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HOW PROXIMITY OF LAND USE AFFECTS STREAM FISH AND HABITAT

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ABSTRACT

This study quantified the unique variation in stream fish and habitat and a land use disturbance index (LDI) at a variety of spatial scales: catchment, eight riparian polygons that varied in width and length (e.g. 50 m to all upstream reaches), upstream polygons of 1.6 and 3.2 km and the residual upland area of each site watershed not accounted for by each polygon. The analyses confirmed a hockey stick-shaped relationship between the fish community and the LDI, with sensitive species only present below an LDI of 11. The largest variation for most metrics was explained by the largest polygons, suggesting that local riparian conditions were not as important predictors of stream condition. LDI in upland areas, where zero-order streams occur, was also an important predictor of fish biomass and taxa richness. Contrary to expected, additive models with both catchment and riparian corridors provided minimal increases in predictive power, and no improvement in model performance occurred when data sets were stratified into sites below the LDI threshold. Finally, there was considerable covariation in the template and stressor predictor variables that made it difficult to quantify the unique variation in biological and physical responses accounted for by land use. That the 1600-m proximal polygon provided the best predictor of the fish community and temperature is supportive of there being some proximal effects of land use. Overall, our findings suggest that stream management must consider processes that occur in the entire upstream catchment and the entire riparian corridor, including the headwaters for success. Copyright © 2012 John Wiley & Sons, Ltd.

KEY WORDS: stream; fish community; habitat; riparian; proximity effects; landscape; land use; modelling

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INTRODUCTION

The influence of land cover on the biological and physical condition of streams continues to be an area of intense study (Klein, 1979; Steedman, 1988; Schueler and Galli, 1992; Shaver and Maxted, 1995; Jones and Clark, 1987; Horner *et al*., 1997; Morse *et al*., 2003; Wang *et al*., 2003a; Van Sickle, 2003; Allan, 2004; King *et al*., 2005; Frimpong *et al*., 2005; Stanfield and Kilgour, 2006; Schiff and Benoit, 2007; Van Sickle and Johnson, 2008; Didier *et al*., 2009; Stevens *et al*., 2010). Changes in stream hydraulics, water quality and temperature occur when landscapes are converted from forests and wetlands to agricultural and urban areas (Leopold, 1968). The resulting physical and chemical alterations can affect biological assemblages, often in predictable ways (Vannote *et al*., 1980; Steedman, 1988; Wang *et al*., 2003a, 2003b; Allan, 2004; King *et al*., 2005; Frimpong *et al*., 2005). Two recent special publications of the American Fisheries Society provided more than 70 new contributions that confirmed the influence of land cover on stream biota in North America (Brown *et al*., 2005; Hughes *et al*., 2006). Many studies have demonstrated a threshold response in the relationship between biotic indicators and measures of development in north temperate streams (Klein, 1979; Steedman, 1988; Schueler and Galli, 1992; Shaver and Maxted, 1995; Jones and Clark, 1987; Horner *et al*., 1997; Morse *et al*., 2003; Wang *et al*., 2003a; King *et al*., 2005; Stanfield and Kilgour, 2006). That is, the relationship between the biotic community and the land use/land cover is often linear at low levels of disturbance and flat at high levels of disturbance.

Such is the case for three recent articles from Ontario (Kilgour and Stanfield, 2006; Stanfield and Kilgour, 2006; Stanfield *et al*., 2006), which demonstrated that the composition of stream fish assemblages (multivariate descriptors, biomass of salmonids, number of fish species) is related to an indicator of landscape disturbance, the percentage of impervious cover. At low levels of imperviousness, the composition of the fish assemblage was highly variable but included cold- or cool-water fishes such as brook trout (*Salvelinus fontinalis*), brown trout (*Salmo trutta*), rainbow trout (*Onchorhynchus mykiss*) and sculpins (*Cottus* sp.). These sites tended to have low correspondence analysis (CA) axis 1 scores. Fish assemblages did not contain cold- or cool-water representatives above an imperviousness of 8% to 10% (CA axis scores >0.5). These levels of imperviousness are produced by land covers of >80% agricultural or >40% urban land cover within a catchment.

Although these studies provide general guidance of the potential impacts to streams from measures of catchment-level land cover alterations, they generally do not provide

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guidance on the spatial scale (the lateral and longitudinal distances) at which impacts from land use are occurring. The variability was presumed to be related to differences in local riparian and proximal conditions. Further, the variability in scores at sites located below the threshold meant that predictive models provided only coarse measures of potential responses to altered land cover scenarios (Kilgour and Stanfield, 2006). Sorting out the relative importance of the riparian and upslope (proximal) land cover effects on stream biological and physical properties is essential for effective restoration and mitigation strategies in developing landscapes.

Barton *et al.* (1985) were one of the first groups to look at the influence of riparian condition on stream biota. They demonstrated that streams in southwestern Ontario, protected by natural, contiguous riparian buffers that were ~1 km long, had cool midsummer water temperatures and a high probability of containing brook trout. More recently, researchers have taken advantage of improvements in geographic information systems (GIS) to evaluate the riparian vegetation and the proximity of land use to a site at varying scales of assessment. Wang *et al.* (2003a) used canonical CA in conjunction with redundancy analysis to determine the unique importance of three scales of influence on fish assemblages: the entire catchment, reach and local riparian conditions. The work of Wang *et al.* showed that the watershed-scale land cover conditions were better predictors of fish assemblages compared with the local riparian conditions, measured within a 30×100-m long polygon. This work also determined that reach-scale water quality parameters explained the greatest amount of variation in fish assemblages. For streams in Indiana, Frimpong *et al.* (2005) applied partial correlation and multivariate linear regression to explain the fish community using varying lengths and widths of riparian polygons. They concluded that buffers that were 30 m wide and 600 m long were the best predictors of the composition of the fish community. The more forested the buffer, the more similar the fish community was to a reference condition. Van Sickle and Johnson (2008), working in the Willamette River basin, used distance-weighting functions to determine that land cover in the riparian zone was important up to tens of kilometres upstream of a site but declined to nearly zero beyond 30 m from the channel. Wang *et al.* (2003b) also found that the land cover within 30 m of streams in Wisconsin and Minnesota explained more variation in fish assemblages than land covers beyond 30 m. It is clear from these efforts that the influence of land cover on biological and physical conditions in tributary watercourses is manifested over long distances and that the upland land cover can be less influential than the riparian land cover influence, but not always.

Wang *et al.* (2006), however, found that the scale at which the land cover best related to fish communities depended on whether the landscape was impaired ($>20\%$ urban or $>70\%$ agriculture). They concluded that instream and riparian habitat improvements would be most effective in catchments that are largely undisturbed, whereas land use management beyond the riparian zone would be more effective at improving stream quality in degraded catchments. Strayer *et al.* (2003) and King *et al.* (2005) have additionally shown that the relationship between development in a catchment and stream conditions varied with the size of the catchment, implying that the patterns were the result of differences in the relative importance of spatial arrangements across watershed size scales. Finally, most, if not all, landscape studies on this subject must contend with a fundamental challenge: that land cover is often confounded with the underlying geology and topography in an area (Van Sickle, 2003). In summary, the variability in results from studies of the scale of effects of landscape influences on streams can be attributed to the confounding within the predictor data sets, the regional variations in responses, the catchment size-dependent responses and the use of different types of metrics and approaches used to measure both the predictor and response variables (Fitzpatrick *et al.*, 2001; Van Sickle, 2003; Strayer *et al.*, 2003). Therefore, there remains substantial work to understand the influence of land cover at various scales, and it may be that the relationships are region specific. Fortunately, there are new tools and digital data that make answering these questions more feasible.

There were four main objectives of this article. The first objective was to evaluate the influence of land cover classification detail on model predictions using approaches that would enable results to be used in a predictive way to test new land cover scenarios. In our earlier efforts, land cover classification was based on an older and coarser digital data set, with only three land cover classes that represented disturbance regimes. In this study, a more recent digital map was used, which included many more classes of land cover that enabled us to refine the measures of land use in each polygon. We hypothesized that these improvements would improve the predictive capabilities of our models. The second objective was to evaluate the influence of land use proximity on the biological and physical conditions of a stream. We predicted that the land use/land cover calculated from the 30-m wide entire-catchment riparian polygon would explain the highest amount of unique variation in fish assemblage composition and instream habitat features (temperature, width-to-depth ratio, *etc.*) and that upland areas would explain less variation. Further, we wanted to quantify the covariation and unique variation explained by the template environmental variables (i.e. geology, slope, catchment area) and land use/land cover. We predicted that there would be considerable sharing of variation by template and land use variables but still significant unique variation explained by land use. Finally, we wished to evaluate

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whether development within riparian areas was more influential in explaining the stream biological and physical properties that were below a threshold of catchment development. A positive outcome of this analysis could potentially provide direction as to where stewardship activities might be expected to provide greater benefits to streams.

METHODS

Study area

The study was carried out in tributaries to Lake Ontario (Figure 1), where the Oak Ridge Moraine and the Niagara Escarpment dominate this landscape and provide strong baseflow to these streams. In this area, the coarsest grained soils (i.e. sands and gravels) and most hummocky topography are found in the north, and the finest soils (silts and clays) and flattest topography are found in the south. Throughout the 1800s and until circa 1920, nearly all lands in this area were cleared of forest cover and used for mainly row-crop agriculture (Puric-Mladenovic, 2003). During the last 100 years, much of the northern lands have been returned to forest cover or low-intensity agriculture (hay and pasture), whereas the land use in the remainder of the area has intensified. As a result, the present land use/land cover in the area consists of mainly forest cover and low-intensity agriculture on top of the moraine, whereas urban areas are predominantly near Lake Ontario and in the western part of the study area. Since Hurricane Hazel (1954) and its resultant legislations and policies, much of the valley lands on the main stem sections of rivers that are owned by government agencies have been restored to natural vegetation communities. However, large areas of riparian habitat and many headwater sections remain in private control where land use/land cover is controlled by local needs; such that considerable variation in riparian condition exists throughout the study area.

Streams in this area were historically known to contain abundant populations of brook trout and Atlantic salmon (*Salmo salar*) (Stanfield and Jones, 2003). Both species are considered cold water and sensitive and were greatly impacted during the European settlement period in this area (Dymond, 1965). Reforestation and other management strategies have and continue to be directed at restoring the abundance of cold water–sensitive species, albeit a different mixture of species, in these streams. In fact, Environment Canada (2001) has identified that any Lake Ontario tributary is considered impaired if the system does not contain cold water–sensitive species.

Study design. Most sites were selected using multiple stratified random designs that were linked through a network of agencies involved in monitoring within this study area. Several studies covered the entire ecoregion. Most studies were stratified based on a measure of stream size and land cover, such that collectively a wide spatial coverage and contrast in both land cover and geology was achieved (Figure 1).

Field methods. Data were collected during a 7-year period (1995 to 2002) by several agencies using modules of the Ontario Stream Assessment Protocol (Stanfield, 2000),

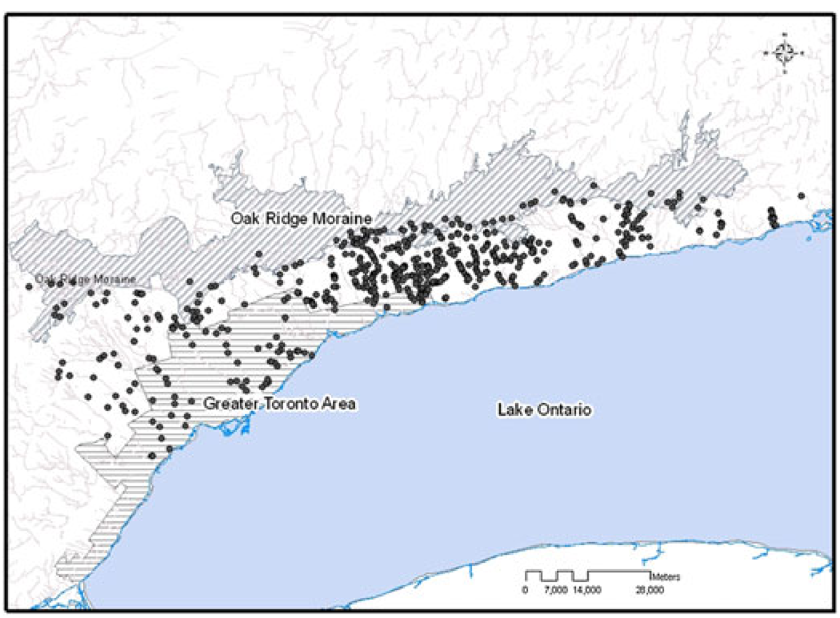

Figure 1. Study area and sample sites used in this study. This figure is available in colour online at wileyonlinelibrary.com/journal/rra.

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which enables project leaders to apply several methods at a specific sampling site. Where multiple years of data were available at a site, the most recent year was used. Data were collected from wadeable streams with maximum depth that rarely exceeded 1.5 m. Stream sites ranged from 40 to 142 m long, with site boundaries defined by crossovers (i.e. the location where the thalweg is through the middle of the stream) that were at least 40 m apart. The types of data collected (not all methods were applied at all sites) within each stratum were designed to meet the objectives and desired precision of each study.

Fish assemblages were characterized using single-pass electrofishing surveys at 312 sites, with all taxa identified in the field to species, except lampreys and sculpin. Lampreys were identified to the family (Petromizonidae) and sculpin were identified to genus (*Cottus*). Physical habitat data were available from 261 sites for which we focus on bank stability, average width and width-to-depth ratio because these three variables have been shown to be representative of the geomorphic condition of streams and have been shown to be correlated with urbanization (Leopold, 1968). These variables were characterized using a point-transect methodology that applies from 10 to 20 transects and 2- to 6-point observations depending on stream width. Depth and substrate size were measured at each observation point. The percentage of fines (particles $\leq$ 2 mm) and other measures of substrate condition (e.g. D16, D50 and D85) were determined from substrate particle size measurements. Four metrics were calculated to provide measures of channel stability (as per Stanfield and Kilgour, 2006): (i) width-to-depth ratio, (ii) bank erosion potential, (iii) sediment sorting and (iv) sediment transport potential.

Instream temperatures were available for 212 sites and were determined between 1600 and 1700 h during low-flow conditions (mid July to mid September) on warm days (>24°C for 3+ days) (Stoneman and Jones, 1996). Water temperatures were standardized to a 30°C air temperature based on models relating daily maximum air to daily maximum water temperatures (Stanfield and Kilgour, 2006).

GIS methods. We used a GIS system (ArcMap) to generate and then calculate the upstream catchment area and slope (determined from elevations 100 m upstream and downstream) for each site. For this analysis a 10-m resolution water layer and a 1:10,000 digital elevation model with 25-m resolution that included a flow-directional grid to improve accuracy of watershed delineation in hummocky terrain was used. Land cover was measured using a hybrid of the Southern Ontario Land Resource Information System (SOLRIS; OMNR, 2007a) land cover layer. This was derived by first digitizing 2002 Landsat images that represented natural land cover categories including woodlots, corridors, meadows and urban lands. Park lands and industrial areas were differentiated from urban subdivisions. At the time of this analysis, roads had not been included in the SOLRIS data set. To include these in the land cover layer, the provincial roads layer (OMNR, 2007b) was merged into this layer using the union process, after assigning the following standard widths of road allowances to the classified road types (primary = 60 m; secondary = 20 m; provincial highways = 100 m). This provided an updated layer for all land cover categories with the exception of agriculture. To include agricultural information in the land cover layer, the new combined SOLRIS and roads data were merged, using a union process within Arc View (version 8.3, ESRI, 2003), into the historic land cover layer (OMNR, 1999). This provided a seamless data set for the study area, with an overall pixel size of between 15 and 25 m. In effect, this land cover layer provided recent information (2001 and 2002) on urban, forest, water and roads data and assumed agricultural land use had not changed since 1992. Given land use practices in our area, this was not an unreasonable assumption.

We converted the land cover to a land use disturbance index (LDI) following Morrison *et al.* (2006). The metric was similar to the intent of the conventional measure of the percentage of impervious cover but differed in several key ways. First, it included a weighting for all land cover categories in the watershed, something that is not consistently performed in measures of impervious cover (Shaver and Maxted, 1995; Brabec *et al.*, 2002; and Shuster *et al.*, 2005). The LDI was calculated using a weighted average index with the following form:

$$\mathrm{LDI} = \frac{\sum \mathrm{Area}_i \bullet D_i}{\sum \mathrm{Area}_i} \tag{1}$$

where Area_i is the area of land use i and D_i is the disturbance index for that land use. Local ecologists and planners collaborated to provide D_i values (between 0 and 1) for each land use/land cover category based on the expected magnitude of effect on sediment transport and flow regimes, the two major drivers of stream channel processes (Leopold, 1968). Imperviousness values provided by Arnold and Gibbons (1996), Shuster *et al.* (2005) and Stanfield and Kilgour (2006) were used where appropriate to set the D_i values (generally for urban land covers; Table I). For this data set, land cover types with limited compaction or tilling were considered to have low levels of disturbance (i.e. $D_i < 0.1$).

The LDI was computed for sites on the basis of the conditions in the whole catchment draining to the site. LDI was also computed within ‘buffers’ that varied in width (30, 100 and 200 m wide from the creek centerline) and length (50, 500 and 1000 m, and the whole main stem of the river) (e.g. Figures 2a and 2b). The main stem of a river was defined as the longest upstream length of stream. LDI

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Table I. Land use disturbance coefficients and the median and maximum values from the fish distribution data set

Land class	Median, maximum	Disturbance coefficient	Rationale for coefficient
Water/wetlands	1, 19	0	No negative effect anticipated
All forests	18, 75	0.01	Some surface runoff due to access roads
Idle lands	1, 5	0.03	Tend to be meadows with heavier vehicle use
Pasture	14, 30	0.04	Low-level use by animals, nutrient runoff
Gravel pits	3, 91	0.1	Interception of groundwater, compaction
Cropland	50, 86	0.1	Nutrients, soil loss, compaction
Intensive pasture	<1, 76	0.15	Intensive cattle use, high tractor use
Urban	<1, 20	0.3	Parks and open areas: heavily cultivated
Roads	3, 17	0.9	Impervious areas

Ratings based on deviation from natural conditions that would influence the flow and sediment transport properties of streams.

was also calculated within 1600- and 3200-m polygons radiating upstream of the site, but within the catchment boundaries as determined from the DEM (Figure 2c). Finally, upland conditions were calculated by subtracting the area of each land cover type for the entire catchment from that of each riparian polygon. A conceptual illustration of these polygons is provided in Figure 2.

The area covered by each class of quaternary surficial geology (1:250,000) (Ontario Geological Survey 1997) was determined for each catchment. A baseflow index (BFI) for each site was derived using the model developed by Piggott *et al*. (2002):

$$\mathrm{BFI} = \frac{\sum \mathrm{Area}_i \bullet \mathrm{Index}_i}{\sum \mathrm{Area}_i} \qquad (2)$$

Here, Area_i was the total area within a catchment covered by surficial geological class i, whereas Index_i was a 'coefficient' that reflected the permeability of a surficial geological class

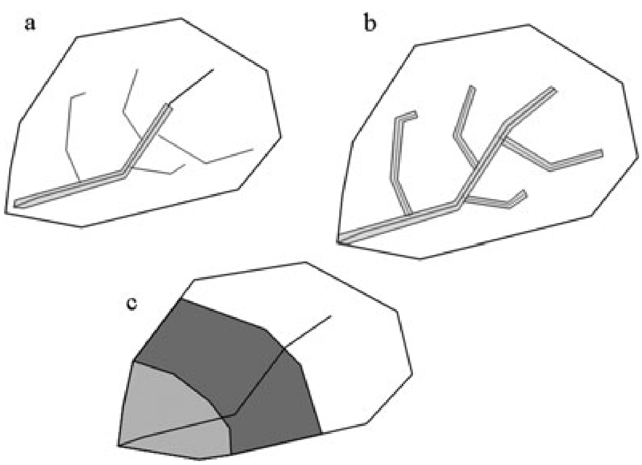

Figure 2. Conceptual illustration of the polygons within which LDI values were calculated: (a) buffers of various widths up the main stem, (b) polygons of various widths up to and including all of the tributaries to a site and (c) arcs of 1600 and 3200 m.

and the likelihood that a soils class would produce groundwater discharge to a stream.

Data analysis. The objectives of the data analyses were to (i) summarize the variations and covariations of the different measures of LDI and template variables against measures of fish community composition and instream physical conditions (response variables) and (ii) to determine the magnitude of variation in the response variables that was uniquely explained by LDI measured at different scales. Principal component analysis (PCA) was used to address the first objective. The catchment data used in the analysis were for those sites for which there were fish community data. Variables used in the analysis included each of the measures of LDI plus catchment area ($\log_{10}$ transformed), BFI and stream slope. This method (PCA) assists in determining how many independent gradients can be represented by the measured environmental descriptors. The PCA was useful in determining how the template variables of catchment area, slope and BFI covaried with the various measures of LDI and for identifying the dominant environmental gradients in the data set. Principal components with an eigenvalue >1 were retained for interpretation. Variables were considered strongly associated with a principal component axis if the correlation (r value) was greater than 0.6. Biplots of combinations of LDI measured at different polygon sizes were also produced to understand how polygon size influenced each land cover metric.

Backward, stepwise, multiple linear regression was used to address the second objective of the data analysis. The response variables included the summary fish community measures (logarithm of total biomass, g/m^2, taxa richness and a multivariate descriptor of the community). The multivariate descriptor was calculated from a CA of the data set analyzed by Stanfield and Kilgour (2006). In that analysis of a similar set of data (geographically and methodologically), the first axis from the CA described the major gradient from salmonine- and sculpin-based communities to a cyprinid- and

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centrarchid-based fish community after removing rare taxa from the data set. The rare taxa represented 34 species including northern pike (*Esox lucius*), central mudminnow (*Umbra limi*), 2 Catostomidae, 16 Cyprinidae, 5 Ictaluridae and 4 Percidae. This gradient has been described as differentiating communities dominated by cold water species sensitive to disturbance, and communities dominated by species preferring warmer water and that are more tolerant of disturbances (see analysis and explanation in Stanfield and Kilgour, 2006). Taxa scores from the previous analysis are replotted here (Figure 4) to provide context for this metric. The taxa scores obtained by Stanfield and Kilgour (2006) were used in this analysis to compute weighted average site scores for new data. The ordination diagram and scores used here are, therefore, exactly comparable with what was produced by Stanfield and Kilgour (2006).

The predictor variables in the multiple regression included the BFI, catchment area, slope and measures of LDI (plus the squared terms for each variable to allow for curvilinear relationships). Both the upland and the riparian LDI values for the entire catchment were included in additive models to partition the unique variation explained by either the upland or the riparian areas. Variables were retained in the stepwise regression if they accounted for a significant amount of variation in the response variable at a significance level (i.e. α) of 0.05. The use of stepwise regression ensured comparability between the results of this analysis with those of the previous analysis (Stanfield and Kilgour, 2006). Model 'runs' included combinations of template variables and LDI (i.e. with one or the other or both). 'Full' models included both template variables and LDI. Variation uniquely explained by template variables was considered equal to the variation explained by the full model (template + LDI), less the variation explained by a reduced model with LDI only. Variation uniquely explained by LDI only was considered equal to the variation explained by the full model, less variation explained by a reduced model with only template variables in the model. This analysis told us which measure of LDI was the most strongly associated with instream biological and physical responses. Only the unique variation explained by each landscape variable is presented, and only adjusted R^2 values of $\geq$0.05 were considered large enough to warrant interpretation (see Barrett *et al*., 2010). This general methodology follows that described by Qinghong and Bråkenhielm (1995) and Anderson and Gribble (1998).

Some of the biological and physical responses, in particular the multivariate descriptor of the fish community (i.e. CA axis 1 scores), produced a 'hockey stick'–type relationship when plotted in relation to LDI: the relationship was somewhat linear when LDI was low and flat when LDI was higher. Regression tree analysis (in SYSTAT 11, 2004) was used to quantify the LDI at which point the relationship changed. The various stepwise multiple regression models, which were performed for all sites, were also performed for sites below the critical LDI. This additional analysis was considered relevant because it dealt with sites that were considered the most 'responsive' (i.e. the response varied linearly in relation to LDI).

RESULTS

Because not all field data are collected at every site, the numbers of sites available for analysis varied for each response variable. Streams used in these analyses were generally small, ranging in catchment area between 0.06 and 300 km^2 (Table II). There was considerable variation in both the BFI, ranging between 14 and 77, and in stream slope, ranging between 1% and 7%. The LDI varied between 0 and 50, representing a gradient from fully forested to nearly fully urbanized. Taxa richness was generally low (~6 species per site) and ranged between 1 and 16 species. Blacknose dace (*Rhynichthys obtusus*, 79%) and creek chub (*Semotilus atromaculatus*, 65%) were the most common fish in the data set followed by white sucker (*Catostomus commersoni*, 53%), sculpins (*Cottus* sp., 40%), rainbow trout (39%) and longnose dace (*Rhynichthys cataractae*, 34%). The summarized fish assemblage results generated a gradient of CA axis 1 scores whereby sites with low scores tended to be more dominated by cold and cool-water fishes including Salmoninae and sculpins, whereas sites with higher scores tended to have few cold and cool-water species and were more dominated by taxa such as sticklebacks (Gasterosteidae), cyprinids and Johnny darters (*Etheostoma nigrum*) that are more tolerant of warm water conditions (Figure 3).

The initial assessment of covariation among land cover/land use data, as well as slope, catchment area and BFI was summarized by the PCA as provided in Table III. The first principal component axis explained 60% of the total variation in the landscape variables, with all but two of the various measures of LDI correlating strongly with the axis. Thus, sites with high LDI in the whole upstream catchment also generally had high LDI in smaller polygons, regardless of the proximity of the polygon to the stream site. Catchment area, BFI and stream slope varied independently of the first principal component axis and also did not correlate strongly with the second principal component axis. The second principal component explained subtle differences in how LDI metrics covaried depending on the spatial scale of the measurement and accounted for only 10% of the variation. LDI measured at the whole catchment tended to have positive correlations with the second principal component axis, whereas LDI measured in smaller polygons (i.e. in buffers adjacent to the streams) tended to have negative

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Table II. Summary statistics for the response and predictors variables used in this analysis

Parameter	Fish Minimum	Fish Maximum	Fish Average	Habitat Minimum	Habitat Maximum	Habitat Average	Temperature Minimum	Temperature Maximum	Temperature Average
Catchment area (km^2)	0.06	300	26	0.06	206	27	0.06	301	29
BFI	14	77	44	18	77	45	18	77	45
Site slope	1.0	6.9	3.1	1.0	6.2	3.1	1.0	6.2	3.1
LDI in whole catchment	3	49	14	3	49	13	3	49	14
LDI beyond the 100-m buffer	3	57	17	3	60	15	10	54	31
LDI beyond the 200-m buffer	3	79	20	0	79	18	0	61	14
LDI beyond the 30-m buffer	3	49	15	0	49	13	0	49	14
LDI in the 1000 × 100-m buffer	2	50	18	2	50	16	2	50	18
LDI in the 1000 × 200-m buffer	3	50	17	3	50	16	3	50	17
LDI in the 1000 × 30-m buffer	1	50	13	1	50	12	1	50	13
LDI in the 500 × 100-m buffer	1	50	15	1	50	14	1	50	16
LDI in the 500 × 200-m buffer	0	50	10	0	50	9	0	50	10
LDI in the 500 × 30-m buffer	1	50	14	1	50	12	1	50	13
LDI in the 50 × 100-m buffer	1	50	16	1	50	14	1	50	15
LDI in the 50 × 200-m buffer	0	50	10	0	50	10	0	50	10
LDI in the 50 × 30-m buffer	0	64	13	0	64	12	0	64	14
LDI in the main stem × 100-m buffer	1	50	15	1	50	14	1	50	15
LDI in the main stem × 200-m buffer	0	63	9	0	63	9	0	63	11
LDI in the main stem × 30-m buffer	1	48	11	1	47	10	1	48	11
LDI in buffer along tributaries × 100 m	1	49	13	1	48	12	1	49	13
LDI in buffer along tributaries × 200 m	0	47	9	0	47	8	0	47	9
LDI in buffer along tributaries × 30 m	1	49	13	1	49	12	1	49	13
LDI in the 1600-m polygon	1	49	13	1	48	12	1	49	13
LDI in the 3200-m polygon	0	46	11	0	46	10	0	46	11
Biomass (g/100 m^2)	1	7014	476						
Richness	1	16	5.5						
CA axis 1	−2.72	2.14	0.17						
Standard temperature (°C)							10.3	31.7	21.2
Bank stability				0.03	1.00	0.52			
Width (m)				0.4	89.0	4.4			
Width-to-depth ratio				2.4	714.9	26.5			
No. observations (n)	333			261			212		

correlations with this axis. There was little difference between the patterns observed for LDI measured along all the tributaries and for the entire watershed, implying a strong degree of association between these two measures of disturbance in the watersheds.

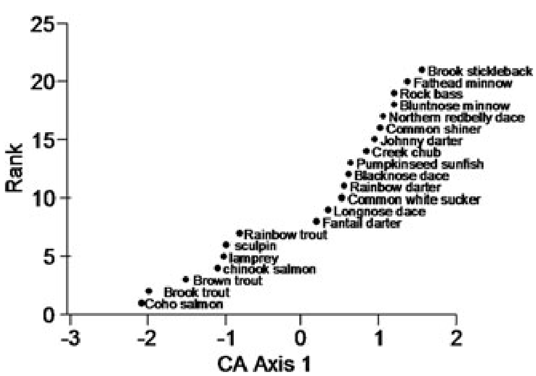

Figure 3. Biplot of CA axis 1 scores for fish species and their rank, illustrating the gradient in the fish community from cold water species, including coho salmon and trout, to warm water species such as stickleback.

Correspondence analysis axis 1 scores varied in somewhat of a 'step' fashion in relation to LDI at the catchment scale (Figure 4a). Regression tree analysis indicated that the relationship between CA axis 1 scores and LDI changed at an LDI of 11, with axis scores varying somewhat linearly below an LDI of 11 and being somewhat invariant at LDI values greater than 11. There was considerably more variability in CA axis 1 scores and the template variables of slope, BFI and catchment area. Sites with low BFI scores (low porosity) and low slope also contained only tolerant fish assemblages (high CA axis 1 scores; Figures 4b and 4c) and likely reflect sites located on the clay and till plains located near the Lake Ontario shoreline. Sites with higher porosity BFI scores and higher slopes (i.e. those in proximity to the Oak Ridge Moraine) contained a variety of fish assemblages (Figures 4b and 4c). A few sites with the lowest CA axis 1 scores of <-2 were only found in catchments of $<$1000 ha in area (Figure 4d).

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Table III. PCA of landscape and LDI data used to analyze fish community composition. Bolded text denotes variables which explain > 0.6 of the variation of an axis

Variable	Principal component 1	2	3	4
Log of area	0.07	0.27	0.40	**0.81**
BFI	−0.41	−0.10	0.22	0.05
Slope	−0.47	−0.15	0.06	0.18
LDI in whole catchment	**0.88**	0.33	−0.17	−0.02
LDI beyond the 30-m buffer	0.43	0.53	**0.64**	−0.25
LDI beyond the 100-m buffer	**0.70**	0.42	0.23	−0.04
LDI beyond the 200-m buffer	0.40	0.47	**0.67**	−0.28
LDI in the 1600-m polygon	**0.89**	0.09	0.00	0.29
LDI in the 3200-m polygon	**0.87**	0.24	−0.02	0.28
LDI in the 1000 × 100-m buffer	**0.93**	−0.24	0.00	0.05
LDI in the 1000 × 200-m buffer	**0.94**	−0.15	0.00	0.12
LDI in the 1000 × 30-m buffer	**0.87**	−0.31	0.02	−0.03
LDI in the 500 × 100-m buffer	**0.88**	−0.40	0.09	0.04
LDI in the 500 × 200-m buffer	**0.92**	−0.27	0.06	0.10
LDI in the 500 × 30-m buffer	**0.78**	−0.49	0.15	−0.03
LDI in the 50 × 100-m buffer	**0.74**	−0.51	0.23	−0.10
LDI in the 50 × 200-m buffer	**0.85**	−0.41	0.16	0.02
LDI in the 50 × 30-m buffer	**0.62**	−0.47	0.29	−0.16
LDI in the main stem × 100-m buffer	**0.92**	0.20	−0.24	−0.09
LDI in the main stem × 200-m buffer	**0.93**	0.22	−0.22	−0.04
LDI in the main stem × 30-m buffer	**0.87**	0.21	−0.24	−0.14
LDI in buffer along tributaries × 100 m	**0.88**	0.27	−0.20	0.03
LDI in buffer along tributaries × 200 m	**0.88**	0.21	−0.27	−0.09
LDI in buffer along tributaries × 30 m	**0.82**	0.26	−0.20	0.05
Percentage of variance explained	60.8	10.7	7.0	4.6

Stream temperature also tended to respond in a threshold manner, being variable at low levels of LDI and always high above a threshold of approximately 18 (Figure 5a). Fish biomass tended to decline in a linear response with

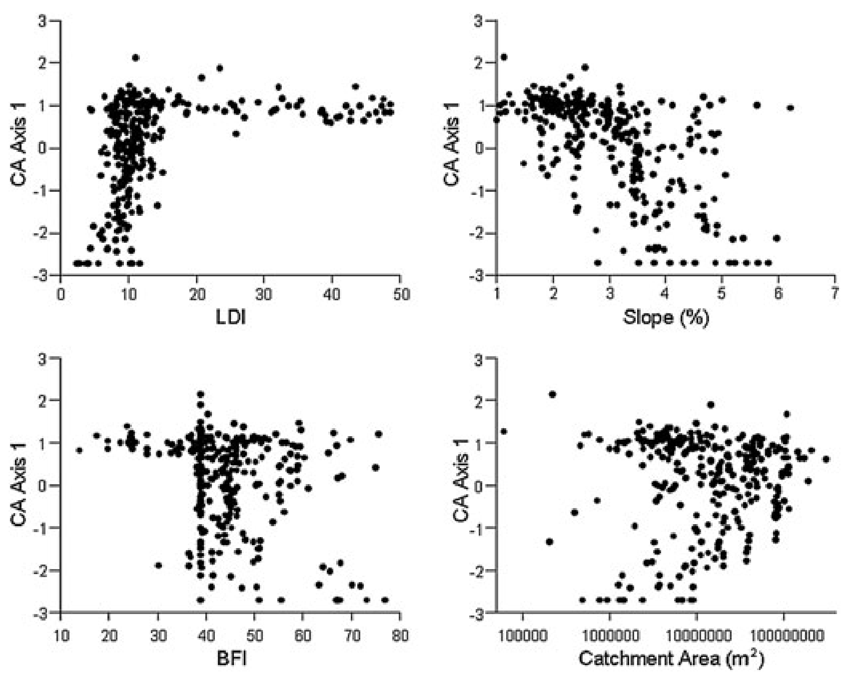

Figure 4. Biplots of CA axis 1 scores for fish and measures of LDI, slope, BFI and catchment area.

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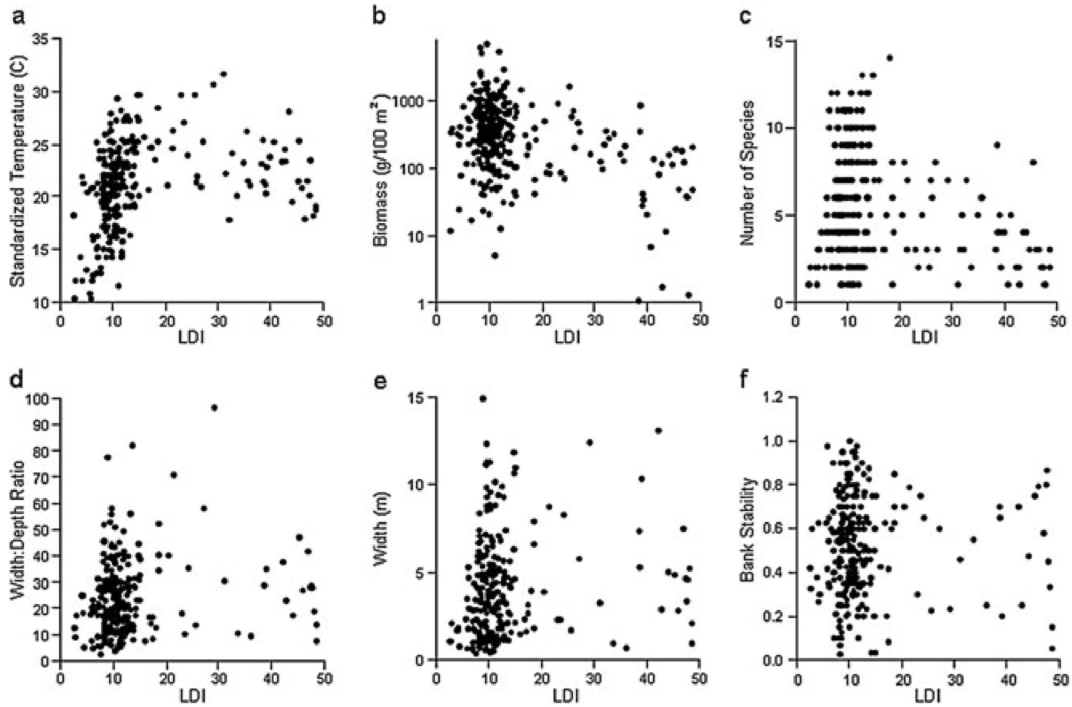

Figure 5. Biplots of physical habitat responses and standardized temperature in relation to LDI for the study sites.

increasing LDI (Figure 5b), whereas taxa richness tended to be low at both high and low levels of LDI and was highly variable at moderate levels of LDI (Figure 5c). Covariation with LDI was less apparent for physical measures (Figures 5d–5f).

The multiple regressions on the entire catchment data sets for the template and LDI model explained between 9% (for bank stability) and 71% (for stream width) of variation in biological and physical responses (Table IV). None of the predictor variables were related to the channel stability measure regardless of the scale at which LDI was computed. Models generated using both the template variables and the LDI polygons explained more variation in response variables than the template variables alone. In general, the template variables explained more unique variation in response variables than did LDI. Fish biomass was an exception to this finding with only 3% of variation being explained by the template variables (Table IV). Fish biomass and to a lesser extent taxa richness were the response variables most strongly associated with LDI. The amount of variation in each metric explained by LDI generally increased with the size of the polygon used to compute LDI. LDI measured in the catchment and in the upland areas beyond the 100-m buffer explained six times more variation in fish biomass than the template variables. LDI measured within the 1600-m polygon explained the greatest amount of unique variation (5%) in CA axis 1 scores, although an additional seven polygon shapes explained a marginally lower amount of variation. Only marginal increases in additional explained variation resulted from adding LDI to the template predictive model for any of the physical habitat models. The 1600-m polygon provided the best scale for measuring the effect of LDI on stream temperature (5% unique variation), although four other large-scale polygons (catchment, all tributaries 200 m wide, the 1000 × 200-m buffer and the 3200-m polygon) explained 4% of the unique variation. There was also very little increase in explanatory ability by developing additive models that incorporated either the riparian or the upland polygons. Only the model for biomass model (8% more variation explained) improved significantly by including the upland and riparian polygons, which was only effective for the 200-m buffer.

With sites with LDI < 11, the multivariate fish assemblage metric (CA axis 1 scores), bank stability, width-to-depth ratio and temperature all had higher correlations with template variables than was observed with the full data set, suggesting that the patterns for these variables are more consistent below the threshold than when the full data set is included (Table IV). Contrary to our hypothesis, the correlation between response variables and LDI in this reduced data set was largely lost. An exception to that finding was for fish biomass, for which LDI in the upland areas beyond the 30-m buffer continued to be an important predictor, although to a lesser degree than was for the full data set. In addition and in contrast to the analysis with the full data set, predictions were improved significantly for the multivariate fish assemblage measure by the inclusion of LDI in the 30-m buffer for the entire

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Table IV. Unique variation (adjusted R^2) explained by both the template variables (catchment area, slope and BFI) and each measure of LDI from the various scales (catchment, riparian buffers and upland areas)

	All sites							Sites with LDI < 11						
Landscape predictors	Biomass	Richness	CA axis 1	Bank stability	Width	Width-to-depth ratio	Standard temperature	Biomass	Richness	CA axis 1	Bank stability	Width	Width-to-depth ratio	Standard temperature
Template	0.03	**0.40**	**0.17**	**0.08**	**0.69**	**0.26**	**0.14**	0.01	**0.21**	**0.27**	**0.19**	**0.70**	**0.42**	**0.22**
LDI in whole catchment	**0.12**	**0.08**	0.04	0.01	0.02	0.02	**0.05**	0.03	0.02	0.03	0.01	0.00	0.00	0.00
LDI beyond 100-m buffer	**0.12**	**0.07**	0.03	0.01	0.02	0.02	0.01	0.01	0.01	0.01	0.04	0.00	0.01	0.03
LDI beyond 200-m buffer	0.02	0.02	0.03	0.01	0.01	0.00	0.02	0.03	0.02	0.03	0.03	0.00	0.00	0.04
LDI beyond 30-m buffer	**0.07**	0.04	0.02	0.00	0.01	0.01	0.01	0.01	0.01	0.00	0.03	0.00	0.00	0.00
LDI in 1000 × 100-m buffer	0.04	0.03	0.04	0.00	0.02	0.02	0.04	0.02	0.00	0.03	0.01	0.01	0.01	0.01
LDI in 1000 × 200-m buffer	**0.05**	0.03	0.04	0.01	0.02	0.02	0.04	0.02	0.00	0.03	0.00	0.01	0.01	0.01
LDI in 1000 × 30-m buffer	**0.05**	0.02	0.03	0.01	0.03	0.02	0.03	0.03	0.01	0.02	0.01	0.01	0.01	0.01
LDI in 500 × 100-m buffer	0.03	0.01	0.03	0.00	0.03	0.02	0.03	0.02	0.00	0.02	0.00	0.01	0.01	0.01
LDI in 500 × 200-m buffer	0.04	0.02	0.04	0.01	0.03	0.03	0.03	0.03	0.00	0.02	0.00	0.01	0.00	0.00
LDI in 500 × 30-m buffer	0.02	0.01	0.02	0.00	0.03	0.01	0.02	0.01	0.00	0.02	0.01	0.01	0.00	0.01
LDI in 50 × 100-m buffer	0.01	0.01	0.01	0.00	0.01	0.01	0.01	0.01	0.01	0.00	0.00	0.00	0.00	0.00
LDI in 50 × 200-m buffer	0.02	0.01	0.02	0.00	0.02	0.01	0.01	0.01	0.00	0.01	0.00	0.01	0.01	0.00
LDI in 50 × 30-m buffer	0.01	0.00	0.03	0.00	0.02	0.02	0.00	0.01	0.00	0.01	0.00	0.01	0.01	0.01
LDI in main stem × 100-m buffer	**0.08**	**0.06**	0.03	0.00	0.01	0.02	0.04	0.04	0.02	0.01	0.02	0.01	0.00	0.02
LDI in main stem × 200-m buffer	**0.09**	**0.06**	0.04	0.00	0.01	0.02	0.04	**0.06**	0.03	0.03	0.02	0.00	0.00	0.01
LDI in main stem × 30-m buffer	**0.08**	**0.06**	0.01	0.00	0.01	0.02	0.02	0.02	0.02	0.00	0.02	0.00	0.01	0.01
LDI in catchment tributaries × 100 m	**0.09**	**0.06**	**0.05**	0.00	0.01	0.02	0.03	0.04	0.02	0.04	0.01	0.00	0.00	0.02
LDI in catchment tributaries × 200 m	**0.08**	**0.05**	0.04	0.00	0.01	0.02	**0.05**	**0.05**	0.02	0.01	0.01	0.00	0.00	0.01
LDI in catchment tributaries × 30 m	**0.08**	**0.05**	0.02	0.01	0.01	0.01	0.02	0.01	0.01	0.04	0.04	0.01	0.00	0.01
LDI in 1600-m polygon	**0.06**	0.04	**0.05**	0.01	0.00	0.01	**0.06**	0.01	0.01	0.02	0.00	0.00	0.00	0.02
LDI in 3200-m polygon	**0.08**	0.04	0.04	0.01	0.01	0.01	**0.05**	0.01	0.01	0.01	0.01	0.00	0.00	0.02
LDI additive, 30-m buffer	0.03	0.02	0.01	0.01	0.01	0.01	0.01	**0.06**	0.03	**0.06**	0.04	0.00	0.00	0.01
LDI additive, 100-m buffer	0.00	0.01	0.02	0.00	0.01	0.02	0.01	0.04	0.01	0.04	0.00	0.01	0.01	0.02
LDI additive, 200-m buffer	**0.07**	0.03	0.02	0.00	0.01	0.02	0.03	0.04	0.02	0.01	0.01	0.00	0.00	0.01
LDI additive, beyond 30-m buffer	0.03	0.02	0.01	0.00	0.00	0.00	0.04	**0.06**	0.02	0.02	0.03	0.00	0.00	0.00
LDI additive, beyond 100-m buffer	0.04	0.02	0.01	0.02	0.02	0.02	0.04	0.01	0.00	0.01	0.03	0.01	0.01	0.03
LDI additive, beyond 200-m buffer	0.00	0.01	0.02	0.01	0.00	0.00	0.03	0.02	0.02	0.02	0.03	0.00	0.00	0.04

Models are generated using general linear modelling and included the square term of each variable. The additive models include both the upland and the riparian LDI, and the values presented represent the unique additional variation explained by this model after accounting for single LDI model results. Values ≥0.05 are shown in boldface to highlight substantial unique variation explained.

catchment polygon. Finally, the additive model that included either the 30-m buffer or the upland areas beyond the 30-m catchment polygon provided significant improvements in the fish biomass model.

DISCUSSION

The analyses presented in this article produced the following significant observations. First, this analysis demonstrated that higher-resolution land-cover data did not result in stronger relationships between instream biophysical responses and LDI than did a lower-resolution land cover layer. Second, the analyses confirmed that fish communities were invariant (warm water impaired) in streams in which the upstream catchment had LDI in excess of a certain minimum (here LDI 11). Third, local conditions (i.e. the LDI within smaller local polygons) explained significant variation in biophysical responses, but the magnitude of the influence was generally small. Fourth, and contrary to our expectation, local riparian condition was not a better predictor of stream condition in streams when the upstream catchment is largely undeveloped (i.e. LDI < 11). Finally, the analysis demonstrated that template conditions (i.e. slope, catchment area and baseflow potential) explain the most variation in instream biophysical conditions in Toronto area streams, but those conditions are confounded with land cover and land use. These findings are expanded upon and their implications discussed in the following sections.

Improving the relationship of stream properties to catchment conditions. This analysis, like our previous study (Stanfield and Kilgour, 2006) and many others (Klein, 1979; Steedman, 1988; Schueler and Galli, 1992; Shaver and Maxted, 1995; Jones and Clark, 1987; Booth and Jackson, 1997; Horner *et al*., 1997; Wang *et al*., 2001, 2003a; Morse *et al*., 2003; King *et al*., 2005; Walsh *et al*., 2005a; Schiff and Benoit, 2007), demonstrated a stepped threshold relationship between the land cover disturbance and the composition of the fish community and other metrics. That a threshold was apparent and at measures that are similar to what we demonstrated in the earlier work and those of Steedman (1988) and Wang *et al*. (2003a) was somewhat of a surprise and is disconcerting for several reasons. First, this analysis used a new land cover/land use layer that had finer and improved resolution in the classification of land cover/land use, yet there were only marginal increases in predictive power of the models. Although we cannot discount the possibility that these findings are at least in part due to the differences in the lower number of sites used in this study, we expected a greater increase than was observed. This study quantified the land development effect using a LDI. The LDI values were derived in part on the basis of the imperviousness values applied in the previous analysis, and much of the refinement of the data was in urban lands, woodlots, wetlands and roadways that have extreme (high or low) LDI values. That there was no refinement in the classification of agricultural lands, which both cover a large proportion of the study area and are inversely correlated with the amount of urban land cover (Stanfield and Kilgour, 2006), suggests that our findings may be correlated with this fact. This implies that agricultural land use could be a major contributor to the patterns observed and that efforts should be directed at improving the classification associated with agricultural lands, if further refinements of the relationships for sites that are below the threshold are to occur. Finally, the standardization of the approaches used to summarize measures of land cover disturbance will help clarify the patterns in stream response in the approach to and the universality of the location of the threshold across north temperate streams.

Riparian influences on stream condition. The results of this study suggest that measures of riparian conditions taken at a large spatial scale provide a comparable, or in one case marginally better, measure of the influence of land disturbance on the biological and thermal properties of streams to measures of the entire catchment. That the amount of additional variation explained by riparian polygons for sites that are below the catchment threshold was not greater than observed was contrary to our hypothesis and may be due in part to the high degree of correlation between measures of catchment LDI and riparian polygons when overall catchment LDI is low. That the multivariate fish assemblage metric is best predicted by LDI in the 100-m buffer that extends the entire catchment in sites with catchment LDI below the threshold provides at least some support for the contention that riparian buffers are more important in streams that are below the threshold for development. These results add to a growing list of research that have also demonstrated that regardless of the riparian condition for sites with development that exceed a minimum threshold, the fish assemblages are altered (Frimpong *et al*., 2005; Wang *et al*., 2006; Roy *et al*., 2007; Burton and Samuelson, 2008).

Proximal influences on stream condition. That the riparian polygon measured along in the 1600-m polygon provided the best correlate with the multivariate fish assemblage metric and stream temperature suggests that scaling effects are influential on structuring the biological and physical properties of streams. For these variables, it seems that land use in proximity to a site can have a greater influence than similar activities further away. Roy *et al*. (2007) also compared several scales of riparian polygons with catchment predictors of fish assemblages and found the strongest correlation to be with the largest spatial scale

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catchment in their instance. Our findings agree with those of Wang *et al*. (2003a) for fish, Wehrly *et al*. (2006) for temperature and Schiff and Benoit (2007) for benthic invertebrates and water chemistry that local conditions can be more influential than catchment conditions, but the findings of this study are not as large an effect as observed in any of these studies. Wehrly *et al*. (2006) attribute their observations to local groundwater influences and shading from forest cover, among other things. These are likely the pathway of effects for this study area as well. Schiff and Benoit (2007) conducted their work in a single coastal watershed of New Hampshire that had a large gradient in land cover conditions from within a 100-m buffer that extended upstream on all tributaries to a distance of 5 km. This scale of analysis is most closely associated with the LDI_100 polygon and the P3200 polygon, which were also two of the scales that provided high correlations with fish assemblages. Our findings disagree with those of Frimpong *et al*. (2005) that the optimal buffer dimension for developing correlations with fish assemblages is a very local 30 m wide and 600 m long. It is likely that the differences in results from these three studies are the result of the higher gradient in conditions observed in our study area. Differences between both Frimpong *et al*. (2005) and Wang *et al*. (2003a) are likely the result of these studies being conducted in mainly agricultural and forested study areas, with only 3% and 5% urban lands being present in their study areas, respectively.

Evaluating the ability of riparian zones to buffer development. Our findings that the additive models did not generally provide any substantial increase in correlative power for the biological and physical variables, regardless of whether the full or partial data set was used, was a surprise finding of this study. We assumed that particularly with the sites with catchments below the threshold of LDI development that there would be significant scaling effects that would increase the predictive power resulting from the addition of riparian data. We attribute our results to a combination of factors, including confounding in our data sets between the catchment and the riparian data (see below). Other studies (Wang *et al*., 2003a; Zorn and Wiley, 2006) have demonstrated that the process through which these scaling effects influence stream condition is complex, involving among other things groundwater contributions, shading effects on water temperatures nutrient cycling, and so on. This is especially true if stream temperature changes are a concern as there is considerable evidence of the benefits of forested riparian zones having a positive influence on this attribute (e.g. Brown and Krygier, 1970; Barton *et al*., 1985; Wehrly *et al*., 2006; Moerke and Lamberti, 2006; Wilkerson *et al*., 2006). Given the complexity of the interactions between the riparian zone and the stream (Nakano and Murakami, 2001) and the driving forces of hydrology that also occur at multiple scales, it is clear that a multiscaler approach is still recommended for even a partial understanding of stream processes. We do not suggest that work in riparian zones in streams that are above this threshold should not be carried out and will not have other benefits to the system, for example, reduced solar radiation, nutrient filtering, wood supply to the channel, and so on, rather that catchment-wide mitigation is also required if positive changes in stream biological assemblages are desired. Regardless, the particular findings in this study suggest that riparian zone enhancements are not enough to overcome the impacts of heavy development in the upland areas.

Importance of upland areas. The importance of upland areas as a predictor of fish biomass was somewhat of a surprise and is also contrary to the findings of Frimpong *et al*. (2005) that upland influence decreases to near 0 after 150 m. The water layer used in this study captures all but the smallest headwater streams, those generally called *zero order*. Therefore, we suggest that the upland layer may represent a measure of the land use in the catchments of the smallest headwater drainage features. These findings provide even more support of the importance of these features in maintaining catchment processes that are integral to stream health, as is suggested by a growing literature on headwater streams (for a synthesis , see Richardson and Danehy, 2007; Schiff and Benoit, 2007). Headwater features, in Southern Ontario, currently have received little attention or protection; hence, they are often buried, ditched, ploughed through or otherwise disconnected from downstream watercourses (Stanfield and Jackson, 2011). We anticipate that headwater streams in other locales are treated likewise (Adams, 2007). Our results suggest that land use in proximity of the headwater streams in these upland areas directly influences the factors that influence both fish biomass and taxa richness of downstream areas, and that future management activities should extend beyond the main river and its valley.

Influence of template conditions and confounded factors. Lands are used in ways that are partly predetermined by their underlying geology, soils and topography and their historic connection to travel corridors. Therefore, it was not a surprise that geology, slope, catchment area and LDI are all correlated in this study area. These covariations, however, make it difficult to extract the singular effect of development on stream properties. By conducting the analysis on residuals (after removing the influence of template conditions), the confounding influence of location, baseflow potential, slope and catchment area were 'removed', thereby providing (to the extent possible) a means of quantifying the unique influence of land use disturbance and the influence of scale of that measure. Subsequent partial redundancy analysis

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(Peres-Neto *et al*., 2006) conducted on these data determined negative fractions of shared variation between both area and geology/slope and for all three predictor data sets (LDI, geology/slope and area), suggesting that the relationship of these groupings with the fish assemblage is nonorthogonal (Legendre and Legendre, 1998). This supports the concept that the relationship between stream properties and land use responds in a nonlinear way with geology and therefore together explain the fish assemblage data better than the sum of the individual effects of each grouping (Legendre and Legendre, 1998). Zorn and Wiley (2006), using covariance structure analysis, demonstrated that a similar set of predictor variables influenced fish biomass through both direct and indirect paths, that also supports the contention that the relationships are nonlinear. Our concern that the patterns observed are not related to LDI are further reduced because the patterns observed here were consistent across the study area and are similar to patterns observed in other studies from temperate locales that have faced the same correlation challenge (e.g. Klein, 1979; Steedman, 1988; Wang *et al*., 2003a, 2003b; Zorn and Wiley, 2006; Schiff and Benoit, 2007, among others).

The location of various land cover types in this study is a direct result of its topography, historic settlement patterns and travel corridors. Therefore, the inherent correlation between land use and geology/position ensures that any attempt to remove the statistical effects of the confounding does not address the fundamental challenge that there may be additive effects that cannot be uniquely partitioned from other factors that contribute to development occurring in areas of highest vulnerability to impacts on streams. Parsing out the individual contributions of geology, slope, area and land use and historic conditions may be impossible because of the combined effect of the lack of data sets to truly describe the pristine state and the additive effects of the urban or 'developed' lands syndrome (Paul and Meyer, 2001; Walsh *et al*., 2005b). Although new approaches are emerging, which may enable clearer partitioning of variation to anthropogenic impacts (see Legendre and Legendre, 1998; and Fortin and Dale, 2005) and will better evaluate spatial variation in dendritic systems, it is possible that the drive for statistical correctness may mask or undermine the intent of the analysis; that is, to ensure that the disturbance on the landscape does not impact ecological integrity. The implications of these factors are that these mathematical models may underestimate the effect development will have on stream properties and argues again for use of the precautionary principle in planning decisions.

Summary and recommendations. In summary, this study has reconfirmed and refined the general relationship between LDI and instream biological and physical responses. The new models can be used to evaluate various land cover modifications in ways that enable planners to test new scenarios but has identified a significant analytical challenge to parsing in a predictive way the unique contribution associated with the various predictor variables. The data also show that LDI computed from the whole upstream catchment is generally a good predictor of instream biological and physical conditions, but in some instances, improved predictability can result from analysis along wide buffers that include all tributaries or within arcs that include proximal conditions. This study suggests that benefits to fish community assemblages and biomass from restoration work should be prioritized to locations that are below the threshold level of LDI development. The suggestion of additive effects between land use and geology/slope and fish assemblage has potentially large implications for land use land cover targets that at present do not consider underlying geology. We recommend these targets be revisited in light of our findings. Finally, populations in this study area are anticipated to grow by another 2 million people in the next 20 to 25 years and that new growth centers will be concentrated in the lands proximal to Lake Ontario, where the greatest sensitivity to increased flow rates may be located (Stanfield and Jackson, 2011) and which this study suggests are also most sensitive to alterations in fish assemblages. There is clearly a need for further work to evaluate the influence of individual land cover categories on streams that could lead to a standardized approach to measuring land disturbance. However, this work will require a much improved agricultural land use inventory for our area. Such improvements will be necessary if planners wish to be able to predict and thereby mitigate impacts from this development with any certainty.

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APPENDIX A

COMPLETE SET OF BIVARIATE PLOTS OF LDI VALUES FOR EACH POLYGON SIZE USED IN THIS STUDY AND EVERY RESPONSE VARIABLE (AVAILABLE AT WWW.TRCA.ON.CA/SOSMART).

APPENDIX B

MODEL COEFFICIENTS FOR EACH SIGNIFICANT MODEL PRESENTED IN THIS ARTICLE (AVAILABLE AT WWW.TRCA.ON.CA/SOSMART).

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