American Fisheries Society Symposium 48:577–599, 2006
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Effects of Percent Impervious Cover on Fish and Benthos Assemblages and Instream Habitats in Lake Ontario Tributaries

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Abstract.——We demonstrate the effects of percent impervious cover (PIC) on biophysical properties of Lake Ontario tributary streams. Biophysical data (fish assemblages, benthic invertebrate assemblages (benthos), instream physical habitat, and temperature) were collected from more than 575 wadeable stream sites. A geographic information system application was developed to characterize the landscape upstream of each site (i.e., drainage area, surficial geology, land use/land cover, slope, stream length, and climate). Total PIC of catchments was estimated from land use/land cover, and a base flow index was derived from the surficial geology. The relationship between PIC and biophysical responses was determined after statistically removing the effects of natural landscape features (i.e., catchment area, slope, base flow index) on those responses. Contrasts in PIC from natural conditions (<3% to 10%) were related to variations in fish and benthos assemblages. Both coldwater sensitive and warmwater tolerant fish and diverse benthos assemblages were found in catchments with low PIC. At more than 10 PIC (i.e., about 50% urban), both fish and benthos consisted of mainly warmwater or tolerant assemblages. For example, trout were absent and minnows were dominant. While some of the apparent PIC effect may have been confounded by land use/land cover and surficial geology, the consistency of the findings even after natural catchment conditions were considered suggests that the threshold response is valid. Percent impervious cover had a weaker effect on instream geomorphic variables than on biological variables. The models derived from this study can be used to predict stream biophysical conditions for catchments with varying levels of development.

INTRODUCTION

Human populations in the Greater Toronto Area (GTA) and central Ontario are predicted to increase by nearly 3 million over the next 24 years (Statistics Canada 2003), and urban development associated with those increases threatens ecosystems dependent on the Oak Ridges Moraine. Human land use has direct and indirect effects on physical, chemical, and biological characteristics of streams and has been modeled using a variety of land-use/land-cover descriptors. Metrics such as catchment population density (Jones and Clark 1987), agriculture (Harding et al. 1999), width of riparian zones (Barton et al. 1985), and land use/land cover (Kilgour and Barton 1999) have been related to instream biological responses. In general, more intensive development degrades fish and benthos assemblages and instream habitats. A quantitative understanding of the relationships between development and ecological conditions enables

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planners to predict and mitigate impacts from future land conversions, particularly to ensure that thresholds are not exceeded that would lead to irreversible damages to the ecosystem.

Percent impervious cover (PIC) is a metric that integrates various types of human development activities in catchments. Impervious lands are those that have been covered by materials such as concrete, asphalt, and rooftops or result in severe compaction or draining of the soils, all of which restrict infiltration. Different land covers are variably permeable (impermeable). The PIC in a catchment is the weighted average imperviousness for the entire catchment. Percent impervious cover is typically low in natural landscapes, intermediate in agricultural landscapes, and high in urban landscapes. Leopold (1968) recognized that the transition from natural forest cover to agricultural and urban landscapes resulted in increased PIC of the land and led to reduced infiltration of precipitation into soils and increased overland flow. Streams in disturbed catchments tended to respond faster and more severely to storm events, had lower base flows during dry seasons, and were wider, shallower, more polluted, and warmer than streams in undisturbed catchments (Leopold 1968).

Percent impervious cover is recognized as a master variable for quantifying both physical and biological stressors in streams. Shaver and Maxted (1995), demonstrated in Delaware that percent Ephemeroptera-Plecoptera-Trichoptera taxa (mayflies, stoneflies, caddisflies) was significantly lower in streams with PIC greater than 10%. Klein (1979) demonstrated that fish species richness declined linearly with PIC such that catchments with 30–50% PIC either had severely impaired fish assemblages or fish were absent. Several authors have identified impairments to fish metrics at PIC levels greater than 10%. For example Steedman (1988) and Wang et al. (2000) both reported that the number of fish species and/or an index of biotic integrity (IBI) decreased in streams above this threshold. Limburg and Schmidt (1990) documented declines in anadromous fish egg and fry densities above a PIC of 10%. Results for benthos are comparable, although evidence of threshold responses are less convincing. Jones and Clark (1987) identified a threshold response at 15% PIC, while Yoder et al. (1999), Shaver and Maxted (1995), and May et al. (1997) determined that benthos taxa diversity declined above 5–8% PIC. Klein (1979) however was unable to identify a threshold response and suggested that confounding factors such as the presence of riparian zones may buffer the overall effect of PIC on streams. Several studies have confirmed even lower threshold responses to geomorphic variables. Dunne and Leopold (1978) found dramatic change in channel dimensions at only 4% PIC. Booth and Jackson (1997) found that even at low levels of effective impervious cover, flow patterns were significantly altered and that by PIC levels of 6–10% PIC, there is a loss of aquatic system function that may be irreversible.

We believe there are three key factors that explain the differences in response to PIC in these studies: differences in resilience to PIC between ecoregions, differences in how instream features (e.g., fish, substrate, etc.) are measured between studies, and differences in how PIC has been estimated. Until comparisons can be made between ecoregions using similar measures of instream conditions and PIC, application of the relationship between PIC and stream condition must rely on locally derived models.

Our objective was to quantify the relationship between catchment PIC and biophysical characteristics of streams. We recognize that much of the variability in the biophysical makeup of streams is due to natural catchment characteristics and the location of the stream within the catchment. Therefore, we accounted for catchment characteristics that might influence stream hydrology and sediments. By including both the biotic and abiotic properties of streams, we attempt to demonstrate how biotic–abiotic relationships can assist managers in predicting changes from future development scenarios.

METHODS

Study Area

Fish, benthic invertebrates, and instream habitat conditions were characterized at wadeable sites along the north shore of Lake Ontario. The Oak Ridges Moraine and Niagara Escarpment dominate this landscape and ensure strong base flow to streams, which historically provided valued salmonid fisheries for the early settlers in the region (Figure 1). Data were collected 1995–2002 by several agencies using methods described in Stanfield et al. (1997). Sites were selected using multiple stratified random designs. Several studies covered the entire ecoregion. Most studies were stratified based on a measure of stream size. Sampling intensity and the types of data collected (not all methods were applied at all sites) within each stratum were designed to meet the desired precision of each study. Once the study design was determined, sites were randomly selected within each stratum. Sites were a minimum of 40 m long, with boundaries at crossovers (i.e., the location where the thalweg is in the middle of the stream) (Stanfield et al. 1997). In streams in our study area, this site length provides a reliable measure of fish biomass (Jones and Stockwell 1995), species richness (L. W. Stanfield, unpublished data), and instream habitat (Stanfield and Jones 1998). This design enabled many more sites to be sampled in a day and provided an opportunity to develop a more robust estimate of fish assemblages within an entire stream than if a single long site (e.g., 40 bankfull widths; Lyons 1992) were sampled.

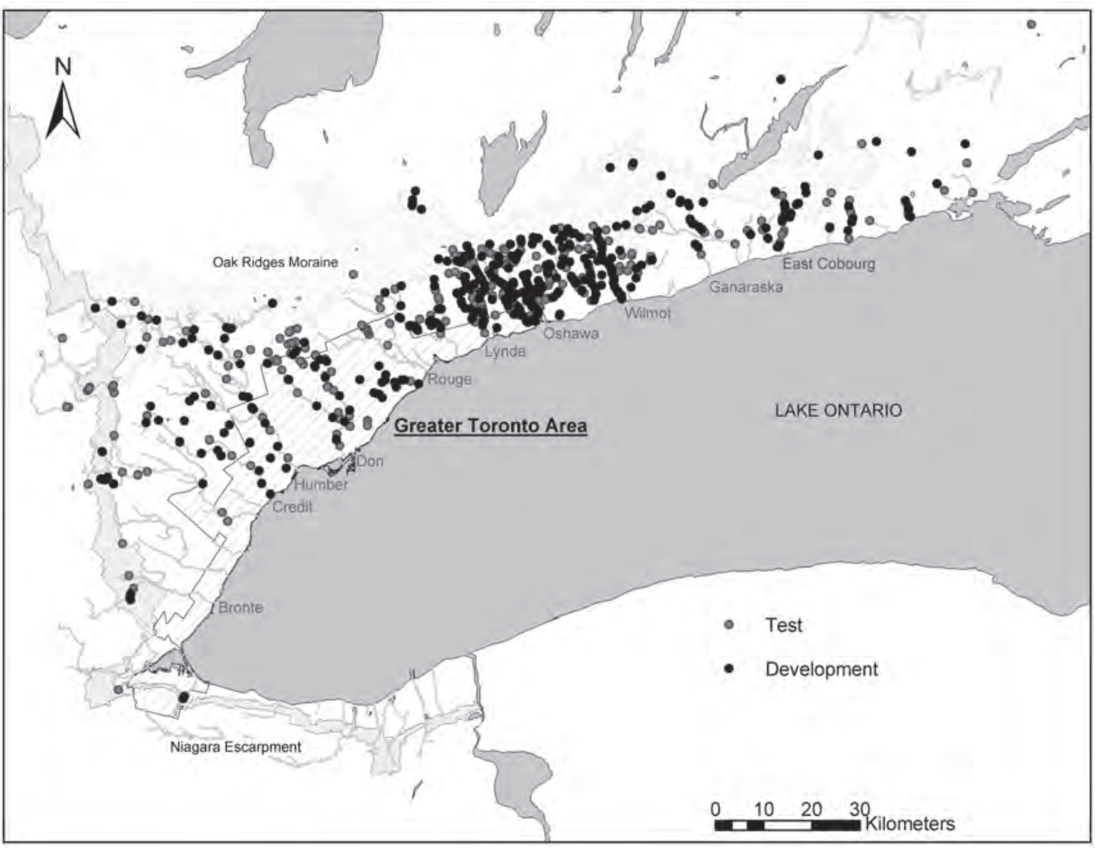

Figure 1. Major features of the study area and distribution of sample sites.

Fish Assemblage Data

Fish were collected by single-pass electrofishing at 721 sites, without bias, for size or species. Effort expended per site varied (4–15 s/m^2), but was sufficient to provide comparable assemblage measures at each site (Stanfield, unpublished data). All fish were immediately weighed, identified to species, and released, except those kept for laboratory identification. Mottled sculpin *Cottus bairdii* and slimy sculpin *C. cognatus* and American brook lamprey *Lampetra appendix* and sea lamprey *Petromyzon marinus* were inconsistently identified and were therefore combined by genus for this analysis.

Benthos Assemblage Data

Benthos were sampled at 583 sites using a modified version of Plafkin et al. (1989). Two benthos samples were collected from within crossovers, using a 2-min stationary kick-and-sweep method, with 1-mm mesh, over approximately 1 m^2. Organisms were picked from sampling trays until at least 100 individuals were obtained for each replicate or the entire sample was processed. Benthos were identified to major taxonomic groups (see Table 1).

Instream Habitat Data

Instream habitat data were collected from 578 sites using a point-transect survey design. Depending on stream width, from 10 to 20 transects (>3 m width = 10) were established at regular spacing within each site and then 2–6 observation points (>3 m width = 6) were identified on each transect. This design explained 90% of the variability in instream habitat from streams within this study area (Stanfield and Jones 1998). At each observation point (total of 40–60/site), depth, substrate, and hydraulic head were measured. Hydraulic head was used as a surrogate for stream velocity and represents the height water climbs a ruler held at right angles to flow. Cover and maximum particle size (largest present) were measured within a 30-cm ring that was centered on each observation point. Cover (converted to percent) was an object 10 cm wide on its median axis that intersected a 30-cm ring, centered on each observation point. Percent fines (particles ≤ 2 mm) and the D16, D50, and D85 were determined from substrate particle size measurements. Proportions of 16 categories of morphological features were determined based on the classes of depth (10, 60, 100 and >100 cm) and hydraulic head (<3, 4–7, 8–17, and >17 mm) at each observation point. A measure of channel homogeneity was determined by summing the proportion of a site within one category of the dominant morphologic feature at the site. The width-to-depth ratio was determined by averaging the sum of the wetted width to depth ratios of each transect.

Four metrics were calculated to provide measures of channel stability (Table 2). The vulnerability to erosion for streambanks was determined at each bank intercepted at a transect. Field measurements of bank angle, bank material, presence of undercuts, rooted vegetation

Table 1. Benthos taxa identified, and adjusted ratings used to calculate the Hilsenhoff biotic index score. Ratings based on a review of the original ratings of families common to southern Ontario.

Taxa	Rating	Taxa	Rating	Taxa	Rating
Hydracarina	6	Amphipoda	6	Simuliidae	6
Oligochaeta	8	Isopoda	8	Other Diptera	5
Hirudinea	8	Chironomidae	7	Tipulidae	3
Hemiptera	5	Coleoptera	4	Megaloptera	4
Anisoptera	5	Zygoptera	7	Trichoptera	4
Gastropoda	8	Pelecypoda	8	Ephemeroptera	5
Decapoda	6	Plecoptera	1	Ostracoda	7

Table 2. Criteria used to score metrics used to create an overall rating of channel stability for each site. Results from field surveys derived summary statistics for each metric. Ratings were developed based on the relative importance of each metric as a measure of channel stability and thresholds for each metric were based on results of local geomorphic studies (John Parish, Parish Geomorphic, personal communication). Final score for channel stability was the cumulative ratings from the four metrics

	Ratings			
Metric	0	0.1	0.2	0.3
Width/depth	>60	>40 ≤ 60	>20 ≤ 40	<20
Bank stability	≤0.4	>0.4 ≤ 0.59	>0.6 ≤ 0.79	>0.8
Sediment sorting	≤70	> 70 ≤ 120	>120	
Sediment transport	>36	>12 ≤ 36	≤12	

and riparian vegetation type were interpreted with a dichotomous key. Bank height information and substrate size at four horizontal distances (0, 0.25, 0.75, and 1.5 m) from the stream edge were used to determine whether the bank angle exceeded 45° and consisted of erodible material. Undercuts more than 5 cm deep were recorded, and the percent of rooted vegetative cover in the first 1 m of bank was measured by counting the number of squares in a grid occupied by live vegetation. Finally, the dominant vegetative type in a 2-m^2 grid at the intersection of each transect was recorded. Bank ratings ranged from 0.2 (e.g., undercut present and no forest cover present) to 1.0 (e.g., no undercut present, bank angle less than 45% and greater than 70% of squares with root cover). The coefficient of variation for the maximum particles was used as a measure of sediment sorting at each site. Sediment transport potential was estimated by dividing the D50 for the point and maximum particle sizes. Channel stability for each site represented the cumulative ratings for the four metrics (Table 2).

Stream Temperature Data

Water temperatures were recorded at 622 sites, between 1600 hours and 1700 hours, during low-flow conditions (mid-July to mid-September), when the daily air temperature exceeded 24°C for three consecutive days. We standardized stream temperatures for each site as follows. The observed water and air temperatures were used to classify each site as either cold, cool, or warm using the nomogram developed by Stoneman and Jones (1996). The appropriate regression line (Table 3) was used to predict the stream temperature at an air temperature of 30°C. Finally, the difference between the observed and predicted temperature was added to the predicted temperature at 30°C to obtain the standardized stream temperature at 30°C for each site. For example, if a stream temperature of 25°C was determined at an air temperature of 27°C, it would be classified as warm. This site is 1.2°C warmer than predicted from the algorithm for a warmwater stream (Table 3), and therefore, the standardized stream temperature would be 26.7°C:

$$\text{standardized temperature} = (30*0.555 + 8.838) + (25–23.8). \quad [1]$$

Table 3. Regression parameters used to predict water temperatures from air temperatures for reference stream types. Sites were assigned a reference class based on the lowest deviation from the observed and the predicted water temperature for the air temperature on the day of collection.

Reference Class	Slope	Constant
Cold	0.251	7.513
Cool	0.583	3.497
Warm	0.555	8.838

Landscape Data

Landscape variables were generally derived for the total catchment, through use of a geographical information systems application, following Stanfield and Kuyvenhoven (2003). Measured attributes from a 1:10,000 DEM with 25-m resolution included drainage area, link number, elevation, stream length, and site slope (determined from elevations at 100 m up and downstream of each site). The Ontario Ministry of Natural Resources developed a land cover GIS layer at 1:10,000 that assigns 1 of 28 classes to each 25-m pixel. Land classifications were based on interpretation of Landsat imageries collected from 1995 and 1996. From this classification, we quantified the amount of forest, urban, pasture, intensive agriculture (row crops and orchard), and water (lake, river, and wetland) in each catchment.

The area covered by each class of quaternary surficial geology (1:250,000) (Ontario Geological Survey 1997) was determined for each catchment. A base flow index (BFI) for each site was derived from relationships demonstrated by Piggott et al. (2002) between this index and base flow (Table 4). The BFI was calculated by summing the ranked percentage of each quaternary surficial geology unit for each catchment:

Table 4. Baseflow index (BFI) ratings for quaternary geology classes from the Ontario geological survey (1997) (Source: Piggott et al. 2002).

Geology type	BFI rating
Bedrock (Paleozoic)	0.4
Tavistock Till (Huron - Georgian Bay lobe)	0.29
Port Stanley Till (Ontario - Erie lobe)	0.27
Newmarket Till (Simcoe lobe)	0.43
Wentworth Till (Ontario - Erie lobe)	0.68
Kettleby Till (Simcoe lobe)	0.38
Halton Till (Ontario - Erie lobe)	0.39
Clay till	0.28
Till	0.4
Glaciofluvial ice-contact deposits	0.67
Glaciofluvial outwash deposits	0.77
Glaciolacustrine deposits	0.14
Glaciolacustrine deposits	0.77
Fluvial deposits	0.38
Organic deposits	0.35

$$\text{BFI per site} = \Sigma_{ij}(\%\text{geology type}_i{*}\text{BFI rating}_i)_j \qquad [2]$$

Impervious ratings for each land-use/land-cover category were selected based on an understanding of how closely the land-cover categories related to published ratings. Intensive agriculture and pasture lands are typically both rated low and similarly (i.e., 0.02, NCDE 2002, 2003; and 0.094, Prisloe et al. 2001). We chose to split these categories and rate intensive agriculture higher (0.10) than pasture lands (0.05) because in our study area, intensive agriculture often involves tile drainage and compaction from heavy machinery. Urban land ratings vary considerably depending on the detail available from the base data: 0.23–0.86 (NCDE 2002), 0.10–0.38 (NCDE 2003), 0.12–0.51 (Prisloe et al. 2001), and 0.20–0.95 (Arnold and Gibbons 1996). We chose a conservative rating of 0.20 for urban lands because of the coarseness of our land-use/land-cover data. Forested lands (0.01) and water/wetlands (0.0) were both given low ratings.

A rating for catchment percent impervious cover (PIC) was estimated as the sum of the products of percent cover by each land-use/land-cover class and the associated impervious rating of each class. An upstream catchment area with 25% water, 25% intensive agriculture, and 50% urban area, for example, would have a PIC of 12.5 (i.e., $0 \times 25\% + 0.1 \times 25\% + 0.2 \times 50\%$).

Data Analysis

The objective of the data analysis was to determine if attributes of fish and benthos assemblages and instream physical habitat characteristics were related to PIC, after accounting for natural landscape influences. Fish assemblage metrics included total biomass (g/m^2) and species richness, while a modified Hilsenhoff biotic index (HBI; Hilsenhoff 1987) and taxa richness were calculated for the benthos assemblage. Both benthos metrics were based on higher level taxonomy, and as such, the HBI ratings were assigned based on a review of the original scores for lower taxa developed by Hilsenhoff (1987; Table 1).

Correspondence analysis (CA) was used separately for fish and benthos assemblages to evaluate assemblage composition. Correspondence analysis ordination calculates a set of "synthetic" variables (axes) that best explain variations in taxa abundances across samples. Calculation of sample and taxa scores on the first ordination axis is done by iteratively estimating the weighted-average sample scores and the weighted average taxa scores. For the first iteration, axis scores are arbitrarily assigned to each taxon. For each sample, the procedure determines the weighted-average axis score, which is the average of the taxa scores weighted by the abundances of each taxon. The next iteration produces new weighted average axis scores for the taxa, calculated from the sample scores. The iterative procedure continues until there is little change in the sample and taxa scores. Estimation of second and third ordination axes follows a similar routine, except that the sample scores of additional axes are orthogonal (uncorrelated) with the first and subsequent axes. Sample scores in CA are usually scaled to a mean of zero and standard deviation of 1 (ter Braak 1992). The distribution of samples in a CA diagram indicates the relative similarities and differences in composition based on taxa abundances. Sites with similar scores have taxa in similar proportions. The scatter diagram for taxa portrays the dispersion of taxa along the theoretical variables (axes). Thus, a sample with an axis-1 score of 2 would be dominated by taxa that also had axis-1 scores close to 2. With CA, the configuration of ordination diagrams tends to be sensitive to rare taxa (Gauch 1982). Therefore, we retained for analysis only those taxa found in more than 5% of sites.

Canonical correspondence analysis (CCA; ter Braak 1992) was used to illustrate how fish and benthos assemblages varied with landscape attributes. Canonical correspondence analysis is an extension of CA, except that the ordination of the response (i.e., fish or benthos assemblage) is constrained to a set of predictor variables (i.e., landscape features). As with CA, CCA was conducted separately for fish and for benthos. The method is commonly used in ecological studies of this nature and has been used to demonstrate fish–landscape relationships (Kilgour and Barton 1999; Wang et al. 2001, 2003). Bi-plots of taxa and environmental variable scores indicate general associations between taxa and environmental conditions.

We used backward stepwise multiple regression to construct empirical models that relate instream biophysical responses to landscape variables. Predictor landscape variables included catchment area, stream slope, and base flow index (BFI). Area was selected because it is a measure of stream size and provides a coarse estimate of the amount of water or space available to fish and benthos. Catchment areas varied considerably (seven orders of magnitude) and were $\log_{10}$ transformed. Slope was selected because it is a major factor determining flow velocity and, together with area, provides a measure of stream power. The BFI was selected because it reflects the water permeability through surrounding soils (Piggott et al. 2002). In addition to these primary landscape variables, multiple regression models also included PIC. Predictors were retained in this backward stepwise regression when they accounted for significant amounts of variation in the response variable (at $P < 0.05$, typically much lower). The fish assemblage variables included biomass, species richness, and site scores for the first CA axes. The benthos assemblage variables included richness, HBI, and site scores from the first two benthos CA axes. Instream habitat variables included average stream width, width:depth, proportion stable banks, stability index, $D50_{point}$, $D50_{max}$, sorting index, sediment transport, homogeneity index, and standardized stream temperature.

Percent impervious cover and BFI scores covaried, with a correlation typically around 0.5, depending on the specific data set. Percent impervious cover was more strongly related to biotic responses than were BFI scores in some cases.

Three sets of models were, therefore, constructed to help us understand how much variation in biophysical responses was solely attributable to PIC. The first model included all possible predictors (including their squared terms to take into account possible curvilinear relationships). The second model included only the primary landscape variables (with their square terms) and excluded PIC. The second model, therefore, demonstrated the variation attributable to the primary landscape variables. The third model related the residual variation from model 2 to PIC (and the squared term). The variation accounted for in model 3 represented the variation attributable to PIC alone.

Model Validation

Prior to constructing these models, data were split into calibration and validation sets. Sites available from each data set (i.e., fish, benthos, and habitat) were randomly selected after first stratifying the data by quaternary catchment and stream order (see model outputs for number of sites used). We applied two approaches to validate the models. First empirical models were used to estimate expected biotic index values or instream habitat features for each validation site, and comparisons were made following the approach of Carr et al. (2003). Differences in precision between calibration and validation data sets were tested using an F-ratio of residual mean squares. The slope of the relationship between observed and predicted index values was also determined, as was the probability that the slope was significantly different from one, indicating that the model did not fit the validation set. The minimum, maximum, median, and mean of the residuals for the validation data, as well as the probability that the residuals were different from zero were determined. A nonzero mean residual implies that the model from the calibration data were poor. Additionally, we were concerned that the power in our data sets, due to the large number of sites, could result in differences in slope due to this factor alone; therefore, we also plotted the data and explored whether patterns and trends were similar between the calibration and validation data sets.

RESULTS

There was considerable contrast in the distribution of sites and level of catchment development (Figure 1; Table 5). Catchments included those that were principally forested (usually smaller headwater sites) and others with high percent agriculture or urbanization. Forested catchments tended to occur in headwater areas, on morainal deposits, with high-porosity soils, while urban areas tended to co-occur with larger catchment areas and lower porosity soils (i.e., clay till plains).

Table 5. Minimum, maximum, and median values of landscape conditions in the data set.

Variable	Minimum	Maximum	Median
Catchment area (km^2)	0.018	873.3	17.8
Stream order	1	7	3
Site elevation (masl)	75	440	176
Precipitation (mm)	775	975	875
% high porosity soils	0	100	25
% moderate porosity soils	0	100	< 1
% low porosity soils	0	100	61
% slope (100 m up and downstream)	0	10	<1
% water and wetlands	0	41	<1
% forest	0	98	24
% pasture	0	68	12
% crop (intensive agriculture)	0	100	52
% urban	0	100	0

Fish Assemblages

We collected 64 fish species; 43 were present in less than 5% of the calibration sites. Of the remaining species, eastern blacknose dace *Rhinichthys atratulus* was found at 73% of all sites with a mean biomass of 67 g/100 m^2 (Table 6). Sculpins *Cottus* sp., creek chub *Semotilus atromaculatus*, brook trout *Salvelinus fontinalis*, brown trout *Salmo trutta*, rainbow trout *Oncorhynchus mykiss*, and white sucker *Catostomus commersonii* were commonly occurring and abundant. Total fish biomass was 1.0–7,000 g/100 m^2, and the number of fish species per site varied from 1 to 14.

The CCA illustrated four fish assemblage clusters (Figure 2). Axis 1 separated sites where salmonids were dominant from those with a more diverse mix of fishes where salmonids were a smaller component of the assemblage. Sites with abundant salmonids tended to have higher forest cover and BFI ratings and lower PIC, whereas sites lacking salmonids tended to have less forest cover, more urban area, lower BFI, and higher PIC. The second CCA axis separated sites with salmonid assemblages into those with brook trout from those with other salmonid species. Brook trout tended to occur in sites with smaller catchments and greater elevations and slopes, while brown trout and rainbow trout tended to occur in sites with larger catchments and lower elevations and slopes. The second axis also separated sites with nonsalmonid taxa. Sites in smaller catchments supported species such as northern redbelly dace *Phoxinus eos*, fathead minnow *Pimephales notatus*, and brook stickleback *Culaea inconstans*, while darters *Etheostoma* spp. and rock bass *Ambloplites rupestris* were more common in sites with larger catchments.

Percent impervious cover was a significant predictor of each of the fish assemblage responses, regardless of whether we modeled raw fish metrics or their residuals after accounting for landscape features (Table 7). Relationships between PIC and fish assemblage metrics were less apparent with residuals than with the original variables (Table 7; Figure 3) because PIC and BFI were related (Figure 2). Biomass weakly decreased linearly with PIC (Figure 3). Species richness was highest at more than 10 PIC, regardless of whether the full or residual model was used (Figure 3). However, there was a weak bimodal pattern, where two sites with more than 15 PIC had species richness comparable to areas with less than 10 PIC. Scores of fish CA Axis 1 increased with PIC. The models for CA Axis 1 predict the presence of salmonids in streams with low PIC and an absence of salmonids in streams with high PIC (Figure 2; Table 8). Scatterplots

Table 6. Distribution and biomass of common species[a] used in the model development.

	Mean log biomass (g/100 m^2)				
% of sites with each taxa	0–0.5	0.5–1.0	1.0–1.5	1.5–2.0	2.0–2.5
5–10	BST	CHS, FTD	COS, LAM, NRD	ROB	
10–20			RBD, PKS, BNM,	CSH	
20–30			FHM		BKT
30–40			JOD		BNT
40–50				LND	RBT
50–60				SCU	CRC, WS
60–70				BND	

[a] BST = brook stickleback *Culaea inconstans*; CHS = Chinook salmon *Oncorhynchus tshawytscha*, FTD = fantail darter *Etheostoma flabellare,* COS = coho salmon *O. kisutch*, LAM = lamprey family Petromyzontidae, NRD = northern redbelly dace *Phoxinus eos*, ROB = rock bass *Ambloplites rupestris*, RBD = rainbow darter *E. caeruleum*), PKS = pumpkinseed *Lepomis gibbosus*, BNM = bluntnose minnow *Pimephales notatus*, CSH = common shiner *Luxilus cornutus*, FHM = fathead minnow *P. promelas*, BKT = brook trout *Salvelinus fontinalis*, JOD = Johnny darter *E. nigrum*, BNT = brown trout *Salmo trutta*, LND = longnose dace *Rhinichthys cataractae*, RBT = rainbow trout *O. mykiss*, SCU = sculpin family Cottidae; CRC = creek chub *Semotilus atromaculatus*, WS = white sucker *Catostomus commersonii*, BND = eastern blacknose dace *R. atratulus*.

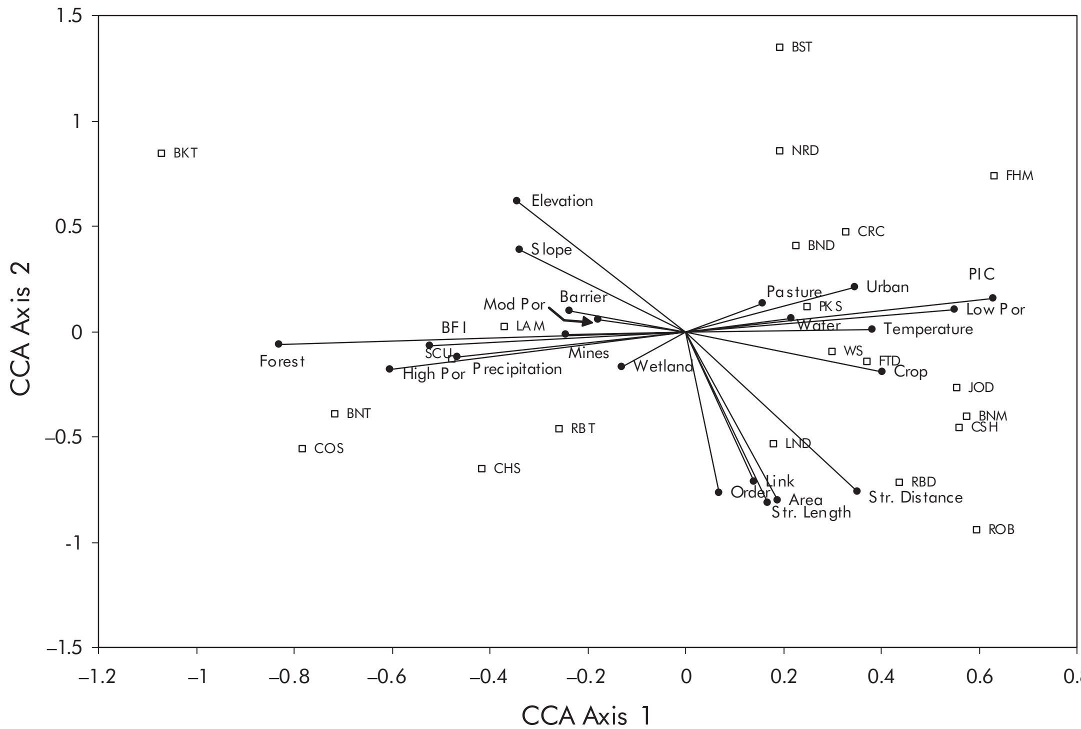

Figure 2. Relationship between landscape and fish assemblage composition as determined through canonical correspondence analysis (CCA). Species acronyms are defined in Table 6. Por = porosity of soils; str = stream; PIC = percent impervious cover; BFI = baseflow index.

Table 7. Regression models relating fish assemblage metrics to landscape and percent impervious cover (PIC) variables. There were three models for each response. Model 1 (the full model) relates the best landscape and PIC predictions to the response. Model 2 (reduced landscape model) relates the best landscape predictors (not including PIC) to the response. Model 3 relates the residuals from Model 2 to PIC.

	Response variable								
	Log fish biomass			Fish taxa richness			Fish canonical axis 1		
Model parameters	1	2	3	1	2	3	1	2	3
Constant	–3.841	–5.075	0.129	–6.582	–3.741	–1.801	–1.625	2.465	–1.243
Area	1.866	1.667							
$Area^2$	–0.125	0.113		0.189	0.184				
Slope		0.125			–0.298		–0.243	–0.499	
$Slope^2$		–0.017					0.027	0.052	
BFI		0.051					–0.016	–0.047	
BFI^2	<–0.001	<0.001							
PIC	–0.066			0.619		0.519	0.476		0.234
PIC^2			–0.002	–0.035		–0.031	–0.016		–0.008
N	361	361	361	361	361	361	361	361	361
MSE	0.215	0.242	0.218	5.895	6.380	5.835	0.804	0.949	0.870
R^2	0.185	0.085	0.087	0.372	0.319	0.085	0.394	0.280	0.081

Note: Each metric was squared to account for possible curvilinear relationships.

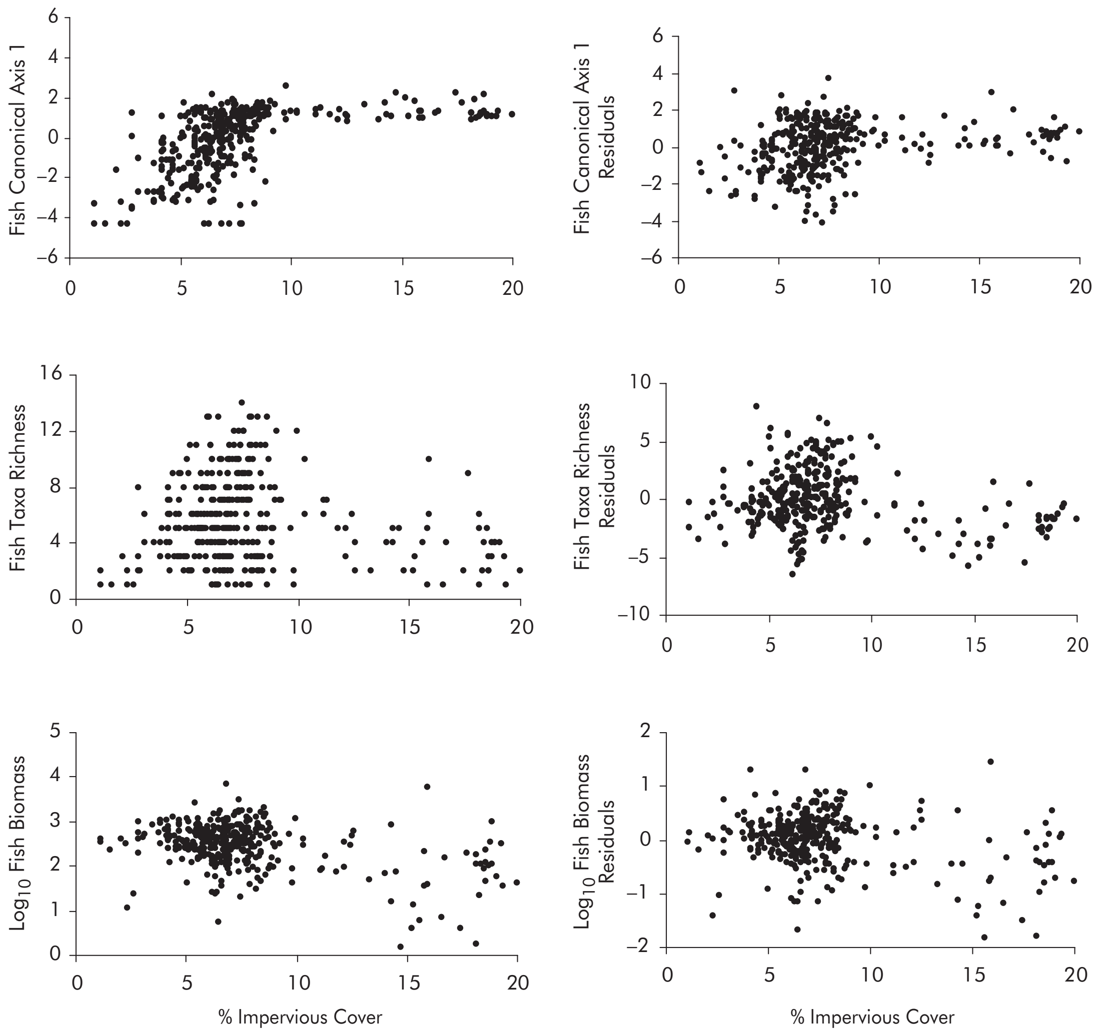

Figure 3. Relationship between percent impervious cover (PIC) and fish variables and their residuals after accounting for landscape variables.

of the raw and residual data indicate large changes in axis scores at less than 10 PIC, with smaller changes in axis scores at more than 10 PIC (Figure 3). There was, thus, a threshold at 10 PIC for CA Axis 1 scores.

Benthos Assemblages

There were 20 benthos taxa collected, including the typical sensitive groups Ephemeroptera, Plecoptera, and Trichoptera and the more tolerant groups Oligochaeta and Chironomidae. Turbellaria, Hirudinea, Isopoda, Amphipoda, Gastropoda, and Pelecypoda were also relatively common. Site taxa richness ranged from 1 to 17, and the modified HBI ranged from 3 to 8.

Tolerant taxa (chironomids, platyhelminths, oligochaetes, isopods, etc.) were generally found at sites with higher PIC and lower BFI scores. Sensitive taxa (Plecoptera, Ephemeroptera, Coleoptera) were generally found in streams with higher forest cover and BFI scores (Figure 4). A

Table 8. Regression models relating benthos assemblage metrics to landscape and percent impervious cover (PIC) variables. There were three models for each response. Model 1 (the full model) relates the best landscape and PIC to the response. Model 2 (reduced landscape model) relates the best landscape predictors (not including PIC) to the response. Model 3 relates the residuals from Model 2 to PIC.

	Response variable											
	Hilsenhoff			Benthos taxa richness			Benthos canonical axis 1			Benthos canonical axis 2		
Model parameters	1	2	3	1	2	3	1	2	3	1	2	3
Constant	8.685	10.961	–0.454	10.068	5.378		16.263	7.514		–0.279	2.783	–0.494
Area	–0.417	–0.360					–3.396	–0.671				
area^2						No sig-	0.191		No sig-			
slope	–0.171	–0.201				nificant	–0.199	–0.206	nificant		–0.094	
slope^2						predic-			predic-			
BFI	–0.017	–0.088			0.159	ors	–0.077	–0.095	tors	–0.011	–0.092	
BFI^2		0.001			–0.001		0.001	0.001			0.001	
PIC	0.092		0.061				0.042			0.102		0.066
PIC^2				–0.008								
n	332	332	332	332	332		332	332		332	332	332
MSE	0.653	0.714	0.664	4.842	5.143		0.760	0.779		0.827	0.896	0.839
R^2	0.306	0.242	0.061	0.080	0.026	N/A	0.255	0.232	N/A	0.180	0.114	0.058

Note: Each metric was squared to account for possible curvilinear relationships.

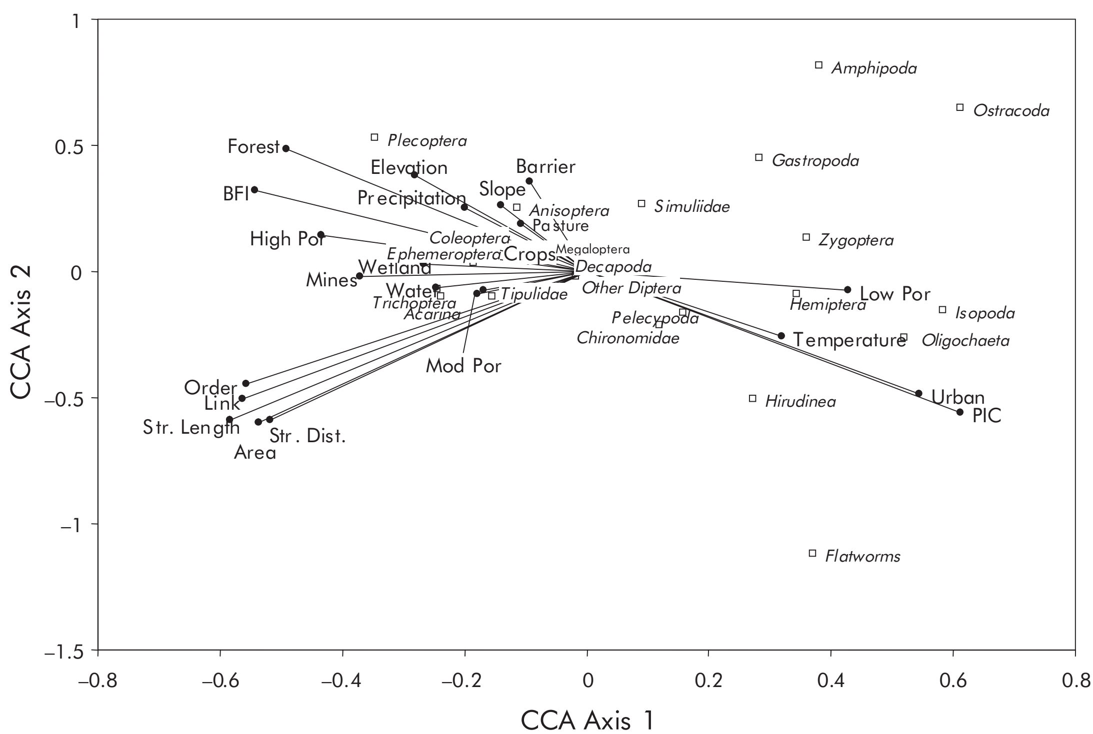

Figure 4. Relationship between landscape and benthos assemblage composition as determined through canonical correspondence analysis (CCA). Por = porosity of soils; str = stream; PIC = percent impervious cover; BFI = baseflow index.

secondary environmental gradient was apparent in the data, with amphipods, ostracods, and gastropods being more prevalent in smaller catchments.

As with fish assemblage metrics, indices of benthos assemblage composition were generally related to PIC, even after accounting for the underlying influences of natural landscape features (Table 8). The HBI predictably increased with PIC, indicating degraded conditions. Values were 4–5 for sites without development and averaged about 6 for sites with full urbanization (i.e., 20% PIC). The HBI exhibited a weak threshold response at a 10 PIC (Figure 5). The relationship between richness and PIC was statistically significant but not convincing. Benthos taxa richness was not significantly related to landscape features other than BFI (Table 8). The benthos CA Axis 1 scores increased weakly with PIC (Figure 5), but only for the full model (not the residuals), indicating that PIC may not be important for this metric. Correspondence analysis Axis 2 scores, however, did relate to PIC after removing the effects of natural landscape factors and exhibited a weak threshold response (Figure 5). Correspondence analysis axis 2 scores varied between –2 and 3.5 at PIC less than 10, and there were no values greater than 0 above PIC of 10. Sites with PIC greater than 10 contained higher proportions of mayflies, chironomids, isopods, and worms.

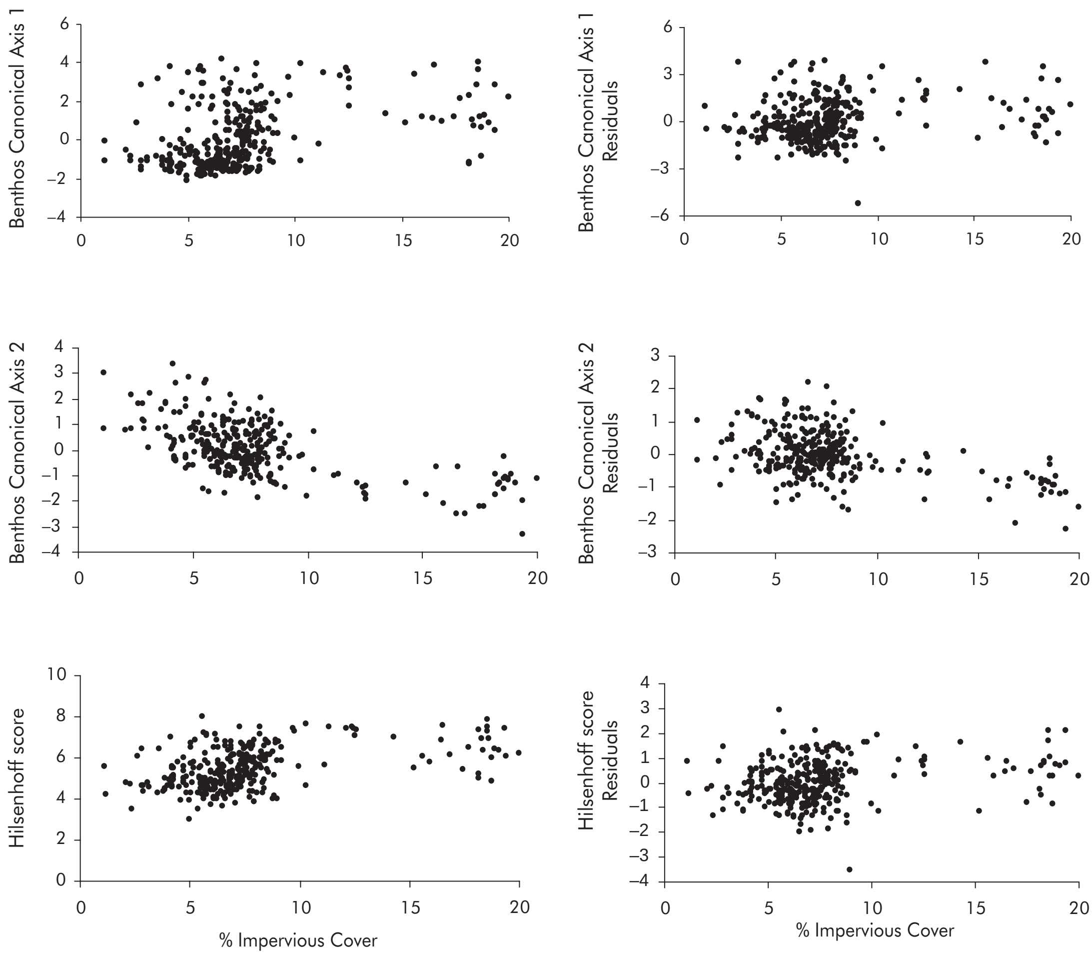

Figure 5. Relationship between percent impervious cover (PIC) and benthos variables and their residuals after accounting for landscape variables.

Instream Habitat

The standardized stream temperature, proportion of stable banks, and mean width were related to PIC, after the landscape conditions were taken into consideration (Table 9). The other physical habitat and channel stability metrics were not related to PIC.

Standardized stream temperature and mean width were the only habitat attributes to demonstrate threshold-type responses to PIC, for both the main and residual models, but these were weak relationships (Figure 6). There was a mix of cold- and warmwater sites at less than 8 PIC, and there were no coldwater sites above this threshold (Figure 6). Mean stream widths were 0.5–20 m in catchments with PIC less than 10, but narrow streams were absent in catchments with higher PIC (Figure 6).

Model Validation

Fish CA axis 1, the modified HBI, standardized temperature, and the log of the width:depth ratio had the highest model fits with landscape data (Figure 7; Table 10). The slope of the predicted values for the validation and calibration data sets differed from unity for the fish CA axis 1 (Figure 7). The validation data sets for all of the variables, however, tended to produce scatterplots that were similar to the calibration scatterplots, suggesting that the models produced were relatively robust and that the data used to construct the models were representative of the larger data sets. The lack of fit for the fish CA axis 1 is likely related to either the extreme power in our data or to other factors (not included in the model) also being important in explaining variation in fish assemblages.

DISCUSSION

There are three principal conclusions from this study. First, landscape measures accounted for significant variability in the responses of fish and benthos assemblages, instream temperature, and some instream habitat metrics. Second, PIC was a significant modifier of the fish and benthos assemblage responses, as well as temperature, width:depth, and percent stable banks, even after removing or accounting for the influences of natural landscape conditions. Third, fish and benthos assemblages were clearly altered above 10 PIC, and there were no coldwater streams above that threshold. Below the threshold, the biophysical responses indicated that change in PIC would change fish, benthos, temperature, and the percent stable banks in an incremental way. The landscape models developed here can be used to predict fish and benthos assemblages and habitat conditions, under a variety of land-use/land-cover scenarios, including an undisturbed reference state. Each of these main points is discussed below.

Natural Landscape Influence Biophysical Responses

In this study, catchment area, slope, and the base flow index were strong predictors of variation in indices of fish and benthos assemblages and stream temperature. These results were consistent with previous studies (e.g., Shaver and Maxted 1996; Richards et al. 1996, 1997; Kilgour and Barton 1999; Wang et al. 2001; Zorn et al. 2002; Wang and Kanehl 2003). As has been observed previously (Horwitz 1978; Kilgour and Barton 1999; Zorn et al. 2002), we observed a strong gradient in the fish assemblages related to catchment size. Similar to Barton et al. (1985) for southwestern Ontario and Zorn et al. (2002) for lower Michigan, we found that brook trout was generally limited to smaller catchments, while other salmonids were found in larger catchments. Stoneman and Jones (2000) provide evidence that this pattern of salmonid abundance is partly due to competition. It is also likely that this relationship relates to the location of sites relative to barriers in the catchment. The CCA of the fish assemblage (Figure 2) indicated a weak tendency for brook trout to be more prevalent upstream of barriers.

Table 9. Regression models relating indices of habitat metrics to landscape and percent impervious cover (PIC) variables. There were three models for each response. Model 1 (the full model) relates the best landscape and PIC to the response. Model 2 (reduced landscape model) relates the best landscape predictors (not including PIC) to the response. Model 3 relates the residuals from Model 2 to PIC.

Model parameters	Response variable														
	Stream temperature			Mean width:depth			Proportion stable banks			Channel stability			Mean stream width		
	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
constant	12.5	21.382	–2.683	0.727	0.962	–0.027	0.992	1.167	–0.052	2.229			–0.455	–0.256	0.123
area	0.989						–0.003	–0.030		–0.571				0.031	
area^2		0.073		0.018	0.017					0.046			0.030		
slope	–1.818	–2.104		0.082	0.027		–0.012	–0.038		0.034			–0.066		
slope^2	0.183	0.206		–0.009				0.004							
BFI				–0.019	–0.023								–0.028	–0.035	
BFI^2	–0.001	–0.001		0.0002	0.0002			–0.0001					<0.001	<0.001	
PIC	0.891		0.560				0.022		0.011						–0.036
PIC^2	–0.034		–0.022	0.0005		0.0004	–0.0008		–0.0004				0.001		0.002
n	385	385	385	370	370	370	353	353	353	353			373	373	373
MSE	10.3	10.611	10.313	0.039	0.040	0.039	0.008	0.008	0.008	0.044	NA	NA	0.491	0.051	0.048
R^2	0.278	0.263	0.023	0.357	0.332	0.022	0.082	0.077	0.010	0.069			0.607	0.590	0.048

Model parameters	Response variable											
	$D50_{point}$			$D50_{max}$			Sorting index			Homogeneity		
	1	2	3	1	2	3	1	2	3	1	2	3
Constant	–1.313	–1.130		–2.493	–2.180		0.635			2.052		
Area												
Area^2	0.040	0.040		0.067	0.066					–0.023		
Slope	0.401	0.406		0.674	0.618					–0.128		
Slope^2	–0.029	–0.029		–0.056	–0.048							
BFI		–0.010					0.049					
BFI^2				–0.0001	–0.0002		–0.0006			0.0001		
PIC	–0.0001											
PIC^2				0.002								
N	369			363			363			363		
MSE	0.345	NA	NA	1.0254	1.037	NA	0.061	NA	NA	0.866	NA	NA
R^2	0.237			0.224	0.213		0.032			0.053		

Note: Each metric was squared to account for possible curvilinear relationships. NA indicates no significant model determined.

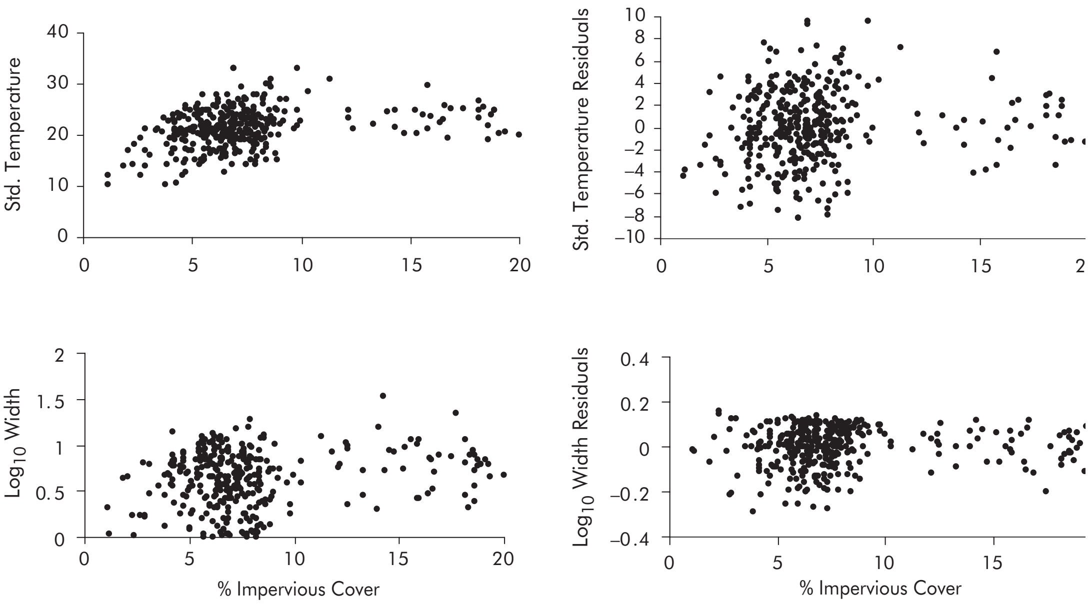

Figure 6. Relationship between percent impervious cover (PIC) and standardized temperature and mean width and their residuals after accounting for landscape variables.

In catchments with poorly drained soils, associations between catchment area and the fish fauna were not surprising. Brook stickleback, northern redbelly dace, and fathead minnow were more common in streams draining smaller catchments, while rock bass, rainbow darter *Etheostoma caeruleum,* and longnose dace *Rhinichthys cataractae* were more common in streams draining larger catchments. These associations were subtle, but have been reported before for southern Ontario (Kilgour and Barton 1999) and agree for the most part with findings from lower Michigan (Zorn et al. 2002). In lower Michigan, brook stickleback and northern redbelly dace were found in larger catchments than in our study and the difference in catchment size between where rock bass and rainbow darters were found was less distinct than what we observed. These differences are likely due to the much larger catchment size range in the Michigan study (i.e., maximum catchment sizes exceeded 10,000 km^2 compared to 873 km^2 in our study).

In this study, salmonids were generally found at sites with higher slopes. The influence of slope has been demonstrated in several other studies but notably by Wang and Kanehl (2003). Streams with greater slopes offer higher energy regimes (Rosgen 1996), higher groundwater contributions (Baker et al. 2001), and potential refuge for brook trout from migratory salmonid competitors. Catchments with higher gradients produce greater head for groundwater movement and streams with higher gradients tend to cut deeper into alluvial materials increasing the potential to intersect the water table.

The importance of surficial geology as a primary influence on fish and benthos assemblages was reconfirmed in this study. Many other studies have demonstrated the significance of surficial geology, notably Portt et al. (1989) for southern Ontario streams. In this study, we used an index of base flow to capture the surficial geology influence and found it highly predictive of fish and benthos assemblages. Though our index of base flow potential differed from others (e.g., Zorn et

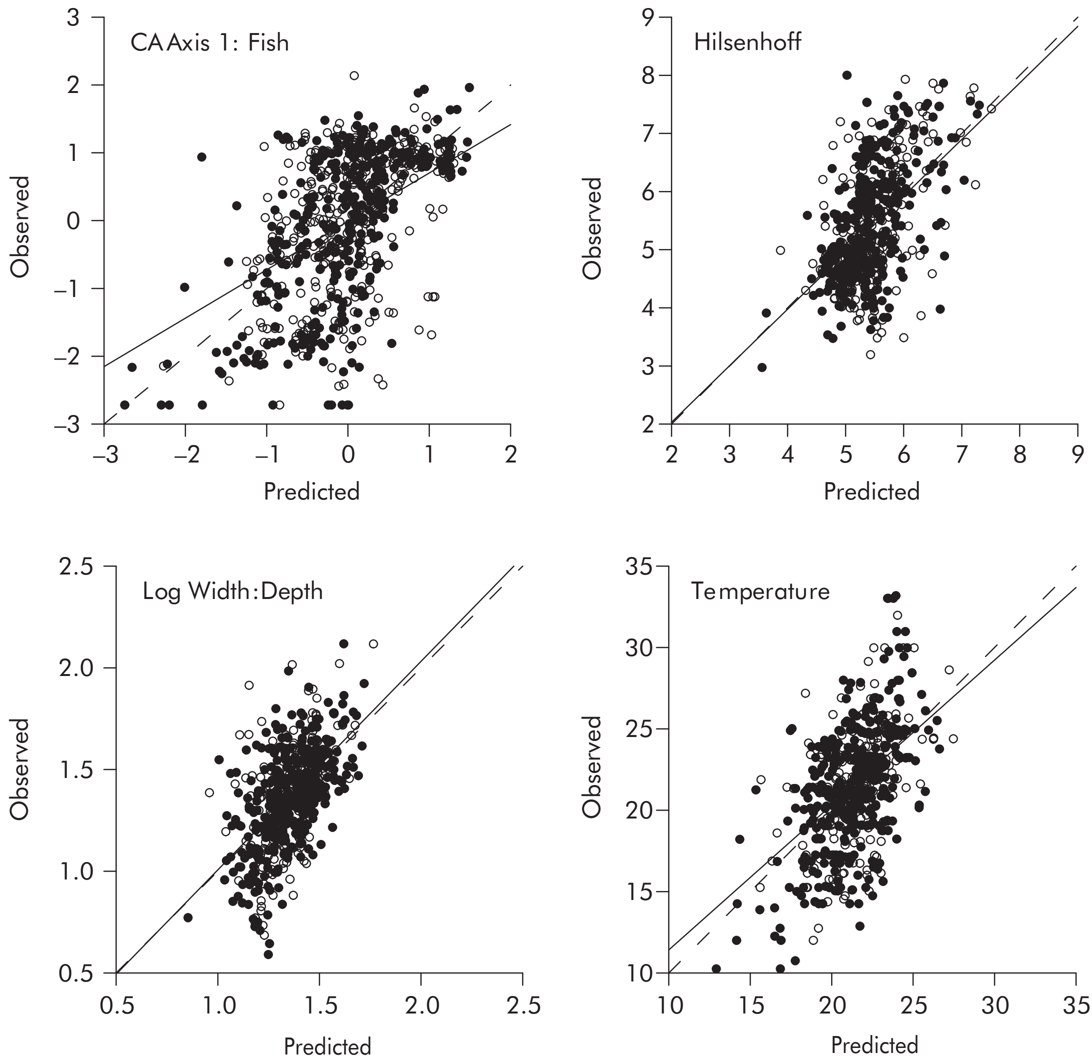

Figure 7. Relationship between observed and predicted biophysical indices illustrating the general fit of the best-fitting models. Filled circles are calibration data, open circles are validation data. The solid line is the 1:1 line of expectation (calibration data), while the dotted line is for the validation data.

al. 2002; Wang et al. 2003), the results were similar in that coldwater species were more frequently observed in streams with high base flow potential (i.e., were draining areas of high porosity glacial materials). Although our models explained relatively little variation in the response variables, the models were apparently robust and reflected what is intuitively known about the relationships between stream biophysical responses and landscape attributes. There should, therefore, be reasonable confidence in using the derived models for understanding the relationships between stream biophysical responses and landscape and land-use/land-cover conditions.

PIC Effects

Even after considering the effects of natural landscape variables (i.e., size, surficial geology/base flow and slope), there were significant variations

Table 10. Validation of the best-fitting biophysical models. Differences in precision between calibration and validation data sets was tested using an *F*-ratio of residual mean squares. The slope of the relationship between observed and predicted index values was also determined, as was the probability that the slope was one, indicating that the model fit the validation set. The minimum, maximum, median, and mean of the residuals for the validation data and the probability that the residuals were zero are also provided. A non-zero mean residual implies a bias in the validation data.

	Precision		Observed vs. predicted		Residual statistics	
Model	$F_{val/cal}$	$P_{val = cal}$	Slope (SE)	$P_{slope = 1}$	Mean	$P_{mean = 0}$
Fish canonical axis 1	1.032	0.384	0.714 (0.078)	0.0003	–0.009	0.852
Hilsenhoff biotic index	1.230	0.044	0.973 (0.112)	0.807	–0.064	0.284
Stream temperature	1.054	0.328	0.921 (0.115)	0.492	0.118	0.588
Log_{10}width:depth	1.301	0.017	1.199 (0.117)	0.090	0.017	0.282

in biophysical responses related to PIC. Metrics of fish assemblages and temperature varied with PIC between background conditions (~0–3) to highly urbanized (>10) (Figures 3 and 6). Below 10 PIC, there was considerable noise in the biological metrics, reflecting influences of other landscape variables (i.e., base flow, catchment size, slope) and local modifying factors such as riparian zones (Barton et al. 1985), adjacent land use, and instream habitat complexity. The observed relationships, however, indicate that increase in PIC will result in changes (i.e., degradation) in biological assemblages. These data also suggest that locally applied best-management practices and restoration activities are likely to be most effective when applied to streams with less than 10 PIC. Wang et al. (2006, this volume) reached a similar conclusion for Wisconsin and Michigan streams.

It was surprising that few geomorphic metrics were associated with PIC (Table 9). The measures used have been demonstrated to be precise (Stanfield and Jones 1998); therefore, the lack of association is unlikely related to measurement error. We also know that geomorphic attributes of streams respond to changes in PIC in the catchment (Leopold 1968) and that hydrologic factors are important in determining the kinds of fish and invertebrates found in streams (Zorn et al. 2002). The geomorphic variables that did vary in relation to landscape variables (including PIC) included width:depth ratio, which is a classic indicator of an urbanized stream (i.e., wider and shallower in urban areas), and percent stable banks. The other geomorphic variables (stability, $D50_{point}$, $D50_{max}$, sorting index, and homogeneity) are essentially measures of substrate. That these factors did not relate well to landscape features indicates that they may be more controlled by local factors, such as sinuosity, gradient and riparian conditions (Rosgen 1996), and potentially local soil types.

An alternative hypothesis is that our data set included an insufficient number of sites exhibiting stable geomorphic conditions. Several studies suggest that channel stability is even more sensitive to PIC than biological variables (Dunne and Leopold 1978; Booth and Jackson 1997). Further, geomorphic processes require hundreds to thousands of years to reestablish equilibrium. The study area was deforested in the 1800s and sustained serious instream modifications until reforestation and soil protection began in the 1930s (Richardson 1944). Stream morphology and stability in this study area likely reflect the historic changes in the landscape and recent development patterns (i.e., urban sprawl). Our findings suggest that more effort is required to sort historic from current impacts on channel geomorphology and to assess the value of these geomorphic metrics as indicators of overall stream condition.

The PIC Threshold

There were no apparent relationships above a 10 PIC, probably because the biological assemblages at those levels were very tolerant (consisted of thermal and pollution tolerant species). Other studies examining PIC have arrived at similar conclusions; that is, critical effects tend to occur at 8–15% PIC. In our data set, the specific threshold (i.e., the critical percentage) depended on the assumed impervious ratings of each land-use/land-cover class. Urban lands were identified where infrastructure covered more than half of a 30-m pixel, ensuring that many smaller structures (including roads) would not be classed in this category, and we also assumed a 0.2 rating for urban PIC. Assuming a less conservative impervious cover rating of 0.5 for urban lands had little effect on the location of the threshold, but did increase the dispersion of points (Figure 8). Similarly, assuming an impervious cover rating of 0.05 (versus 0.1) for agriculture halves the threshold to 5% (Figure 8). This supports our hypothesis that biological and physical conditions were influenced by the combined effects of agriculture and urbanization and that there is value in developing an overall metric of catchment disturbance such as PIC. In our study, a catchment with a PIC of 10 was 40–50% urban, 80–100% agriculture, or more frequently a combination of the two. These data then indicate that fish, benthos, and instream temperatures vary widely with incremental changes in urban area to about 40%, or 80% agriculture, above which there is little additional degradation in fish and benthos assemblages and instream temperatures.

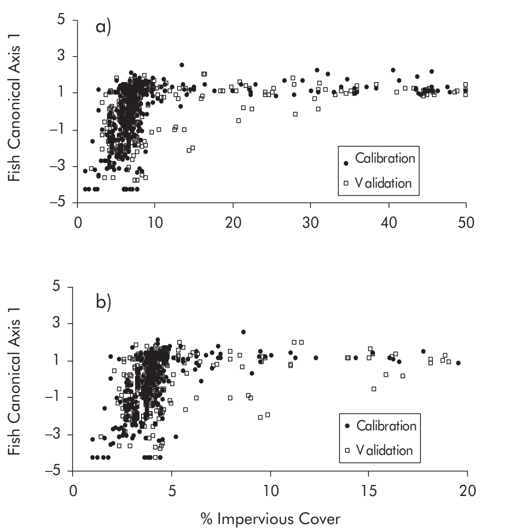

Figure 8. Relationship between fish CA axis 1 scores and two alternate scenarios for estimating imperviousness. In (a), urban lands are rated at 0.5 PIC (versus 0.2), while in (b), intensive agriculture is rated at 0.05 PIC (versus 0.1). Remaining ratings for land use/land class were unchanged from previous analysis.

The correlation between surficial geology (BFI), land use/land cover, and catchment size, as well as potential autocorrelation in our data set, should be considered when interpreting the PIC effect. The correlation (r) between BFI and PIC was about 0.55. Urban development tends to occur near Lake Ontario on clay till plains, while forested areas tend to occur in the smaller upstream tributaries on morainal deposits. Clearly, some of the variation in biological responses, temperatures, width:depth, and percent stable banks that are related to PIC are also related and thus confounded with geology. There are three factors that provide comfort that the PIC effect is real. First, while many of our sites are close together, the geographic coverage of our study area is extensive, and it is unlikely that neighboring site correlations affect our conclusions. Second, our various analyses were designed to determine how much of the residual variation in biophysical responses was related to PIC, after we accounted for the effects of the other landscape variables. That the observed patterns were consistent and that we had a large sample size of sites from catchments with high PIC provide confidence in our conclusions. Finally, many other studies have come to similar conclusions about PIC. None of the studies that we reviewed examined covariation of base flow (surficial geology) and PIC, but it is difficult to believe that all studies would be confounded, particularly those studies conducted in places like Delaware that have not experienced glaciations. Regardless, our understanding of the PIC effect in southern Ontario would benefit from additional data from sites on morainal deposits, with higher levels of PIC.

Use of Models for Hindcasting

Our models illustrate the magnitude and nature of relationships between biophysical responses and landscape features (natural and anthropogenic), and they can be used for two contrasting but interrelated purposes: hindcasting expected reference conditions and predicting future conditions assuming development scenarios. The ability to hindcast allows one to predict the biophysical makeup of a stream in the absence of development. The reference-condition approach (Hughes 1994; Bailey et al. 1998), in which regional reference sites are used to characterize acceptable biological conditions, requires least or minimally disturbed reference sites. In southern Ontario, there are no unaltered catchments. Defining reference condition, therefore, is biased toward disturbance and the definition might have to change with catchment size because there are no large catchments lacking development. Our models, however, have taken catchment size and other variables into consideration. They can therefore be used to estimate what conditions might have been in the absence of development. Current and future conditions can then be compared to the predicted historical condition, to estimate the magnitude and nature of change in condition. One limitation to the models is that they should not be used to hindcast to conditions that did not exist as part of the calibration data set. Thus, the hindcast reference condition for large catchments might not be 100% forest cover. Kilgour and Stanfield (2006, this volume) are using this approach to compare current conditions with hindcast historical conditions in the Lake Ontario study area as a means of characterizing the state of the ecosystem.

Other Considerations

We developed models for a few fish, benthos and instream habitat metrics, which likely differ from those others might have chosen. The methods we selected provide both an overview of basic features of assemblages (i.e., biomass for fish and number of taxa for fish and benthos), as well as axis scores from correspondence analysis. As a result, we are confident that the models produced here are robust and that it would be unlikely that a different conclusion would be reached with a different set of variables. Given that the results obtained for fish and benthos in southern Ontario are similar to results obtained for other parts of North America, we are confident that the patterns identified in this data set are robust.

Finally, this study employed fairly coarse measures of land use/land cover and no analysis of proximity effects (i.e., the degree to which features closer to a site influenced its condition). Future efforts will be directed at refining the relationships shown here, using finer measures of land use/land cover and proximity effects and to identifying those additional variables that contribute to explained variability in fish assemblages. In addition, we intend to explore the degree to which riparian best management practices and instream habitat complexity buffer the effects of PIC.

ACKNOWLEDGMENTS

Mike Stoneman developed the database and data queries. Randal Kuyvenhoven and Frank Kenny developed the GIS applications for extracting land-use/land-cover information. Brent Harlow assisted with data compilation and ran the GIS application. Colleen Lavender validated the location of each site on the landscape. Laurie Allin, Jeff Anderson, Jon Clayton, Marc Desjardins, Scott Jarvie, and Bernie McIntyre assisted with data collection and conceptual discussions. Helen Ball, Jason Borwick, Scott Gibson, Steve Holysh, Don Jackson, Chris Jones, Rob

McLaughlin, Nick Mandrak, Mark Peacock, Tim Rance, Keith Somers, and Chris Wilson provided advice on the approach and data analysis techniques. Gen Carr assisted with data analysis and Keith Somers reviewed the analysis. Cheryl Lewis ensured support for the project. Bob Hughes, Al Curry, and one anonymous reviewer provided valuable comments on an earlier draft. Ottawa, Toronto, Environment Canada, Michigan Department of Natural Resources and the Ontario Ministry of Natural Resources provided funding at various stages.

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