg/l) concentrations measured on grab samples according to Standard Methods for the Examination of Water and Wastewater (1992).
Hydrologic stability	
Base flow	-Discharge (cm^3/sec) measured at base flow and adjusted for basin area (see text).
High flow	-Discharge (cm^3/sec) measured during sustained period of high flow and adjusted for basin area (see text).
Riffle sediments	
Fine sediments	-Proportion of sand, silt and clay particles in riffles
Rocky sediments	-Proportion of cobble and boulder particles in riffles
Fish cover	
Large woody debris	-Proportion of stream area containing wood > 0.3 m diameter and > 0.5 m in length.
Undercut banks	-Proportion of stream bank length with undercut banks.
Overhanging vegetation	-Proportion of stream bank length with overhanging vegetation.

Results

Land use/cover patterns

GIS analyses of satellite-derived land use data in the 20 catchments revealed that, at the whole-catchment scale, land use was largely composed of gradients in agriculture, forest and urban land uses. Other land uses including wetlands, water, and barren land categories represented relatively minor components of the watershed (< 2% combined for all 20 catchments). Agriculture land use was pervasive in all 20 catchments, ranging between 38% and 74% (median = 56%) (Figure 2). Likewise, a significant amount of forest cover was observed in all catchments and varied between 22% and 53% among catchments (median = 33.5%) (Figure 2A). Eight catchments had a significant amount of urban land use (i.e., > 7%), with a maximum of 28% for Abrams Creek (site 2). Two catchments had no measurable urban land use at this scale and 10 catchments had an intermediate level (1–4%) (Figure 2A). At the catchment scale, agricultural land use was inversely related to both urban ($r = -0.70$) and forest ($r = -0.77$), but urban showed no meaningful association with forest ($r = -0.08$).

As with the whole-catchment scale, urban, agriculture, and forest land uses dominated riparian zones at the two measurement scales. Total contribution of all other land use types combined averaged < 2% in riparian zones measured at either scale. However, within each land use category there was considerably more variation among riparian areas than among catchments. At the riparian-reach scale, aerial photos in 18 of the 20 catchments indicated that agriculture land use ranged between 0% and 65% (median = 32%), forest cover between 18% and 97% (median =

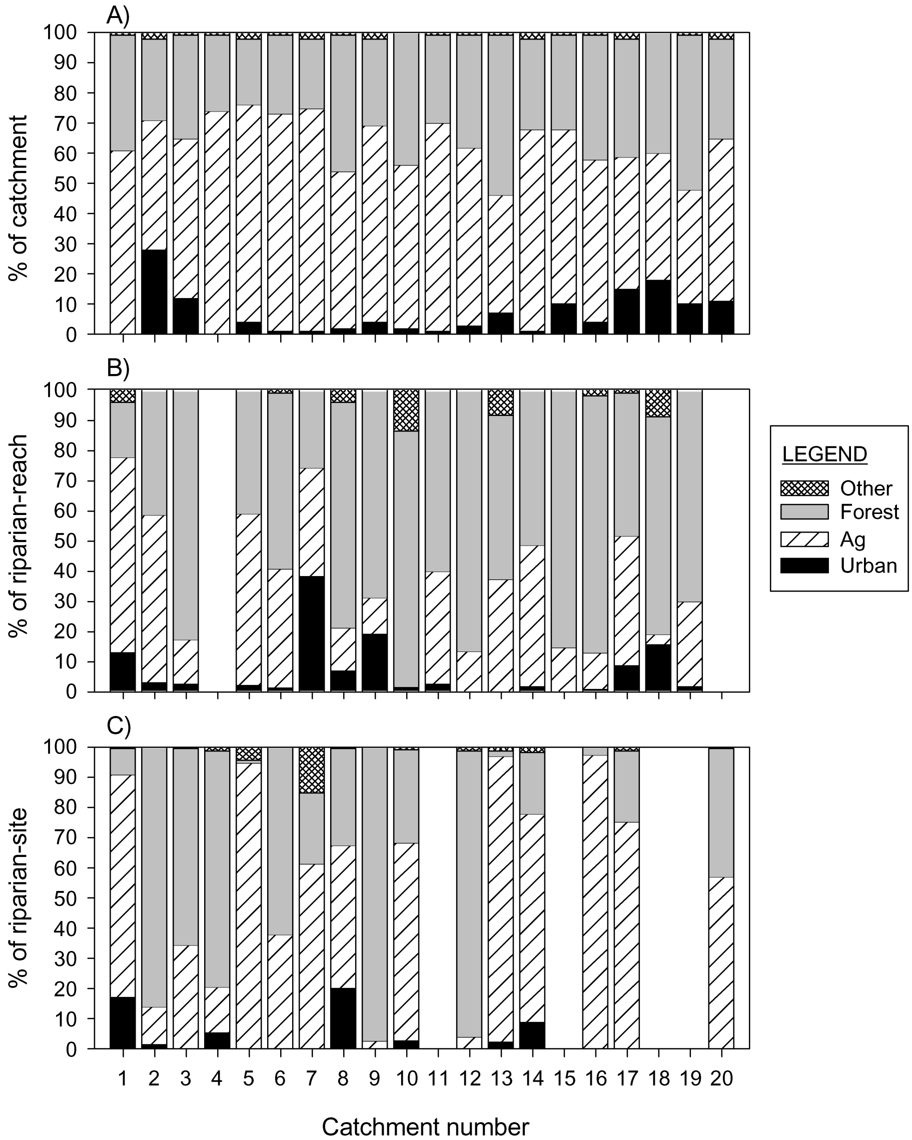

Figure 2. Percentages of urban, agriculture, forest and other land use/land cover measured at three spatial scales for the 20 catchments sampled in Opequon Creek watershed.

65%), and urban between 0% and 38% (median = 2.2%) (Figure 2B). Field surveys in the smaller riparian areas adjacent to fish sampling sites (i.e., riparian-site) indicated that agriculture land use ranged between 3% and 97% (median = 59%), forest cover between 1% and 95% (median = 31%), and urban between 0% and 20% (median = < 1%) (Figure 2C). At the riparian-reach scale, forest land cover was negatively related to both agriculture (r = –0.82) and urban (r = –0.56), but urban and agriculture land uses were not correlated (r = 0.19). At the riparian-site scale, forest land cover was negatively correlated with agriculture land use (r = –0.95) but urban land use did not correlate with either forest (r = –0.20) or agriculture (r = 0.15).

Stream channel slope was highly correlated with land use/land cover in Opequon Creek catchments. Average stream channel slope measured at the whole catchment scale ranged between 17 and 62 m/km and was negatively correlated with agriculture land use (r = –0.42) and positively correlated with forest land cover (r = 0.52) in individual catchments. Average stream slope measured at the riparian-reach scale ranged between 11 and 45 m/km and was negatively

Table 4. Pearson correlation coefficients between two indices of biotic integrity and three land use land cover classes measured at three scales. MD = Maryland index of biotic integrity (Roth et al. 2000); and MAH = Mid-Atlantic Highland index of biotic integrity (McCormick et al. 2001).

Scale:	Whole-watershed			Riparian-reach			Riparian-site		
Land use:	Urb	Ag	For	Urb	Ag	For	Urb	Ag	For
IBI:									
MD	–0.80	0.66	–0.12	0.41	–0.21	–0.03	0.38	0.14	–0.25
MAH	–0.73	0.40	0.16	0.16	–0.23	0.11	0.26	0.27	–0.32

correlated with agriculture ($r = -0.51$) and positively correlated with forest land cover ($r = 0.57$) in the riparian zone. Slope measured at the riparian-site scale varied between 13 and 40 m/km and was not correlated with land use. Urban land use was not correlated with channel slope at any of the scales measured ($r = 0.10$ at the whole-catchment scale, $r = -0.21$ at the riparian-reach scale, and $r = 0.09$ at the riparian-site scale).

Index of biotic integrity

We found that % urban land use in individual catchments had a strong, negative association with both MD-IBI and MAH-IBI scores (Table 4). A weaker, positive association was observed between whole-catchment agriculture and MD-IBI scores. Correlations between the remaining land use categories and IBI scores were weak at all three scales (Table 4).

We used regression analysis to explore the effectiveness of multivariate models in predicting site IBI scores. First, we were interested in whether models incorporating multiple land uses were better predictors of site IBI scores than percent urban by itself. Because urban land use measured at the whole-catchment scale was the strongest correlate of both IBI's (Table 4), we regressed IBI scores against% urban at the catchment scale and examined the relationships between the residuals and each of the other land use variables including% urban at the two riparian scales. The univariate regressions of% urban land use on site IBI scores explained 63% of the variation in MD-IBI and 60% of the variation in MAH-IBI. None of the other land use variables at any of the three scales explained a significant ($p < 0.10$) amount of the remaining variation in IBI after accounting for the effects of urban land use. For the MD-IBI, percent forest measured at the riparian-reach scale was best ($p = 0.11$), accounting for 13% of the remaining variation in MD-IBI. For the MAH-IBI,% agriculture at the riparian-reach scale was best ($p = 0.41$) accounting for only 2.1% of the remaining variation in MAH-IBI.

Secondly, we used multiple regression to test whether stream channel slope improved IBI predictions based on percent urban land use alone. We tested three regression models for each IBI, each of which included percent urban land use measured at the whole-catchment scale and stream channel slope measured at one of the three measurement scales along with their respective interactions. We found that the effects of urban land use on both indices of biotic integrity were greater in steeper catchments (Figure 3). Including mean channel slope measured at the whole-catchment scale into regression models increased the amount of variance explained from 63% to 83% for the MD IBI, and from 60% to 76% for the MAH IBI, relative to the amount of variance explained by urban land use alone. Partial F-tests indicated the multivariable models were a significant improvement over univariate models that included percent urban alone ($F = 9.94$, df = 2,16, $p = 0.002$; $F = 5.30$, df = 2,16, $p = 0.017$; for the MD-IBI and MAH-IBI respectively). For the MD-IBI, Roth et al. (2000) defined streams with site scores less than two as ″very poor″. The model predicts that MD-IBI in lower-gradient streams (i.e., mean stream channel slope < 30 m/km) would become very poor when the percent urban in the catchment exceeds about 21%. In contrast, MD-IBI in high gradient streams (i.e., mean stream channel slope > 30 m/km) would become very poor when about 9% percent of the catchment is in urban land use (Figure 3). Models with channel slope measured at either the riparian-reach or riparian-site scales did not explain significantly more variation in either IBI than the univariate models with percent urban alone ($p > 0.18$ for all).

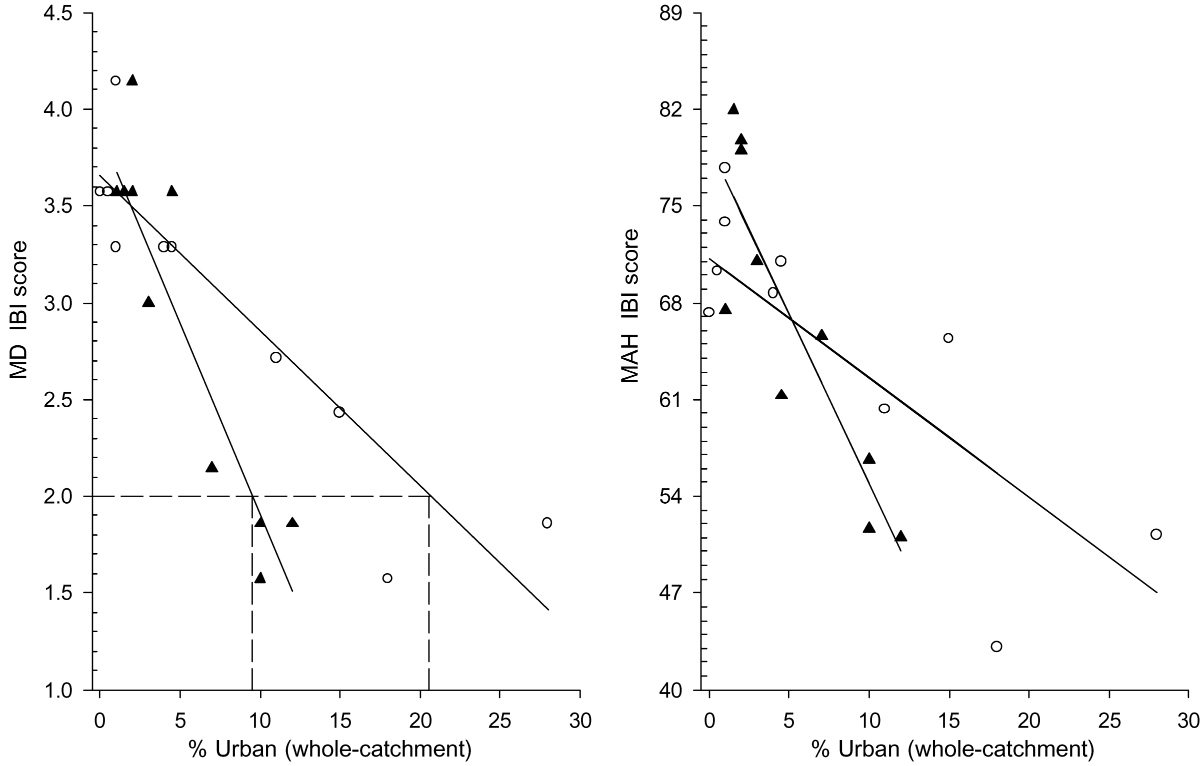

Figure 3. Relationship between IBI and percent urban land use in individual catchments as influenced by stream channel slope (measured at the whole-catchment scale). The MD-IBI (Roth et al. 2000) is shown on the left and the MAH-IBI (McCormick et al. 2001) is shown on the right. For each graph, open circles depict low-gradient sites (i.e., stream channel slope < 30 m/km) and filled triangles depict high-gradient sites (i.e., stream channel slope > 30 m/km). Lines were generated from regression models and symbols represent actual data. For the MD-IBI, dotted lines show threshold at IBI = 2 for comparison.

Habitat assessment

We found strong correlations between measures of stream habitat and land use. At the whole-catchment scale, percent urban had a strong positive association with average stream width and a strong negative association with basin area-adjusted base flows (Table 5). Total nitrate concentrations were positively associated with percent agriculture and negatively associated with percent forest, and ammonia concentrations were negatively correlated with agriculture and positively correlated with urban in individual catchments (Table 5). In addition, the amount of forest in individual catchments was negatively correlated with the proportion of fine sediments in riffles and with basin area-adjusted high flows (Table 5), but both of these habitat measures exhibited stronger correlations with riparian land use patterns.

Other habitat variables exhibited stronger correlations at the two riparian scales. At the smallest scale (i.e., riparian-site), two of the three fish cover measures, large woody debris and undercut banks, exhibited strong positive associations with forest land cover and negative associations with agriculture land use (Table 5). Likewise, riffle-pool ratios were positively associated with forest land cover and negatively associated with agriculture land use (Table 5). At the intermediate scale (i.e., riparian-reach), basin area-adjusted high flow was positively correlated with urban land use, and the percent fine sediments in riffle areas was positively correlated with agriculture land use and negatively correlated with forest land cover (Table 5). Variation in pool depth was positively associated with forest land cover and negatively correlated with agriculture land use at both riparian scales (Table 5).

Table 5. Associations among 16 habitat variables and land use summarized at three spatial scales. Values are Spearman rank correlation coefficients. Strong correlations (r > 0.50) are in bold.

	Whole catchment			Riparian – reach			Riparian – site		
Habitat class/Variable	Urb	Ag	For	Urb	Ag	For	Urb	Ag	For
Channel morphology									
Mean width	**0.71**	**–0.59**	0.09	0.09	–0.13	0.01	–0.21	–0.14	0.17
Maximum Depth	0.48	–0.30	0.23	0.30	–0.33	0.16	–0.42	0.10	–0.09
Riffle-Pool ratio	–0.14	–0.01	0.21	0.07	–0.38	**0.58**	0.27	**–0.60**	**0.52**
Hydrology									
Base flow	**–0.86**	**0.59**	**–0.62**	0.19	0.35	–0.37	–0.10	0.09	–0.03
High flow	0.49	0.05	**–0.50**	**0.65**	–0.01	–0.21	**–0.51**	–0.16	0.27
Water quality									
Nitrates	–0.42	**0.77**	**–0.70**	0.16	0.20	–0.21	–0.13	0.09	0.01
Ammonia	**0.52**	**–0.54**	0.17	–0.08	–0.30	0.07	0.02	0.06	–0.07
Turbidity	0.34	–0.31	0.02	0.38	–0.14	–0.02	0.13	0.15	–0.11
Fish Cover									
Large wood	0.10	0.13	–0.27	0.33	–0.05	0.22	–0.28	**–0.69**	**0.77**
Over-hanging veg.	–0.05	–0.01	–0.15	0.22	0.13	–0.11	0.31	0.09	–0.12
Undercut banks	0.13	0.05	–0.25	0.19	–0.09	0.22	–0.16	**–0.69**	**0.79**
Habitat diversity									
Patch richness	0.24	–0.26	0.26	0.36	–0.25	0.36	–0.20	–0.39	0.49
CV pool depth	0.19	–0.22	0.20	–0.14	**–0.51**	**0.59**	–0.09	**–0.55**	**0.52**
CV riffle velocity	–0.08	–0.03	0.06	–0.02	–0.02	–0.13	0.34	–0.12	0.17
Sediments									
% fines in riffles	–0.15	0.43	**–0.61**	–0.14	**0.77**	**–0.74**	–0.16	0.45	–0.35
% large rocks	0.03	–0.05	0.21	0.02	–0.16	0.05	–0.14	0.21	–0.11

Discussion

Relative impacts of urban and agricultural land uses

We used two different indices of biotic integrity to evaluate land use effects on Opequon Creek tributaries. The two indices differed markedly in two important ways. First, several of the component metrics incorporated into each index were different. The seven-metric MD-IBI and the nine-metric MAH-IBI had only three metrics in common, although several others were similar and probably correlated (Table 1). Second, and perhaps more important, were the numerous differences in the ecological classifications of individual fish species that were ultimately used to calculate metric scores (Table 2). In many respects, assigning fish species to broad ecological categories based on the literature is a subjective process. As with most animals, there is often considerable variation in life history within a species depending on such factors as age, food and habitat availability, and abundance of competitors and predators (Wootton 1990). Moreover, investigators frequently use different literature sources to make assignments. In our case, Roth et al. (2000) made finer trophic but broader habitat distinctions for the MD-IBI than McCormick et al. (2001) did for the MAH-IBI. In addition, the tolerance classifications differed substantially. For the MD-IBI, 16 of the 40 fish species collected on Opequon Creek were considered intolerant compared with only five species for the MAH-IBI (Table 2). Yet despite these differences, responses of the two indices to land use were remarkably similar bolstering the conclusions reached in this study.

Based on the strength of land use and fish IBI associations observed in this study, we conclude that urban land use in individual catchments had a disproportionately large effect on biotic integrity in Opequon Creek tributaries. Both indices of biotic integrity used in this study declined sharply with increases in urban land use, and all eight sites that received overall integrity ratings of poor or very poor (i.e., MD-IBI < 3) had greater than 7% of their respective catchments in urban land use. Our finding that urban land use was most disruptive to the biological integrity of stream ecosystems is consistent with previous studies that found strong negative effects of urban land uses on biotic integrity (Steedman 1988; Klauda et al. 1998; Lydy et al. 2000; Schleiger 2000; Wang et al. 2000).

Inferences regarding effects of agricultural land use on biological integrity in Opequon Creek tributaries were less clear. In fact, we observed a positive relationship between the extent of agriculture in individual catchments and site IBI scores. Roth et al. (1998) also found IBI to be negatively related to the amount of urban and positively related to the amount of agriculture in individual catchments. It is likely that the positive association between agriculture and IBI in both studies is due to the fact that the extent of agriculture and urban land uses were negatively correlated with each other. Consequently, fish assemblages in streams draining catchments with a relatively limited amount of agriculture experienced the negative effects of urban land use, while those in catchments with considerable agriculture did not. In addition, the range of agricultural land use in Opequon Creek may not have been wide enough to detect negative effects of agriculture land use on IBI. Wang et al. (1997) found that negative effects of agriculture land use on fish IBI in Wisconsin streams were only observed for sites where the proportion of agriculture exceeded 50%. In Opequon Creek watershed, agricultural land use ranged between 38 and 72 percent (median = 56%) in individual catchments. Moreover, much of the agricultural land use within Opequon Creek watershed is pasture, and although both grazing and row-crop agriculture involve removal of native vegetation, row-crop agriculture also involves the addition of fertilizers and pesticides and direct changes to the soil that make it more erodible (Correll et al. 1992). On the River Raisin in Michigan, where agriculture represented a broader range and nearly all of it row-crop agriculture, Roth et al. (1996) found that percent agriculture was the best predictor of fish IBI scores. Nevertheless, it is clear that urbanization is more disruptive to fish assemblages in Opequon Creek catchments than agriculture on a per-unit-area basis.

Habitat correlations suggested that urban land use disrupted flow patterns (i.e., lower base flows and higher high flows), altered channel size (i.e., increased channel width), and degraded water quality (i.e., increased ammonia concentrations). These results combined with the fact that correlations between urban land use and measures of habitat diversity, fish cover, and substrate characteristics were weak support the conclusion of Wang et al. (1997) that water quality and hydrological impacts of urban land use may be more important than direct effects on physical habitat.

Effects of measurement scale

Our results suggest that catchment-wide land use patterns were more strongly related to biological integrity than riparian land use patterns in the Opequon Creek watershed. None of the univariate relationships between land use measured at either riparian scale and site IBI scores was significant. Even after accounting for the predominant effects of urban land use through partial regression analysis, riparian land use patterns failed to explain a significant amount of the remaining variation in site IBI scores suggesting that forested buffer zones were of little value in mitigating the deleterious effects of urban land use on fish communities. Since riparian land use was measured at fewer sites than whole-catchment land use (riparian-reach = 18 sites, riparian-site = 16 sites, whole-catchment = 20 sites), it is possible that less significant relationships between land uses and fish IBI observed at riparian scales were due to lower sample sizes. However, given the weakness of observed associations, we believe that it is unlikely that data for a few more sites would have enough leverage to significantly improve land use – fish IBI associations at riparian scales.

Although riparian land use patterns were not predictive of biological integrity as defined by the integrated measure of fish assemblage structure, there was strong evidence that riparian land use patterns influenced instream habitat. In particular, fish cover, habitat variability, and sediment characteristics exhibited stronger associations with riparian land use than whole-catchment land use. This is consistent with a wide body of literature (reviewed in Naiman and Décamps (1990)) that documents riparian influences on stream habitat. Moreover, instream habitat variables have been found to be major determinants of fish community structure in some systems (e.g., Gorman and Karr 1978; Schlosser 1982; Sheldon and Meffe 1995). Nevertheless, despite the importance of riparian vegetation in controlling stream habitat, reductions in water quality and alterations in stream hydrology associated with whole-catchment land use patterns appeared to overwhelm the capacity of riparian vegetation to maintain biological integrity in Opequon Creek. Similar conclusions have been reached by others that evaluated land use effects at multiple scales (Richards et al. 1996; Roth et al. 1996; Wang et al. 1997), and these results suggest that protection or restoration of riparian buffers is not sufficient to maintain ecological integrity.

In contrast, results from other studies suggest that forested riparian areas provide substantial protection to streams draining heavily farmed or urbanized catchments (Steedman 1988; Lammert and Allan 1999; Stauffer et al. 2000). Site-specific differences in the relative importance of upland and riparian zones probably relates to differences in other landscape features that interact with land use to determine habitat quality and biological integrity. Of particular importance may be variation in local topography. Few studies have examined land use effects at multiple scales (Wang et al. 1997) and most of those have been conducted in regions where there is little meaningful variation in channel slope (Steedman 1988; Roth et al. 1996; Wang et al. 1997; Lammert and Allan 1999; Stauffer et al. 2000). Furthermore, to our knowledge, there have been no studies explicitly designed to test the interaction between channel slope and the potential of riparian zones to mitigate upland land use effects.

Opequon Creek is typical of basins throughout the Ridge and Valley Physiographic province in that there is considerable variation in channel slope among tributaries, and forested riparian zones are largely limited to relatively high gradient reaches where agriculture is impractical. We found that catchments in steeper terrains were more severely impacted by urban land use. The influence of gradient was only important when channel slope was averaged over the entire stream length within individual catchments, suggesting that slope influences mostly emerge at larger scales, or that the digital elevation maps used to measure gradient did not have the resolution required to accurately measure slope at the smaller riparian scales.

Based on the comparison of our results in the topographically diverse Opequon Creek with those from studies in lower gradient watersheds, we hypothesize that riparian zones in lower gradient systems exert more influence on stream communities, and have a greater potential to mitigate human-induced disturbances such as agriculture and urban development. Gradient is a primary determinant of channel morphology including the distribution and stability of stream habitat (Rosgen 1994). As a result, streams draining high-gradient catchments may be subjected to more frequent disturbances (e.g., channel modifying floods) and stronger landform controls (e.g., landslides and canyons), whereas in flatter topographic settings, watersheds are often characterized by broader, lower-gradient valleys which allow longer periods of surface and subsurface water flows across floodplains and riparian zones (Wissmar and Swanson 1990). The increased contact time between riparian areas and stream channels may increase the efficiency of riparian vegetation in regulating stream flows and in filtering nutrients and possibly other contaminants from runoff, two of the most important upland land use effects on stream habitat (Wang et al. 1997; this study).

Conclusions

Results of this study have significant implications for stream and watershed management in the mid-Atlantic highlands. The observation that even relatively low levels of urban land use are particularly disruptive to biotic integrity is disturbing in light of trends in suburban development in the region. We should expect marked declines in the biological integrity of streams if current land use trends continue. Our results also suggest that efforts to moderate the impacts of urban/suburban sprawl by protecting riparian areas may not be sufficient to maintain biotic integrity, at least in high-gradient catchments. Protection of natural wetlands if they exist or the use of constructed wetlands where they do not, in order to help stabilize flow patterns might be a more successful management prescription.

The exploratory approach used in this study and that of most others designed to examine fish responses to land use (see above) have yielded important findings. However, these site-specific efforts are often complicated by the problem of autocorrelation among land use categories and untestable interactions between land use and other landscape features such as topography and geology. Future research should include studies explicitly designed to test these potentially important interactions. Regional landscape analyses should be employed up front to identify sites and watersheds that reflect the pertinent contrasts and to control for other, potentially confounding variables. In addition, studies are needed to examine the influence of more specific land use types (e.g., grazing vs row-crop agriculture) and how land uses are distributed within the watershed. Finally, the robustness of stream quality indices like the IBI need to be evaluated across a wider range of environmental conditions and landscapes.

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