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Are Toronto's streams sick? A look at the fish and benthic invertebrate communities in the Toronto region in relation to the urban stream syndrome

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Abstract Impacts of urbanization on aquatic ecosystems are intensifying as urban sprawl spreads across the global land base. The urban stream syndrome (USS) identifies "symptoms" associated with urban development including changes in biotic communities, hydrology, water chemistry, and channel morphology. Direct relationships between road density (as surrogate of urbanization) and indicators of the USS were identified for streams in the Toronto region. Significant negative relationships were revealed between road density and biological (fish and benthic macroinvertebrate) richness, diversity, and fish Index of Biotic Integrity scores. Significant positive relationships were found between road density and tolerant fish/benthic macroinvertbrates, benthos Family Biotic Index scores, mean summer stream temperature, stream flashiness, and several water quality variables. Analysis of biological data showed that only four fish species and a reduced number of benthic macroinvertebrate families remained at the most urbanized sites. Road density was found to be a major determinant in both the fish and benthic macroinvertebrate community structure.

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Introduction

One of the major challenges of our generation is balancing the growth and development associated with our increasing population with the necessity to protect and maintain natural systems that provide numerous ecological services. Nowhere is this challenge more pronounced than in urban watersheds. Urbanization results in an increase in impervious surface cover (e.g., paved surfaces, buildings, etc.) which alters the hydrological regime of streams by decreasing the amount of water that infiltrates the soil causing an increase in the volume of surface runoff. This causes urban stream flow, both volume and velocity, to rise dramatically following rainfall or snowmelt events. Urban catchments are invariably influenced by chemical input from runoff, as well as non-chemical stressors such as land development within the catchment. Changes to the hydrology, and water quality ultimately influence the lotic ecosystem and affect the biota that inhabit it. Walsh et al. 2005 coined the moniker "urban stream syndrome" (USS) to describe the ecological degradation of streams draining urban land. Streams suffering from USS consistently have a flashier hydrograph, elevated concentrations of nutrients and contaminants, altered channel morphology, reduced biotic richness, and increased dominance of tolerant species (Allan 2004a, b; Paul and Meyer 2001; Walsh et al. 2005).

Road density was used in this paper as a surrogate for urbanization. In urban areas, road density provides a rough estimate of imperviousness (Zandbergen 1998). The amount of impervious surface cover has been linked to the overall condition of urban watersheds (see Paul and Meyer 2001) and is thought of as a key indicator of watershed health (Arnold and Gibbons 1996). Road density is a relatively easy variable to calculate compared with some of the impervious surface models currently in use by others, and it is also a concept that is easily conveyed to the general public. Road density has been used by other researchers to relate changes in the biotic community with increased human disturbance (e.g., Carnefix and Frissell 2009; DeCatanzaro et al. 2009). Road density incorporates direct impacts associated with roads (e.g., increased road salt and contaminants from fuel, tires, and vehicle components) as well as the indirect impacts from increased developed areas (residential, commercial, and industrial land uses) that are associated with a greater road density.

Many authors have described the effects of urbanization on fish and benthic macroinvertebrate (BMI) communities separately, but few have had the opportunity to study more than one community simultaneously in more than one watershed. The Toronto and Region Conservation Authority's (TRCA) Regional Watershed Monitoring Program monitors BMI, fish and aquatic habitat (water quality, stream discharge, and channel morphology) across a range of urbanization intensity. This large dataset provides the opportunity to investigate trends within the Toronto region in relation to the USS. Specifically, the objectives of this paper are to:

1. Test the prediction that road density is highly correlated with BMI and fish richness and diversity, water quality, and stream flow;
2. Explore the relationship between BMI and fish communities with in-stream habitat and landscape variables using multivariate techniques; and
3. Determine if specific BMI and fish communities are related to specific road densities.

Materials and methods

Study area

Samples were collected from nine watersheds across the 3,467 km^2 of TRCA jurisdiction (Fig. 1). All nine watersheds drain to Lake Ontario. The watersheds range in size from 27 to 911 km^2 (Fig. 1). The geology of the study area is characterized by soils comprised mainly of glacially deposited sediment (Eyles 2002). The headwaters of most of the watersheds in the region originate in the Oak Ridges Moraine; as the water flows south, it passes through till plain (unsorted glacial sediment) and then through the sand plain of glacial Lake Iroquois before it empties into Lake Ontario (Chapman and Putnam 1984). Land surface topography in the jurisdiction of TRCA varies from approximately 75 m above sea level (mASL) at Lake Ontario to a maximum of approximately 475 mASL.

Early development of the City of Toronto was centered near Lake Ontario, and along the Humber and Don Rivers (TRCA 1994; Hayes 2008). By 1861, the population of Toronto was 44,821 (Campbell 1863). The mass production of the automobile and the building of freeways facilitated the expansion into the suburbs to the north, east and west of the city core. Currently, this urban expansion continues further north as the population increases. The population in the TRCA jurisdiction was approximately 4.3 million in 2004 or 37 % of the population of the province of Ontario. In the TRCA jurisdiction, about 9 % of the region is agricultural/rural, 48 % is urbanized, 11 % is designated for future development, and 31 % is protected under legislation from most development (TRCA 2011a). Currently, only 25 % of the jurisdiction is considered natural cover (forest, wetland, meadow, successional, and beach/bluff). The proportion of these land use designations varies within each individual watershed. Generally, urbanization increases from headwaters to mouth and the upper third of the jurisdiction is rural in nature with relatively low population density. The pattern of urbanization within the jurisdiction is mirrored by the pattern of stormwater management infrastructure. Prior to 1975, few stormwater management practices were in place in the TRCA jurisdiction. Therefore, 77 % of the urban area in the TRCA jurisdiction does not have stormwater management facilities (TRCA 2011b). Older portions of the jurisdiction are also plagued by hardened channels and combined sewer overflow systems. There has been an improvement in stormwater management over time and new urban developments have implemented the most up-to-date stormwater management practices available.

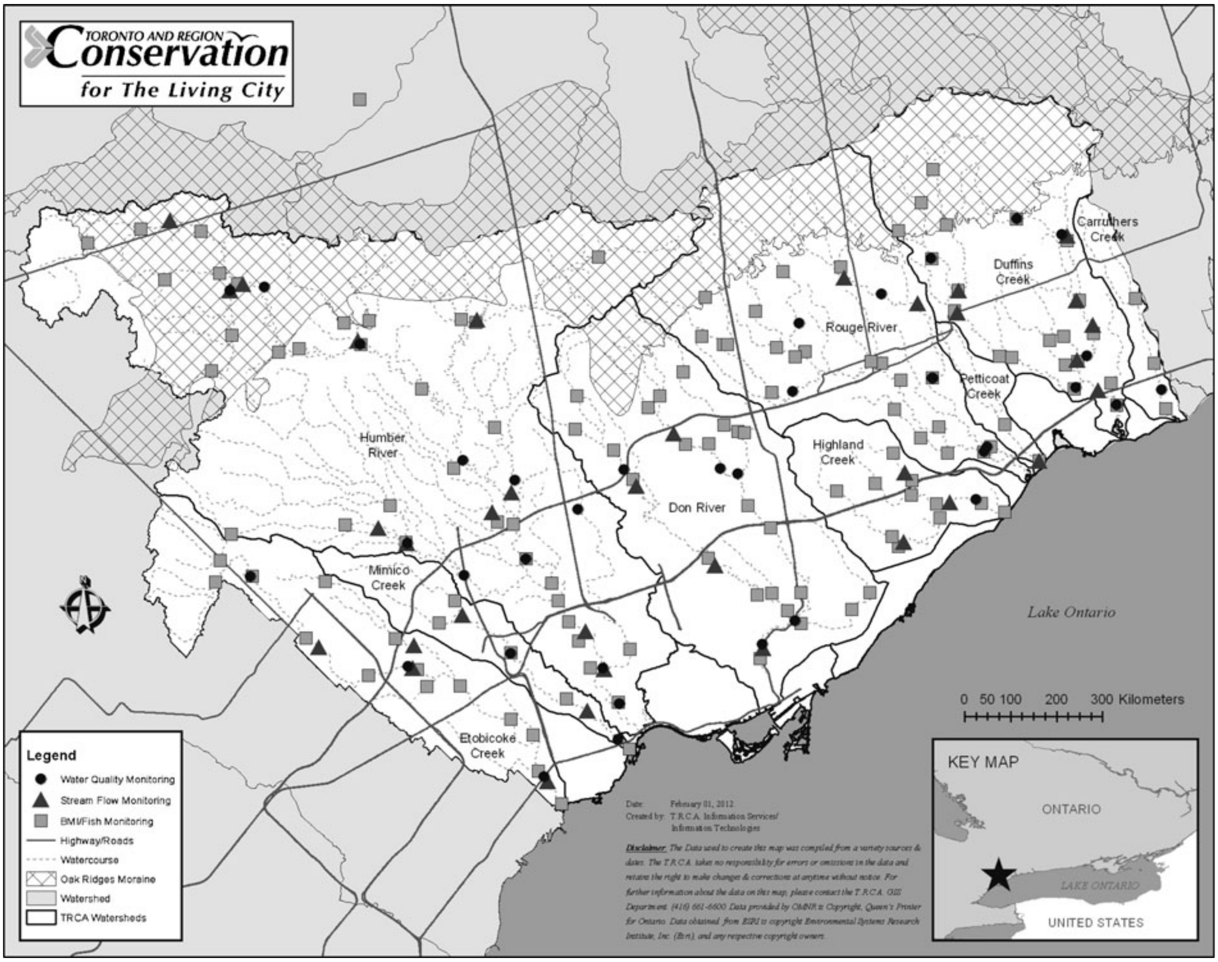

Fig. 1 Fish and BMI sampling sites within the TRCA jurisdiction

Data collection and sampling procedures

BMI and fish community collection

Biological sampling was conducted according to the Ontario Stream Assessment Protocol (OSAP; Stanfield et al. 2001). BMI samples were collected annually at 133 sites during the summer months from 2001 to 2008 (Fig. 1) using the traveling kick and sweep method. A 500-μm D-net was used along 10 to 20 transects depending on stream width. Samples from all transects were combined into a single composite sample and preserved in the field. Samples were picked and identified to family level in the laboratory. Invertebrates were identified to family (exception: Acari (subclass), Nemata (phylum), Oligochaeta (subclass), Ostracoda (class), and Turbellaria (class) but were treated as families for analytical purposes) using appropriate keys. Whole samples were identified from 2001 to 2003 and a 100+ sub-sample was identified using the teaspoon method as described in Jones et al. (2005) for the remaining years. To account for rarefaction, the whole samples were reduced to 100+ samples using a virtual Marchant box Excel macro (Walsh 1997). Fish communities were sampled with a backpack electrofisher (Smith Root model SR-12 or LR-24) using a single pass approach with no block nets. Fish were sampled at the same 133 sites as BMI on a three-year cycle (2001–2003, 2004–2006, and 2007–2009) (Table 1) between June and October. Electrofishing effort was undertaken at 7 to 15 s/m^2. Captured fish were identified to species, weighed and measured (total length), and then released. Species were identified in the field, but individuals that were difficult to identify were sent to an independent fish taxonomist for identification.

Habitat variable collection

In-stream habitat data were collected according to the OSAP protocol on a 3-year rotation in association with fish collections (Table 1). Aquatic habitat surveys were completed subsequent to the fish community surveys at the 133 biological sites. Habitat variables that were collected included average wetted width, average depth, median sediment size (D_{50}) from pebble count data, percent pool (%pool), percent riffle (%riffle), and percent glide (%glide). Water temperature data were collected on the same 3-year cycle as fish community sampling. Data were collected using digital temperature loggers installed in the streams from April to October. Loggers were programmed to record temperature at 15-min intervals. Average summer (July–August) temperatures were calculated as the average of daily minimums plus average of daily maximums divided by two (S. Allen, Environment Canada, personal communication, 2012). Water chemistry and stream flow samples were not collected at the same sites as the biological samples. Water chemistry samples were collected monthly, independent of precipitation, at 35 sites (Fig. 1). Grab samples were collected year-round from 2006 to 2010. Samples were analyzed at three different accredited laboratories for general chemistry, metals, nutrients, and *Escherichia coli*. Stream discharge data were collected at 34 sites (12 TRCA sites, 22 Water Survey of Canada (WSC) sites) across the jurisdiction (Fig. 1) from 2006 to 2008. Stage (i.e., water height) was monitored using pressure transducers and converted to discharge using rating curves developed for each station. The streams were manually gauged to capture a range of stream flows and stages that were used to create the individual rating curves. Rating curves were validated with manual measurements every year.

Determination of catchment scale "landscape" variables

ArcGIS 9.2 was used to calculate the landscape variables for the catchment upstream of each sampling site. The drainage area (in square kilometers) and stream order was calculated upstream of each site using TRCA Digital Elevation Model (DEM). Road density (in kilometers per square kilometer) was calculated using the 2007 road network provided by the Ontario Ministry of Natural Resources. Stream order was calculated according to Strahler (1957) also using the DEM. Natural cover (percent forest (%forest), percent wetland (%wetland), and percent meadow (%meadow)) was calculated based on TRCA natural cover layer which was digitized from 2002 orthophotos (TRCA 2011a).

Data analysis

Data analysis was completed using Excel 2007 (Microsoft Corporation, Redmond, Washington), JMP8

Table 1 General watershed characteristics (watersheds listed from east to west) for the TRCA jurisdiction

Watershed	Area (km^2)	Stream length (m)	Stream density (m/km^2)	Mean basin slope (%)	Mean annual discharge (m^3)	Mean road density (km/km^2)
Etobicoke Creek[a]	212	240,788	1,138	3.28	7.1×10^7	4.48
Mimico Creek[b]	77	57,672	748	3.36	2.5×10^7	8.28
Humber River[a]	911	1,136,828	1,248	5.98	21.6×10^7	2.78
Don River[b]	358	371,261	1,037	5.18	12.5×10^7	8.64
Highland Creek[b]	102	118,021	1,162	4.14	3.5×10^7	9.48
Rouge River[c]	333	428,929	1,289	4.35	9.2×10^7[d]	4.61
Petticoat Creek[a]	27	31,593	1,178	3.79	1.2×10^7	3.01
Duffins Creek[c]	287	331,218	1,156	6.14	8.0×10^7	1.90
Carruthers Creek[c]	38	48,404	1,269	3.84	1.2×10^7	1.81

[a] Fish and habitat sampled 2001, 2004, and 2007

[b] Fish and habitat sampled 2002, 2005, and 2008

[c] Fish and habitat sampled 2003, 2006, and 2009

[d] Due to stream gauge location, mean annual discharge is an underestimate

(SAS Institute, Carrey, North Carolina), and Canoco v4.5 (Plant Research International, the Netherlands).

BMI and fish community composition

A total of 133 sites monitored on three separate occasions over 9 years for fish and habitat, were used for the analysis. Corresponding BMI were sampled a minimum of 5 out of 8 years with 82 % of the sites sampled seven or more times. Biological community data were summarized by four indices that correspond to the biological "symptoms" of the USS (Table 2). Correspondence analysis (CA) was used to explore the relationships among the ecological communities. Both datasets were converted to percentage (fish based on catch-per-unit-effort) and log (X+1) transformed to improve normality. BMI families which were not found at greater than 5 % of sites and fish species not found at greater than 3 % of sites were removed from the datasets to reduce the influence that rare taxa have on CAs (ter Braak and Smilauer 1998).

Habitat data

In-stream habitat data were averaged together for the three sampling dates, and water chemistry data were averaged together for the 5-year time period to produce a single mean value per site. Because the stream flow data were collected at different time intervals, the average daily flow was calculated for each site. A spread measurement (nonparametric analog of standard deviation) was used to describe the flow responsiveness or flashiness at each site. The spread of the data was calculated as: $S = (q_{90} - q_{10})/q_{50}$, where q_x is the flow corresponding to the percentile x (Richards 1989). Normalizing by dividing by the median flow makes the spread scale independent and results in a measure that is analogous to the coefficient of variation in parametric statistics (Richards 1989).

Analyzing the response to urbanization

Road density (kilometers of road per catchment area (in square kilometers)) was used as a surrogate for

Table 2 Biological metrics used to analyze fish and BMI data along with their predicted response to urbanization

Metric		Explanation	Predicted response to urbanization
BMI	Fish		
Family richness	Species richness	Richness—reflects the diversity of the aquatic assemblage. Increasing diversity correlates with increasing health of the assemblage and suggests that niche space, habitat, and food source are adequate to support survival and propagation of many taxa	Decrease (↓)
Tolerant BMI (% Oligochaeta)	% tolerant fish	Tolerant taxa—a predominance of Oligochaeta (worms) at a site generally indicates poor water quality. Percent tolerant fish species is based on CPUE data for 9 species[a] with tolerance scores of 6 or less based on Wichert (1995)	Increase (↑)
Simpson's diversity index (SDI)	SDI	Diversity—SDI is related to the proportion of total organisms contributed by each taxon (Simpson 1949). Diversity is low when the biological community is dominated by a few taxa. High diversity indicates better environmental conditions. The index ranges from 0 (no diversity) to 1 (infinite diversity)	Decrease (↓)
Family biotic index (FBI)	Index of biotic integrity (IBI)	Summary Index—FBI is weighted index based on BMI tolerance to organic pollution. Tolerance values range from 1 to 10 and increase as water quality decreases (Hilsenhoff 1988; Bode et al. 1996). The FBI has also been shown to be sensitive to the effects of thermal pollution and some types of chemical pollution (Bazinet et al. 2010). IBI is a multi-metric measure of stream quality that uses fish as a biological indicator (Steedman 1988). IBI scores range from 9 (poor) to 45 (very good)	FBI, increase (↑); IBI, decrease (↓)

CPUE catch-per-unit effort

[a] Blacknose Dace, Bluntnose Minnow, Brassy Minnow, Brown Bullhead, Common Carp, Fathead Minnow, Goldfish, Round Goby, and White Sucker

urbanization. Correlation analysis was first completed to determine which, if any, environmental variables were correlated. The relationships between road density and metric values were examined using regression analysis. Since the orientation of the streams and the urbanization gradient both follow the same pattern with rural low-order streams to the north and urban high-order streams to the south, stream order was tested for confounding effects. This was completed using analysis of covariance (ANCOVA). Presence data were used to investigate the approximate road density at which BMI families and fish species were no longer present. The analysis assumed that if a family/species was present at a high road density, it can also survive at all lower road densities. The road density at which a family/species was assumed to be lost was the road density at which the species was last present. Thus, if a fish or macroinvertebrate species was present at a road density of 6 km/km^2 and absent at a road density of 7 km/km^2, it would be assumed that the species was lost in the transition from a road density of 6 to 7 km/km^2.

Landscape, habitat, and biological community structure

Canonical Correspondence Analysis (CCA) was used to examine the association between biological community composition and environmental gradients. CCA is a form of multivariate linear regression, and it is an ideal method to determine the relationship between species and environmental data when there are a large number of sites, species, and environmental variables (ter Braak and Verdonschot 1995). The biological data as well as the habitat data were log (X+1) transformed prior to analysis. Redundancy among the in-stream habitat data was tested using a pair-wise correlation matrix. Correlation indicates interdependence and using both variables provided no additional information about the community composition. A correlation coefficient (r) of ≥0.65 and −0.65 or less was arbitrarily chosen, and only one variable from each redundant pair was retained for analysis.

Results

BMI

A total of 965 BMI samples were collected across nine watersheds. The BMI community was composed of 111 families representing 24 orders. The most common families were Chironomidae (100 % of sites), Oligochaeta (85 %), and Baetidae (72 %). Forty-five (45) families were found at less than 5 % of the sites. In general, the Humber River, Rouge River, and Duffins Creek watersheds had highest family richness and diversity with the lowest percent Oligochaeta (Table 3). The Mimico, Don, Highland, and Petticoat watersheds showed opposite results. CA of the BMI community (Fig. 2) revealed a separation of the sensitive versus tolerant BMI families along the first axis (CAI). Pollution-tolerant families (e.g., Glossiphoniidae, Haliplidae, and Erpobdellidae) were located on the left side of the ordination. The first three CA axes accounted for 30.4 % of the variation in the invertebrate community (Table 4). Only sites from the Humber River, Duffins Creek, and Rouge River were located on the right side of the ordination. No clear separation along the second axis (CAII) was evident.

Fish

The fish community was composed of 53 species collected from 399 samples. The species belonged to 12 families (Catostomidae (7.3 %), Centrarchidae (1.3 %), Cottidae (2 %), Cyprinidae (69.8 %), Esocidae (0.001 %), Gasterosteidae (1.4 %), Gobiidae (0.4 %), Ictaluridae (0.5 %), Percidae (14.9 %), Petromyzontidae (0.5 %), Salmonidae (1.9 %), and Umbridae (0.02 %)) The most common species were Blacknose Dace (*Rhinichthys obtusus*; 89 % of sites), Creek Chub (*Semotilus atromaculatus*; 88 %), and White Sucker (*Catostomus commersonii*; 87 %). Four sites did not yield any fish during the three sampling attempts. These four sites were not used in the analysis. CA of the fish community (Fig. 2) revealed a distinct thermal gradient along the CAI. Cool/coldwater fish were located on the right side cool/warm-water fish were on the left side of the ordination. The second axis (CAII) appears to be based on sensitivity whereby the lower quadrats were fish species with higher sensitivity to perturbation while the upper quadrats contain more tolerant species. Sites in the lower right quadrat represent the undeveloped upper reaches of Duffins Creek, Humber River, and Rouge River watersheds. Sites in the lower left quadrat were in middle reaches of the Duffins Creek, Humber River, and Rouge River watersheds. The sites in the upper two quadrats were predominately in the Don River, Mimico Creek, and Highland River watersheds which are thoroughly urbanized. The first three axes of the CA

Table 3 Average biological metric results (±1 standard deviation) for the TRCA jurisdiction and individual watersheds

	Watershed	N[a]	Species richness	Family richness	Tolerant fish[b]	Tolerant BMI[c]	SDI	IBI
Fish	Jurisdiction	133	6.76±3.66		40.29±24.13		0.59±0.19	23.30±5.20
	Etobicoke	14	6.69±3.10		36.63±13.73		0.56±0.24	21.10±5.48
	Mimico	5	2.73±3.71		70.12±27.60		0.27±0.23	17.00±5.20
	Humber	34	8.56±4.03		30.50±20.12		0.67±0.14	26.30±4.02
	Don	23	3.75±1.74		51.66±25.35		0.49±0.16	19.07±3.03
	Highland	11	3.45±2.73		73.80±20.80		0.43±0.17	19.07±4.38
	Rouge	21	8.84±2.85		32.44±17.39		0.66±0.12	25.56±4.63
	Petticoat	2	7.00±1.42		70.13±10.31		0.51±0.04	23.33±3.30
	Duffins	20	8.07±2.19		29.16±17.73		0.69±0.10	25.37±4.09
	Carruthers	3	5.33±1.20		40.47±18.37		0.59±0.14	25.22±4.86
BMI	Jurisdiction	133		10.46±2.77		13.1±13.5	0.69±0.11	6.46±0.77
	Etobicoke	14		10.39±2.75		8.9±4.9	0.63±0.14	6.74±0.46
	Mimico	5		7.28±0.66		19.1±9.7	0.58±0.04	7.26±0.29
	Humber	34		11.83±1.90		10.2±10.3	0.70±0.07	6.10±0.59
	Don	23		7.94±1.65		27.5±17.9	0.63±0.12	7.26±0.76
	Highland	11		6.98±1.56		25.4±14.8	0.55±0.11	7.18±0.56
	Rouge	21		11.81±2.03		6.5±5.2	0.70±0.07	6.03±0.39
	Petticoat	2		7.99±0.22		1.9±0.1	0.55±0.08	6.37±0.26
	Duffins	20		12.56±2.24		5.6±5.4	0.73±0.07	5.85±0.49
	Carruthers	3		10.90±1.05		5.6±2.6	0.74±0.03	6.49±0.28

[a] Number of sites

[b] Based on absolute CPUE

[c] Percent Oligochaeta

accounted for 36.7 % of the variation in the fish community (Table 4).

Habitat

Stream habitat varied between watersheds in several variables (Tables 5 and 6). For example, Duffins Creek was found to have the highest %forest, lowest stream temperature, lowest spread of stream discharge, and lowest concentration of analytes. Several in-stream habitat and landscape variables were correlated and the following variables were removed from further analysis: drainage area, %pools, %glides, and %wetlands.

Urbanization and stream order

ANCOVA was used to test if stream order had confounding effect on the relationship between the dependent variables and road density. ANCOVA revealed that homogeneity of slope assumption was not violated, indicating that there was no interaction between stream order and road density for all variables in Table 7 ($p>0.05$), except for copper, width/depth ratio, and discharge. Copper showed a significant difference between stream orders when assuming a separate slopes model. This is likely due to the Lester B. Pearson International Airport. When the two sites influenced by the airport are removed from the dataset, there was no longer an interaction effect between road density and stream order. The homogeneity of slope assumption was violated for discharge and width/depth ratio as a result of the influence of sixth-order streams and the way that road density was calculated. Of the 133 sites, there were only five sites that had been classified as sixth-order streams (four sites in the lower Humber River and one site in the lower Duffins Creek). The ANCOVA revealed a significant difference between stream orders in the following variables: total fish species richness, Simpsons diversity index (fish species only), amount of tolerant fish species, IBI scores, and mean temperature and %riffles. The

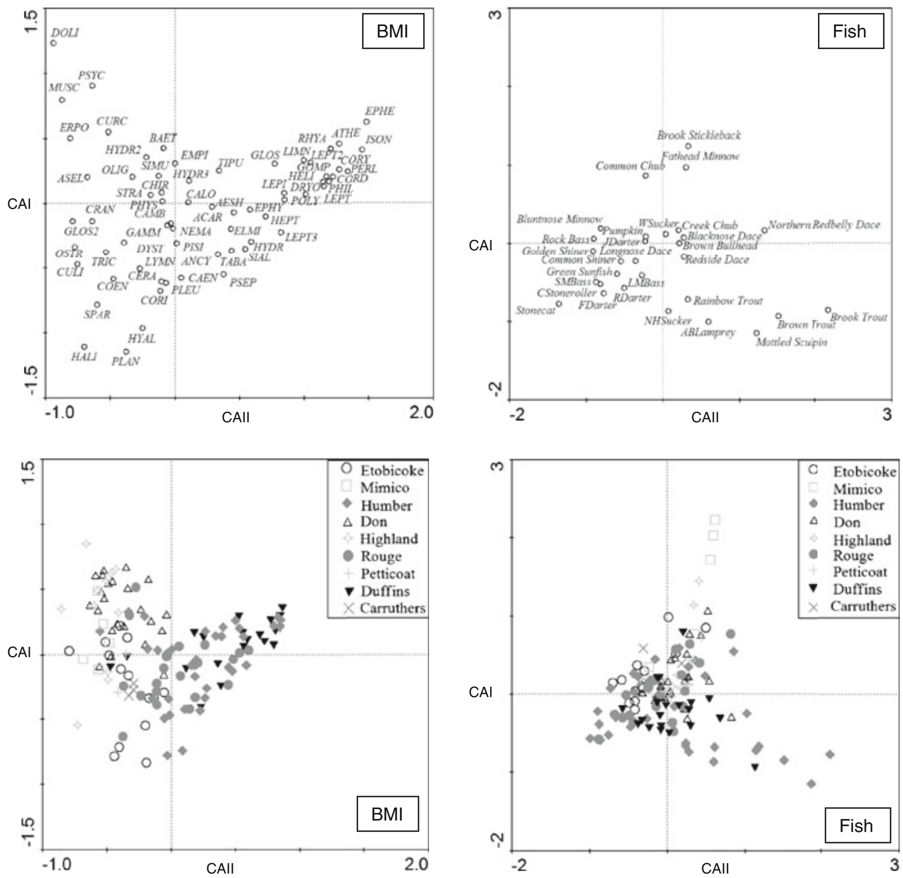

Fig. 2 Correspondence analysis results for BMI community data (family (*top left*) and site data (*bottom left*)) and fish community data (species (*top right*) and site data (*bottom right*)) for 133 sites across the TRCA jurisdiction. *ACAR* Acari, *AESH* Aeshnidae, *ATHE* Athericidae, *ANCY* Ancylidae, *ASEL* Asellidae, *BAET* Baetidae, *CAEN* Caenidae, *CALO* Calopterygidae, *CAMB* Cambaridae, *CERA* Ceratopogonidae, *CHIR* Chironomidae, *COEN* Coenagrionidae, *CORD* Cordulegastridae, *CORI* Corixidae, *CORY* Corydalidae, *CRAN* Crangonyctidae, *CULI* Culicidae, *CURC* Curculionidae, *DOLI* Dolichopidae, *DRYO* Dryopidae, *DYST* Dytiscidae, *ELMI* Elmidae, *EMPI* Empididae, *EPHE* Ephemerellidae, *EPHY* Ephydridae, *ERPO* Erpobdellidae, *GAMM* Gammaridae, *GLOS* Glossosomatidae, *GLOS2* Glossiphoniidae, *GOMP* Gomphidae, *HALI* Haliplidae, *HELI* Helicopsychidae, *HEPT* Heptagenidae, *HYAL* Hyalellidae, *HYDR* Hydrophilidae, *HYDR2* Hydroptilidae, *HYDR3* Hydropsychidae, *ISON* Isonychiidae, *LEPI* Lepidostomatidae, *LEPT* Leptophlebiidae, *LEPT2* Leptohyphidae, *LEPT3* Leptoceridae, *LIMN* Limnephilidae, *LYMN* Lymnaeidae, *MUSC* Muscidae, *NEMA* Nemata, *OLIG* Oligochaeta, *OSTR* Ostracoda, *PERL* Perlidae, *PHIL* Philopotamidae, *PHYS* Physidae, *PISI* Pisidiidae, *PLAN* Planorbidae, *PLEU* Pleuroceridae, *POLY* Polycentropodidae, *PSEP* Psephenidae, *PSYC* Psychodidae, *RHYA* Rhyacophilidae, *SIAL* Sialidae, *SIMU* Simuliidae, *SPAR* Sparganophilidae, *STRA* Stratiomyidae, *TABA* Tabanidae, *TIPU* Tipulidae, *TRIC* Tricladida, *ABLamprey* American Brook Lamprey, *CStoneroller* Central Stoneroller, *FDarter* Fantail Darter, *JDarter* Johnny Darter, *LMBass* Largemouth Bass, *NHSucker* Northern Hog Sucker, *Pumpkin* Pumpkinseed, *SMBass* Smallmouth Bass, *WSucker* White Sucker

ANCOVA showed that the 17 water quality variables did not appear to have a significant difference between stream orders when road density was statistically controlled. More importantly, the results of regression analysis and the ANCOVA showed similar relationships with road density. With the exception of copper, the magnitude of change and direction of relationship between the abovementioned variables and road density appeared to be the same within all stream orders although significant differences in the mean amount of these variables do exist between stream orders.

Response to urbanization

Many watershed studies encounter the challenge of correlation of multiple environmental variables (Van Sickle 2003). For the BMI metrics, SDI and FBI were

Table 4 Eigenvalues from the first three axes of correspondence analysis (CA) and canonical correspondence analysis (CCA) for 133 sites across the TRCA jurisdiction

		Eigenvalues	Cumulative % of Eigenvalues	Sum of Eigenvalues
BMI	CA	0.197	15.3	1.289
		0.111	23.9	
		0.084	30.4	
	CCA	0.144	45.5	1.252
		0.077	70.0	
		0.042	83.4	
Fish	CA	0.269	14.0	1.923
		0.231	26.0	
		0.206	36.7	
	CCA	0.171	32.9	2.039
		0.145	60.9	
		0.112	82.4	

correlated with tolerant BMI and family richness (R^2> 0.7) while for the fish metrics, SDI and IBI were correlated to species richness (R^2>0.7). Several water quality variables were highly correlated (R^2>0.8). Chloride, conductivity, sodium, TDS, copper, and zinc were correlated to each other. Iron and aluminum along with *E. coli* and ammonia were also correlated. Although correlated, these variables were maintained in the regression analysis to show their specific relationship with road density. Several in-stream habitat and landscape variables were correlated. The variables %pools, %glides, and %wetlands were removed from further analysis because land use sums to 100 % and therefore do not provide unique information (Allan 2004).

Habitat

Although there was no significant relationship with either width/depth ratio or D_{50} and road density, there was a significant positive relationship with road density and water temperature as well as discharge with increasing road density (p=0.0031 and 0.0153, respectively; Table 7). The opposite relationship was found with %riffle and %forest, where there was a significant decrease with an increase in road density. The spread of the stream discharge data by site had an increasing relationship with road density (R^2=0.17, p=0.015) indicating that streams in areas with higher road density experienced flashier flows (Table 7). With the exception of TSS, all water chemistry variables had a significant positive relationship with road density (p<0.05; Table 7) and sodium, conductivity, and chloride had the strongest relationship with road density (R^2=0.786, 0.783, and 0.754, respectively).

BMI and fish

All metrics followed the predicted response to urbanization (Fig. 3) and had significant relationships with road density (p<0.05; Table 7). Richness, SDI, and IBI all decreased with increasing road density while the tolerant organisms and FBI increased with increasing road density. BMI richness and the IBI had the strongest relationships with road density (R^2=0.593 and 0.404, respectively). Based on presence data, sites with road density of 11–13 km/km^2 only had four native fish species present: Blacknose Dace, Longnose Dace (*Rhinichthys cataractae*), Creek Chub, and White Sucker (Fig. 4). The greatest decrease in native fish species occurred at road density of 9–12 km/km^2 where 15 species fail to be present past a road density of 9 km/km^2.

Landscape, habitat variables, and biological community structure

The CCA biplot was based on seven environmental variables. In the CCA biplots (Fig. 5), the environmental variables are represented as vectors and taxa are represented as dots. The length of the vectors represents the importance of the variable (ter Braak and Verdonschot 1995). CCA of the BMI revealed that the community was strongly influenced by road density and %forest along axis I (CCAI) and %riffle and the width to depth ratio along axis II (CCAII). CCA of the fish community revealed that the community structure was strongly influenced by water temperature and percent forest along CCAI and stream order and road density along CCAII. The first three axes accounted for 30.4 % of the variation in the BMI data and 82.4 % of the fish data (Table 4).

Discussion

The footprint of cities can be expected to increase in the future, therefore understanding and mitigating the impacts of this change is critical. This paper looked at Toronto area streams to determine if they were suffering from the USS. The USS is characterized by

Table 5 Average habitat and landscape variables by watershed

Watershed	Width/depth[a]	D_{50} (mm)[a]	Riffles (%)[a]	Temperature (°C)[a, b]	Discharge (m^3/s)[c]	S^c	Forest (%)[a]
Etobicoke	33.42	27.31	14.5	21.18	1.44	4.54	5.63
Mimico	34.35	18.04	17.8	22.11	0.87	5.17	1.51
Humber	29.99	20.36	23.7	19.13	1.70	3.09	21.01
Don	27.93	15.83	21.9	19.66	1.80	2.75	7.65
Highland	35.73	16.35	15.2	20.33	0.85	4.17	2.36
Rouge	28.06	15.90	25.6	19.85	0.88	4.07	13.08
Petticoat	33.62	14.35	4.4	19.39	0.43	4.28	12.34
Duffins	30.61	12.17	34.8	18.91	1.30	2.40	21.17
Carruthers	12.75	2.84	18.7	21.12	–	–	14.12

[a] One hundred thirty-three sites

[b] Average based on hourly July and August temperature measurements

[c] Thirty-four sites (Etobicoke=4, Mimico=7, Humber=4, Don=3, Highland=11, Rouge=2, Petticoat=1, Duffins=2, and Carruthers=0)

geomorphical, hydrological, and biological changes to the streams. The main indicators of USS include: flashier hydrograph, elevated concentration of nutrients and contaminants, altered channel morphology, and reduced biotic richness/increased dominance of tolerance species. An attempt was also made to link some of the drivers and responses of fish and BMI communities using multivariate statistics.

Stream degradation in urban areas has numerous causes but hydrological alteration seems to be a common element suggesting that hydrological alteration is a fundamental determinant of biotic changes (Booth

Table 6 Mean values for water quality parameters by watershed for 35 sites across the TRCA jurisdiction

Analyte	Etobicoke	Mimico	Humber	Don	Highland	Rouge	Duffins	Carruthers
	N							
	3	2	11	4	1	7	6	1
Aluminium (μg/L)	400	619	405	525	250	317	273	490
Ammonia (μg/L)	56	153	84	148	96	59	51	85
Chloride (mg/L)	457	1,021	286	525	620	210	55	117
Conductivity (μS/cm)	1,855	3,607	1,343	2,132	2,091	1,118	652	876
Copper (μg/L)	5.54	9.28	3.65	5.72	4.90	2.50	1.77	2.22
Escherichia coli (CFU/100 mL)	1,428	3,279	1,675	3,579	2,652	808	295	513
Iron (μg/L)	573	788	638	827	433	419	411	597
Nickel (μg/L)	4.41	6.07	4.77	6.80	6.84	5.33	5.10	6.62
Nitrates (mg/L)	1.48	0.88	0.77	1.41	1.24	1.02	0.70	0.88
TKN (mg/L)	0.79	0.98	0.66	0.76	0.59	0.71	0.45	0.63
pH	8.3	8.2	8.3	8.2	8.3	8.3	8.4	8.3
Phosphate (mg/L)	0.02	0.02	0.01	0.02	0.01	0.01	0.01	0.01
Phosphorus (mg/L)	0.07	0.09	0.06	0.08	0.05	0.05	0.05	0.05
Sodium (mg/L)	66	642	232	312	307	121	31	65
TDS (mg/L)	514	215	1,050	1,210	1,203	640	395	522
TSS (mg/L)	19	31	24	29	14	20	24	17
Vanadium (μg/L)	1.84	2.68	1.54	1.85	1.29	1.39	1.16	1.50
Zinc (μg/L)	17.96	43.90	16.26	24.93	17.97	11.86	9.10	11.21

No samples collected in Petticoat Creek watershed

TKN total Kjeldahl nitrogen, *TDS* total dissolved solids, *TSS* total suspended solids

Table 7 Regression relationships between BMI and fish community metrics and road density (in kilometers per square kilometer) for 133 sites across a gradient of urbanization

	Metric	R^2	F	p	Equation	Sig. Rel.
BMI	Family richness	0.593	190.960	<0.001[b]	Y=13.510−0.625*RdDen	↓
	% Oligochaeta	0.288	52.975	<0.001[b]	Y=2.800+2.123*RdDen	↑
	Simpson's diversity	0.278	50.490	<0.001[b]	Y=0.748−0.016*RdDen	↓
	FBI	0.452	108.072	<0.001[b]	Y=5.721+0.152*RdDen	↑
Fish	Species richness	0.397	86.133	<0.001[b]	Y=10.048−0.674*RdDen	↓
	%tolerant fish	0.206	32.964	<0.001[b]	Y=24.884+3.258*RdDen	↑
	Simpson's diversity	0.303	55.221	<0.001[b]	Y=0.737−0.031*RdDen	↓
	IBI	0.404	85.999	<0.001[b]	Y=27.965−0.986*RdDen	↓
Water quality	Aluminum	0.127	4.498	0.042[b]	Y=284.993+22.522*RdDen	↑
	Ammonia	0.518	33.309	<0.001[b]	Y=15.800+16.043*RdDen	↑
	Chloride	0.754	95.239	<0.001[b]	Y=−59.491+85.319* RdDen	↑
	Conductivity	0.783	111.856	<0.001[b]	Y=310.131+256.391*RdDen	↑
	Copper	0.731	84.375	<0.001[b]	Y=0.906+0.658*RdDen	↑
	Escherichia coli	0.673	63.901	<0.001[b]	Y=−323.899+456.066*RdDen	↑
	Iron	0.254	10.541	0.003[b]	Y=403.453+37.857*RdDen	↑
	Nickel	0.216	8.538	0.006[b]	Y=4.246+0.261*RdDen	↑
	Nitrates	0.233	9.397	0.005[b]	Y=0.641+0.081*RdDen	↑
	TKN	0.337	15.723	<0.001[b]	Y=0.512+0.036*RdDen	↑
	Phosphate	0.278	11.965	<0.001[b]	Y=0.009+0.001*RdDen	↑
	Phosphorus	0.476	28.121	<0.001[b]	Y=0.042+0.005*RdDen	↑
	Sodium	0.786	73.363	<0.001[b]	Y=−23.495+44.096*RdDen	↑
	TDS	0.731	54.434	<0.001[b]	Y=224.511+134.570*RdDen	↑
	TSS	0.072	2.389	0.132	Y=19.166+0.924*RdDen	–
	Vanadium	0.334	15.555	<0.001[b]	Y=1.133+0.094*RdDen	↑
	Zinc	0.676	64.778	<0.001[b]	Y=4.390+2.768*RdDen	↑
Habitat	Width/depth	$2.44e^{-5}$	0.0032	0.955	Y=31.586+0.036*RdDen	–
	D50	0.005	0.7512	0.387	Y=15.58+0.399*RdDen	–
	%riffle	0.034	4.474	0.0364[b]	Y=27.884+0.995*RdDen	↓
	%forest	0.416	93.345	<0.001[b]	Y=24.029−2.204*RdDen	↓
	Water temperature[a]	0.065	9.096	0.0031[b]	Y=18.988+0.142*RdDen	↑
	S	0.170	6.560	0.0153[b]	Y=2.546+0.204*RdDen	↑
	Discharge (m^3/s)	$6.91e^{-4}$	0.0221	0.883	Y=1.298+0.0134*RdDen	–

Sig. Rel. significant relationship, *TKN* total Kjeldahl nitrogen, *TDS* total dissolved solids, *TSS* total suspended solids, D_{50} median sediment size

[a] Water temperature based on average July and August measurements

[b] Regression was significant at $\alpha \leq 0.05$

2005). Increased flashiness is one of the most consistently reported symptoms of the USS (Roy et al. 2005). A flashy stream is one that exhibits significantly increased flows immediately following the onset of a precipitation event and quickly returns to pre-event levels. Although the relationship between stream flow flashiness (S) and road density was significant, the R^2 value (0.170) was low indicating that there was a large amount of variation within the dataset. This may be due to the coarse mean daily stream discharge measurement used for the spread calculation. Baker (1989) suggested that averaging flows over a 1-day period will reduce peak flows more drastically on small tributaries than on large ones because runoff from storm events may only

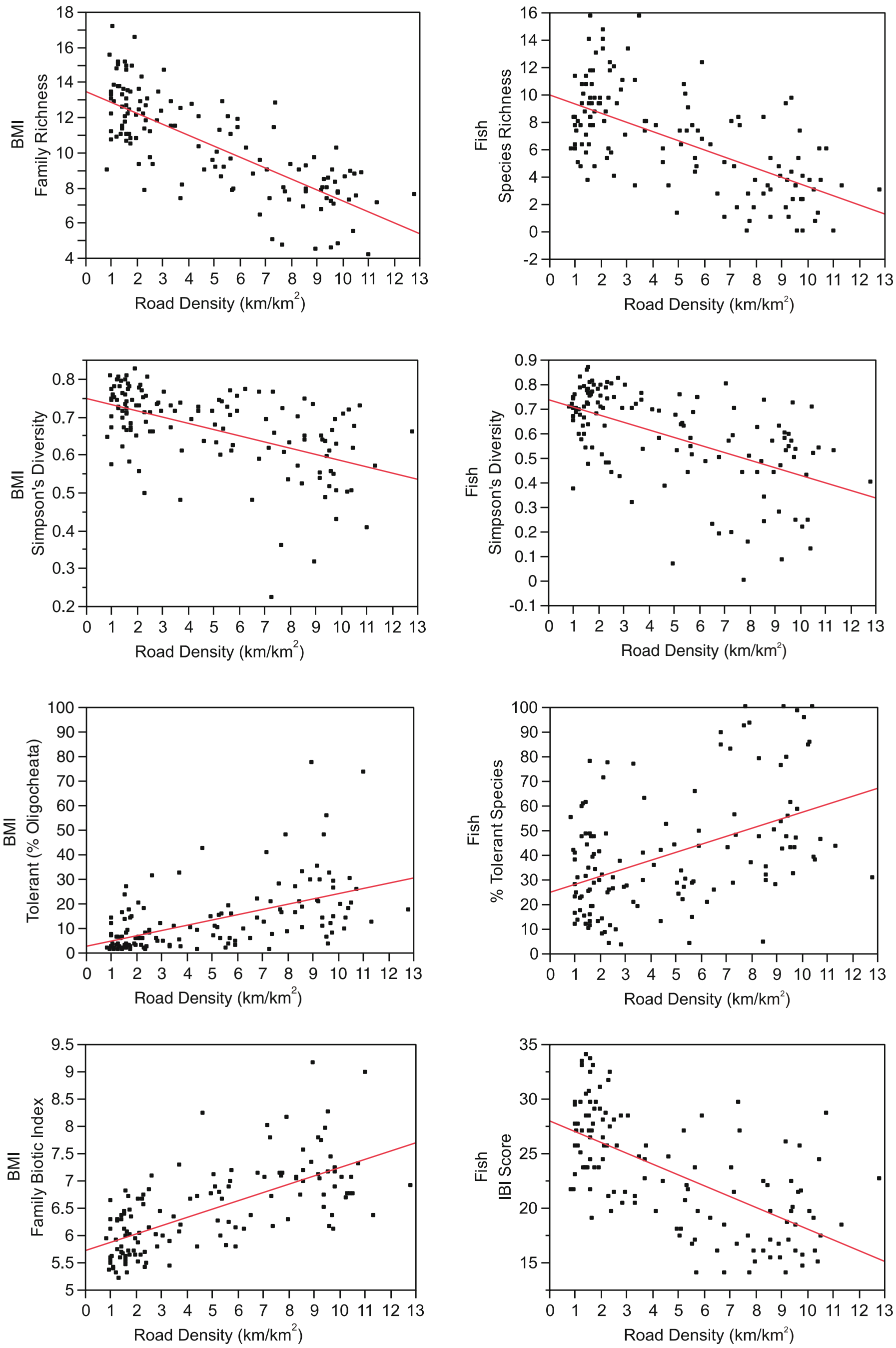

Fig. 3 Regression of biological indices and road density for 133 sites across TRCA jurisdiction

last a few hours for small catchments. The effect would be to reduce the strength of watershed size effects in comparison to what would have been seen using flows collected on a shorter interval. Using a smaller time interval is expected to improve the strength of this relationship. Nevertheless, a significant relationship does exist whereby urban streams are flashier than their non-urban counterparts.

The majority of the physical characteristics associated with water (e.g., velocity, depth, temperature, turbidity, and nutrient availability) and with the geomorphical features of the channel (e.g., width, bank height, and

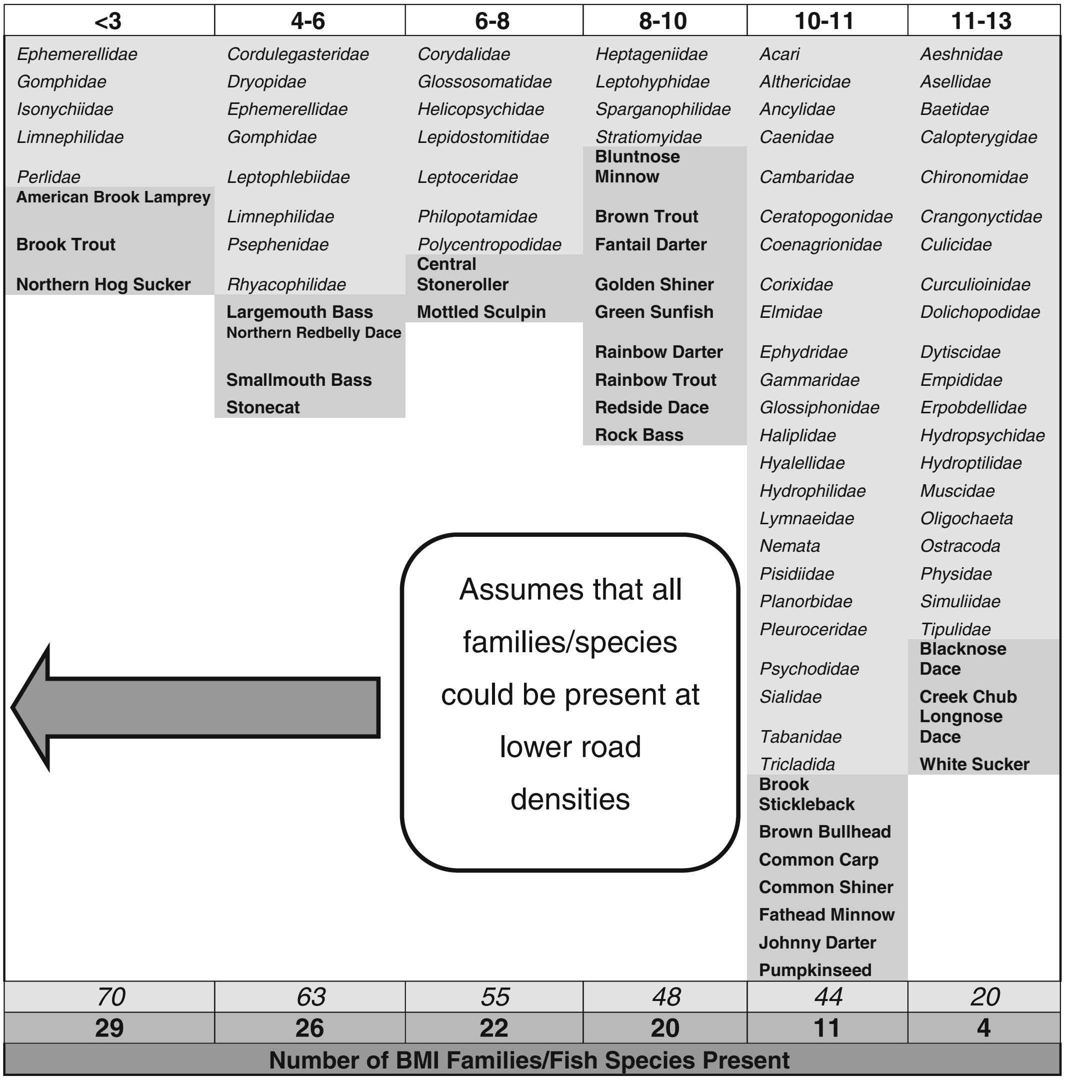

Fig. 4 BMI family/fish species lost at particular road densities (N=133 sites; analysis based on fish found at >3 % sites and BMI found at >5 % sites)

bed material) depend on stream flow (Konrad and Booth 2005). This study showed no significant relationship between road density and channel width, depth, width/depth ratio, or median sediment size (D_{50}). This may be simply a power/detection issue; however, these results are contradictory to most studies whereby streams draining urban land have shown changes in these factors (see Paul and Meyer 2001; Walsh et al. 2005). Changes to channel morphology usually result in the cross-sectional area increasing to accommodate larger bankfull discharge. Cross-sectional area was not analyzed in this study but may have proved to be a better metric to describe the changes in stream morphology. Furthermore, several of TRCA's more urbanized watersheds have been hardened (e.g., concrete channels, gabion caging, etc.) in some areas which would influence this relationship. The lack of relationship with median sediment size (D_{50}) and road density is also contradictory to other literature (e.g., Robert 2003; Pizzuto et al. 2000) which shows changes to sediment supply through erosion and altered sediment transport regimes due to changes in stream velocities as a result of land use change. The lack of relationship may have been an artifact of the collection method as it may not have been able to detect subtle changes in stream sediment size. The percentage of riffle habitat was found to be

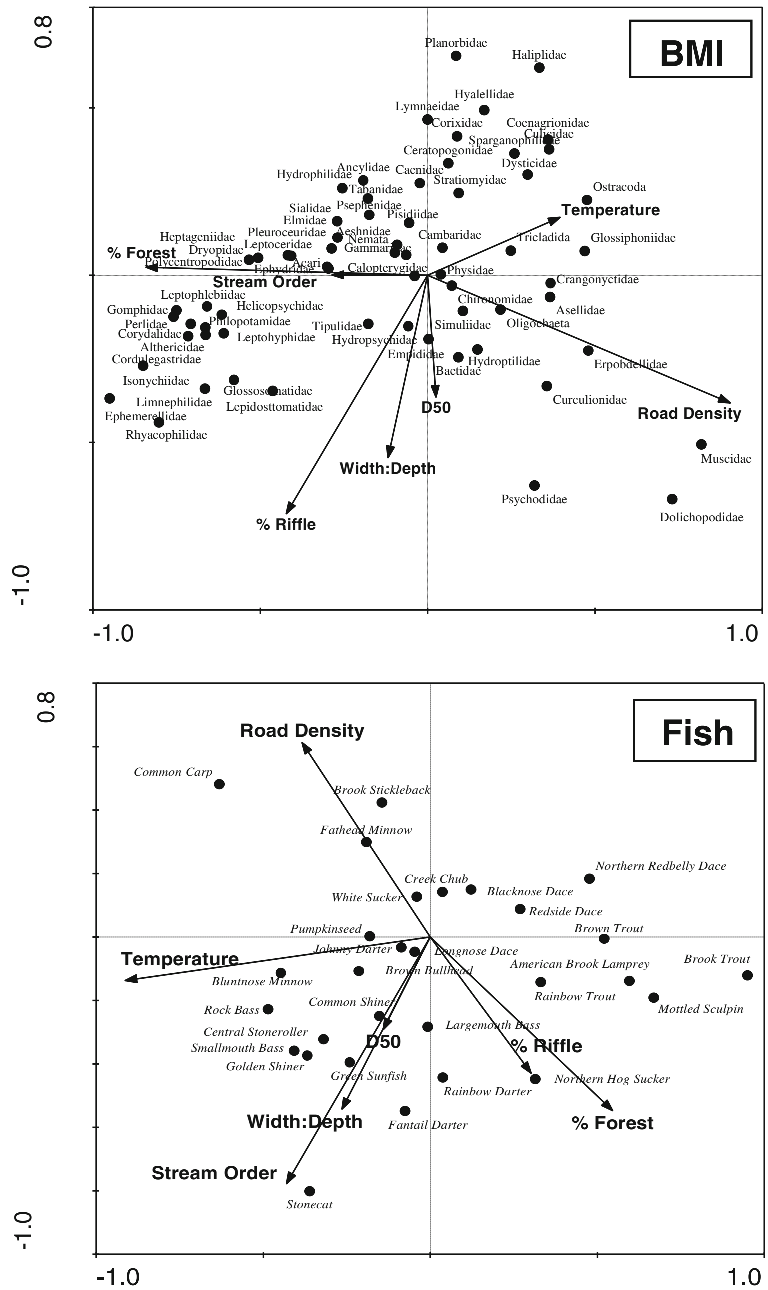

Fig. 5 CCA for BMI (*top*) and fish (*bottom*) for 127 sites located across the TRCA jurisdiction

significantly related to road density where the amount of riffle habitat decreased with road density suggesting some sort of erosional deposition did occur in the streams. Like other authors (e.g., see Paul and Meyer 2001; Booth 1991; Hatt et al. 2004), water temperature was also found to be significantly related to increasing urbanization (Table 7). This is likely due to a variety of process, such as the removal of riparian vegetation (reduced shading to streams), decreased groundwater recharge (decreasing the amount of cold water entering the stream), and increased stormwater entering streams as it flows over impermeable surfaces or is detained in shallow stormwater management facilities (increasing water temperature).

Changes in flow have also been associated with changes in the water quality regime of streams (e.g., Rhodes et al. 2001; Brett et al. 2005; Hatt et al. 2004). With the exception of TSS, all water chemistry analytes tested were found to increase with increasing road density. Several variables (chloride, copper, *E. coli*, sodium, and zinc) had strong relationships (all R^2>0.65) with road density suggesting that they are more abundant in urban areas and that impervious cover may serve to concentrate and convey these variables quickly to local waterways. Increased concentrations of chemical pollutants may be exacerbated in the southern portions of the study area as a result of a lack of stormwater management infrastructure as well as combined sewer overflows.

Urbanization has been repeatedly linked to declines in assemblage richness and diversity for various aquatic organisms including BMI and fish. Differences among the BMI and fish communities across a range of urbanization levels were obvious for the Toronto region. CA of BMI data showed that sites and species were separated according to pollution tolerance. CCA revealed that road density and %riffle were the two environmental factors having the most influence on the BMI community. Previous studies have suggested that stream BMI distributions are controlled by broad-scale factors rather than local habitat characteristics (Palmer et al. 1996; Poff and Huryn 1998; Vinson and Hawkins 1998). Road density represents a broad-scale anthropogenic factor associated with increased flashy flow, water temperatures, and contaminants. Percentage riffle represents a local habitat characteristic suggesting that both broad scale and local habitat play a role in shaping the BMI community composition. Unlike the BMI community, CCA showed that water temperature played a major role in influencing the fish community.

CCA for both BMI and fish revealed that sensitive organisms were found at sites with high forest cover and low road density, which happen to be coincident with areas of higher groundwater discharge. Forest cover within the upstream catchment also played a role in structuring the biotic community composition. Booth et al. (2002) recommended that maintaining forest cover in watersheds is more important than limiting imperviousness to protect the hydrological properties of streams. Road density had a strong influence on the distribution of the both the BMI and fish communities suggesting that all environmental factors correlated to road density (e.g., flow and various water chemistry variables) may be influencing the biotic populations. Sodium and chloride, the two main constituents of deicing chemicals used in the Toronto region, had the strongest relationship with road density suggesting that these two water chemistry analytes may be having a strong influence on the biotic community composition (Crowther and Hynes 1977; Kersey and Mackay 1981; Williams et al. 1997; Williams et al. 2000; Ourso and Frenzel 2003).

BMI and fish communities respond in a similar fashion to urbanization with decreased species richness and diversity along with increased number of tolerant organisms. Increased concentration of various metals, water temperature, water flow, and other water quality parameters associated with urbanization directly influence fish on an individual and community level. Individual responses include the ability of a fish to reproduce, their fecundity, metabolism, osmoregulatory and respiratory functions, growth and development rate, and immune system function. This influences the animals overall fitness and the amount of stress the fish can tolerate by effecting its ability to evade predators or parasites, capture prey, find mates and suitable habitat, and overall survival (e.g., Rahel and Hubert 1991; Hanson 2009). Furthermore, in lotic systems flow flashiness together with stream temperature and water quality influence the survival of fish eggs and fry (e.g., Coglan and Ringler 2005; Skoglund et al. 2011; Korman et al. 2011), as well as the community composition of BMI that are often used as prey items for many young and adult fish.

Most other studies only report on the relationship between the variables used to quantify community diversity and urbanization and fail to specifically identify the organism's present at most urban sites. This study found that fish and BMI communities changed as road density increased, such that at high road densities only the most common and tolerant species and families remained. At high levels of road density (>11 km^2/km), the number of BMI families was reduced to approximately one quarter of the number of families found at lower levels of road density. Although several BMI families may appear to be "sensitive" (e.g., Baetidae is a family of Ephemeroptera, and Ephemeroptera are generally considered a sensitive group), this is most likely a function of the level of taxonomy whereby there is a combination of sensitive and tolerant genera within a family. If completed on a more detailed taxonomic level, it is expected that these anomalies would not be present at high levels of road density. At road densities greater than 11 km/km^2, only four fish species were present in Toronto streams: Blacknose Dace, Creek Chub, Longnose Dace, and White Sucker. These four fish are considered ubiquitous and tolerant. The Blacknose Dace usually remains abundant in urban streams that are devoid of most other fishes as a result of its ability to adapt to change produced by the altered hydrological regime which allows for survival in urban flashy streams (Nelson et al. 2008). Creek Chub, Longnose Dace, and White Sucker may also have similar mechanisms allowing them to be successful in urban waters. Creek Chub, Blacknose Dace, and White Sucker are considered generalists, which may aid them in inter-specific competition for food resources and allow them to adjust to necessary dietary shifts as the BMI community changes as a result of anthropogenic pressure. These four fish species were reported as the dominant species present in the degraded Don River during the 1980s (Martin-Downs 1988).

When faced with the prospect of ongoing development in the region, it is important to be able to identify areas capable of withstanding increased urbanization. This paper identifies the range of road density that certain fish species can tolerate and can be used as a tool in urban planning. Documents such as Fisheries Management Plans could identify specific road density gradients that should not be exceeded in order to meet specific fish management goals. For example, if a specific reach is managed for coldwater Brook Trout (*Salvelinus fontinalis*) and American Brook Lamprey (*Lethenteron appendix*) then the road density should not exceed 3 km/km^2.

A multitude of studies have attempted to develop relationships between urbanization and aquatic communities (e.g., see Paul and Meyer 2001; Stanfield and Kilgour 2006; Schiff and Benoit 2007). Subsequently, a "threshold-of-effect" was developed which suggests that development greater than 5–15 % total imperviousness leads to poor stream health. Other studies have suggested that the relationship between biological communities and urbanization is linear (Groffman et al. 2006; Bazinet et al. 2010). In this study, strong linear relationships were identified between biotic metrics, water quality parameters, habitat variables, and road density. Non-linear curves were fit with all the variables but there was not a marked improvement in the R^2 over the linear fit. Utz et al. (2009) identified both linear and threshold responses with urbanization, depending on the scale of analysis suggesting that land use analysis must be completed at a finer scale in order to determine a threshold effect.

Consistent with the symptoms listed in Walsh et al. (2005), this study shows a correlation between road density and increased flashy hydrology, increased nutrients and toxicants, increased temperature, and decreased sensitive/increased tolerant BMI and fish. The results from this study generally agree with those of other researchers, some of whom used metrics other than road density to quantify urbanization (e.g., see Paul and Meyer 2001; Brown et al. 2005; Meyer et al. 2005; Stanfield and Kilgour 2006). Many studies have focused on a single, or perhaps two, alterations associated with increased urbanization but this study looked at geomorphical, hydrological, and biological changes simultaneously. In addition, similar results were seen between using two different biological communities. This study corroborates previous findings which associated the deterioration in the biological quality of streams with watershed urbanization. Urbanization is a complex process which affects various landscape, terrestrial and aquatic habitat, and water quality variables which in turn act to affect biota in a variety of ways. Understanding the linkage between the nature of the built environment and the responses of stream ecosystems will enable ecologists to provide valuable guidance to help improve the quality and function of streams draining urban land (Meyer et al. 2005). This study advances the field of urban stream ecology by highlighting some of the mechanisms influencing both BMI and fish communities.

In order to restore urban streams, ecologists need to identify the primary drivers of impairment and the actions needed to mend the problems. As well, stream ecologists, resource managers, and land use planners need to work together with the latest scientific data to support policies for sustainable communities to help create successful solutions for maintaining stream functions as areas continue to urbanize. Several studies (e.g., see Brabec et al. 2002; Walsh 2004) have discussed the importance of the degree to which impervious areas are directly connected to streams. These studies have shown that the generic measurement of total impervious area (TIA) is not as effective of an indicator as effective impervious area (EIA). TIA includes roofs, roads, parking lots and other non-infiltrating surfaces whereas EIA only includes impervious areas which have a direct hydraulic connection (e.g., piped storm sewer) to streams (Brabec et al. 2002). Walsh (2004) identified piped stormwater drainage connection as a likely causal factor explaining the widely observed loss of taxa from streams draining urban areas. Thus, the connectivity of impervious surfaces to streams is more likely a determinant of taxa loss than the impervious surfaces themselves. Low impact design (LID) has been identified as an approach with the potential to help improve urban stream conditions. LID is a landscape-based stormwater management planning tool which aims at reducing and/or slowing down the runoff that reaches the stream. TRCA has been promoting the use of LID for several years and has developed The Low Impact Development Stormwater Management Guide (CVCA and TRCA 2010) to help developers use the most up-to-date and effective technologies and thereby help ensure the continued health of the streams, rivers, lakes, fisheries and terrestrial habitats. Future work to evaluate the effectiveness of LID technologies in the Toronto region is planned.

As the population and footprint of cities expand worldwide, so does the need to effectively monitor and track changes in stream biota and habitat. This information should feed into an adaptive management process to develop low impact development approaches that ensure sustainable development and the persistence of stream habitat that continues to support a diverse biological community and healthy ecosystem function. The consequences of urbanization and the impacts on the aquatic ecosystem is a function of the decisions and action we take today for a sustainable tomorrow.

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