academicJournals

Vol. 5(12), pp. 396-406, December 2013
DOI: 10.5897/JENE2013.0415
ISSN 2006-9847 ©2013 Academic Journals
http://www.academicjournals.org/JENE

Journal of Ecology and the Natural Environment

Full Length Research Paper

Fish-habitat relationships in the Tonawanda and Johnson Creek Watersheds of Western New York State, USA

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Accepted 22 November, 2013

Warm water stream fish assemblages (2005) and habitat variables (2004 and 2005) were examined from May to September at 108 sites in the Tonawanda and Johnson Creek Watersheds of Western New York. Seventy species and > 27,500 fishes were identified; ~98% were from Families Cyprinidae, Centrarchidae, Catostomidae and Percidae. Data were analyzed at 16 spatial scales using best subsets and backward stepwise multiple linear regression to explore associations between individual fish species ≥9% of total catch and fish assemblage variables [catch per unit effort (CPUE), species richness, Simpson's diversity] with six habitat variables (pool type, maximum depth, substrate size, instream wood, bank cover, aquatic vegetation). CPUE was the only fish assemblage variable related to habitat variables, especially aquatic vegetation and pool type. Only two species (johnny darter, *Etheostoma nigrum*; round goby, *Neogobius melanostomus*) were significantly associated with habitat variables. The results reflected inherent difficulties understanding the complexities of habitat use by warm water stream fishes and their assemblages and how to manage them on a broad scale.

Key words: Warm water stream fishes, fish species-habitat associations, fish assemblage-habitat associations, statistical fish-habitat models.

INTRODUCTION

Management and conservation of aquatic resources requires the ability to identify species' distributions and habitat requirements (Argent et al., 2003). Lotic ecosystems are inherently difficult to study due to many factors that affect their transfer of mass and energy across the landscape (Fausch et al., 2002). Pool development and depth are among the most significant habitat attributes affecting stream fishes (Schlosser, 1982; Platts et al., 1983). Pools in general support more and larger fish than runs or riffles (Gillette et al., 2005; Sharma and Jackson, 2007; McGarvey and Hughes, 2008), but a pool's proximity to runs or riffles determines habitat suitability for certain stream fishes (Quist et al., 2006) by contributing to habitat heterogeneity (Lau et al., 2006). Stream systems are complex, and associations between fishes and habitat features vary considerably over spatial and temporal scales (Angermeier, 1987; Closs et al., 2004). With a few exceptions (cf. Smith, 1979; Smith, 1985; Pfleiger, 1997; Moyle, 2002), fisheries literature lacks information on specific habitat preferences of stream fishes.

Physical habitat commonly influences fish assemblages in lotic systems at various spatial scales (Angermeier, 1987; Lau et al., 2006). For instance, large woody debris stabilizes sinuous streams and increases local habitat

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diversity (Hunter, 1991; Flosi et al., 1998) and complexity (Angermeier and Karr, 1984); submerged aquatic vegetation also creates local structural complexity in aquatic systems (Brazner and Beals, 1997).

Understanding the patterns of fish assemblages in a watershed is dependent on the spatial scale of study; too coarse a sampling design may limit spatial (zonation) analysis (McGarvey and Hughes, 2008). Often, only a fragment of the entire ecosystem is covered (Fausch et al., 2002) making it difficult to locate fishes with specific habitat requirements. In addition, the transport of materials and organisms down the hydraulic highway is highly temporal (Fausch et al., 2002), and spatial variation is also high (Gorman and Karr, 1978). Substantial variation in habitat (for example, depth; Powers et al., 2003) reduces the ability to detect statistical associations with fish species or assemblages (Gerhard et al., 2005; McGarvey and Hughes, 2008).

Elucidating fish-habitat relationships has proven difficult (Beals, 2006), and most comparisons have not been statistically robust (Guy and Brown, 2007). Regression analysis is commonly used but has difficulty handling collinear variables (Beals, 2006). The objectives in this study were to evaluate warm water stream fishes and their assemblages in relation to habitats (for example, pool type) at watershed and sub-watershed scales. Best subsets and multiple linear regressions were used to explore associations and test null hypotheses that stream habitat features and fish species or their assemblages were unrelated at various scales in the study streams.

MATERIALS AND METHODS

Study area

This study was conducted in the Tonawanda and Johnson Creek Watersheds (TCW, JCW) of Western New York State (NYS) (Figure 1). Both are warm water streams supporting similar fish assemblages dominated by Families Cyprinidae, Centrarchidae, Catostomidae and Percidae. The TCW covers an area 5.6 times larger than the JCW and extends across Erie, Niagara, Genesee, and Wyoming Counties, ultimately draining into the Niagara River via the western portion of the NYS Barge (Erie) Canal. The JCW borders the northeastern corner of the TCW, mostly in Niagara County, and then flows through Orleans County into Lake Ontario. The TCW is more complex and urbanized than the JCW but both are affected by canals, dams, storm water and agricultural runoff.

The Erie Canal creates hydrologic anomalies in the TCW near river kilometer 18 (rkm) (main channel/canal confluence) where canal flow reverses when lock E34 opens and stream flow increases when the canal is lowered during the winter season. The canal is also a vector for invasive species in many watersheds from Buffalo to Albany across NYS (Carlson and Daniels 2004), including the JCW where canal water enters from discharge valves and the lower TCW that is connected directly to the canal. Despite the high density of lowhead dams in the TCW (109) compared to that of the JCW (18), disturbance to the natural flow regime may be greater in the main channel of the JCW because of the Lyndonville Dam near rkm 18 that creates an expansive impoundment and impedes fish migration from Lake Ontario. Smaller lowhead dams on the main stem of the TCW also impound some water and create fish passage barriers but a narrow lotic channel is maintained.

Surface runoff from many farms is the major non-point source pollution vector in both the TCW and JCW; but storm water runoff from eastern suburbs of Buffalo affect the lower subwatershed (canal reach) of the TCW. The TCW is also much more turbid than the JCW as a result of its numerous exposed clay banks, especially in the lower 50 rkm. Although wild and stocked trout persist in the very upper portions of the TCW, most of the main stem and major tributaries of both watersheds are warm water systems which do not receive the same level of environmental protection in NYS as cold water ‘trout’ streams.

Survey protocols

Stream habitat data were gathered at 68 sites in the TCW and 40 sites in the JCW from May to September, 2004. Following methods similar to Murphy and Willis (1996) and Platts et al. (1983), six habitat variables were assessed: pool type, maximum depth, substrate composition, instream wood, bank cover, and aquatic vegetation (Table 1). These six variables were selected with regard to longear sunfish (*Lepomis megalotis*) habitat preferences (Wells and Haynes, 2006) and ease of visual observation or semi-quantitative estimation in the field. For fish sampling in 2005, 72 sites were selected randomly from the 292 potentially fruitful sampling sites identified in 2004 but 36 additional sites were purposely chosen in the field because of access issues and habitat changes from the wet summer of 2004 to the dry summer of 2005 .

Fish surveys were conducted mostly during the day, guided by the NYS Depertment of Environmental Conservation’s (DEC) Centrarchid Sampling Manual (Green, 1989). Site length (m) was estimated after 15 min of power-on electrofishing effort. To avoid pseudo-replication (Hurlbert, 1984) across the many kilometers of stream sampled in each watershed, no sampling sites were closer than 100 m (average 8.4 rkm) or 30 m (average 1.8 rkm) in the TCW and JCW, respectively. Where depth permitted, an 18-ft electrofishing boat (Type VI-A Pulsator and 5000 W generator, Smith-Root, Inc., Vancouver, WA, USA) was used. Other sites with water <1.5 m deep were sampled with a backpack electrofisher (HT-2000, Hall-Tech, Ltd., Guelph, Ontario, Canada).

In addition, two small beach seines (4.0 × 2.1 m and 6.4 × 1.2 m; 6.4 mm mesh; no bags) and a larger 15.2 × 1.8 m seine (9.5 mm mesh, center bag) were used immediately after backpack electrofishing at sites <1.5m deep to improve the effectiveness of collecting small fishes. Seines were pulled parallel and perpendicular to the shoreline until the desired effort was achieved. Hauls ranged from a maximum ten to a minimum of four (without fouling) per sampling site. The objective was to representatively and semi-quantitatively sample the fish assemblage at each site so as to maximize species diversity in the catch. Specimens were identified to species in the field (Smith, 1985; Page and Burr, 1991; Knopf, 2002; Nelson et al., 2004) and counted. Unidentified species, young-of-the-year and suspected hybrids were preserved in 10% formalin and returned to the laboratory for identification.

Spatial extent of sampling

Sixty-eight sites were sampled during 29 trips covering 155 rkm in the Tonawanda Creek watershed in 2005 (Table 1, Figure 1). Sites ranged from the western most extent of the Erie Canal near its confluence with the Niagara River, eastward 18 rkm to the main stem of Tonawanda Creek (14 sites), then upstream past a waterfall and dam (these defined the sub-watersheds) to the headwaters. Sampling occurred at 33 sites in the lower main stem of Tonawanda Creek (TC), including six sites in the lower reaches of tributaries (Table 1). In the combined middle and upper main stem of TC, 18 sites were sampled, including four sites in the lower reaches of tributaries.

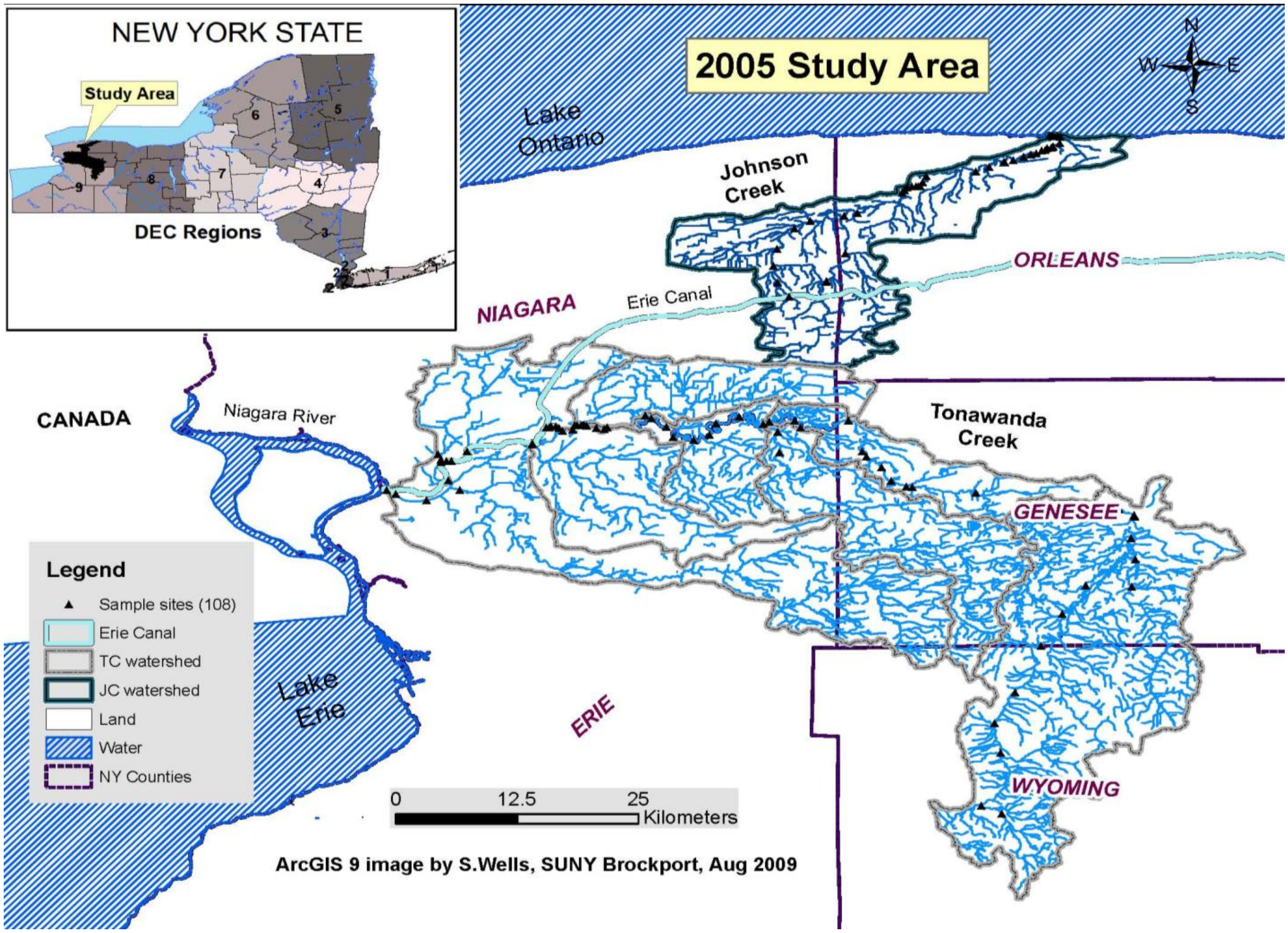

Figure 1. The sampling area in western New York State USA, May to September, 2004 and 2005.

Forty sites were surveyed during 19 trips covering 44 rkm in the Johnson Creek watershed from the mouth at Lake Ontario to the Erie Canal overpass, including its major east branch, Jeddo Creek (Figure 1). A total of 24 sites were surveyed in the lower main stem of JC from Lake Ontario upstream to the Lyndonville Dam (18.3 rkm). Another 13 sites were sampled in the upper main stem above the dam in Lyndonville, NY upstream to the canal, plus three more sites in Jeddo Creek (Table 1).

Statistical analysis

Several important assumptions were inherent in the design of this study and analysis of these data: 1) Random selection of most fish sampling sites also provided random physical habitat data, 2) Intensive sampling using two techniques provided a representative sample of a site's fish assemblage, and 3) Intra-stream movement of fishes was minimal during the hot/dry summer of 2005 due to low flows, consistent in-stream temperatures and no flooding events across either watershed. Low water concentrated fish in shallow pools, increasing sampling effectiveness, and consistent in-stream summer water temperatures negated thermal advantages of fish movement.

Habitat, fish species and fish assemblage data were tabulated for each sampling location in each watershed which was then separated into sub-watershed spatial units (Table 1) and pool types (Table 2). Raw habitat measurements were standardized (Table 2). To minimize observer bias, the first author scored all habitat variables. Catch per unit effort (CPUE), species richness and Simpson's diversity were calculated for each of the 108 sampling sites. CPUE and species richness data were transformed (square root) to meet assumptions of equal variance and normality.

Using JCW data only, preliminary best subsets (BSR) and backward stepwise (SWR) linear regressions (Statistix, 2003) were used to explore relationships between the three fish assemblage variables (CPUE, species richness, Simpson's diversity) and six habitat variables (Table 2) for the entire watershed and its two sub-watershed units (Table 1). Associations between habitat variables and species ≥ 1% of total abundance (Table 3), plus two rare (longear sunfish; redfin shiner, *Lythrurus umbratilis*) and two invasive (round goby, *Neogobius melanostomus*; rudd, *Scardinius erythrophthalmus*) species sampled in the JCW, were explored. Except for a few species > 9% abundance at one or more spatial scales, initial examinations by BSR and SWR of associations between the six habitat variables and fish species ≥1% of total abundance (19 species) were not informative for any of the spatial scales. We then chose a threshold of ≥ 9% total abundance to examine the most abundant fishes (2 to 4 species per spatial scale) in the JCW. Based on the criteria developed during the preliminary JCW analyses, the TCW dataset also was analyzed for species >

Table 1. Fish richness and total abundance, catch per unit effort at each spatial scale, and Simpson's diversity examined by spatial unit (N = 8) and pool type (N = 4) in the Tonawanda and Johnson Creek watersheds, excluding hybrids, subspecies and unidentified juveniles.

Spatial Unit	Number of sites	Fish richness	Fish caught	Catch per unit effort	Simpson's diversity
Tonawanda Creek Watershed	68	64	21,310	868	0.794
[1]Erie Canal	8	38	1,366	97	0.750
[2]Lower main stem	33	57	8,557	327	0.816
[2]Middle and upper main stems	16	32	12,752	541	0.736
Tonawanda Creek and Erie Canal tributaries	11	44	3,074	137	0.732
Pool Type 1	22	49	2,966	142	0.808
Pool Type 2	12	43	1,711	74	0.790
Pool Type 3	11	44	3,762	139	0.756
Pool Type 4	23	52	12,871	512	0.800
Johnson Creek Watershed	40	47	6,218	270	0.810
[3]Lower main stem	24	42	3,158	143	0.823
[3]Upper main stem and east branch	16	37	2,919	197	0.790
Pool Type 1	14	32	907	69	0.855
Pool Type 2	7	32	1,115	37	0.796
Pool Type 3	8	34	1,745	65	0.752
Pool Type 4	11	37	2,454	100	0.802
Total (sites, species, fish)	**108**	**70**	**27,528**		

[1]Sampling sites in the Erie (NYS Barge) Canal ranged from just above its confluence with the Niagara River upstream to its confluence with Tonawanda Creek at river km 18. [2]Main stem basins were delineated by barriers to upstream fish passage. Lower Tonawanda Creek included the reach from the confluence with the Erie Canal upstream to Indian Falls, middle Tonawanda Creek included the reach from the Indian Falls upstream to the Batavia Dam, and upper Tonawanda Creek included the reach above the Batavia Dam into the headwaters. [3]Upper and Lower Johnson Creek were divided by the Lyndonville Dam at river km 18.3.

9% total abundance. Due to few sites sampled, data from the middle and upper subwatersheds of the TCW were combined to form one spatial unit.

Because pool type was included in 14 of the 21 significant preliminary JCW BSR models, and the explanatory power of the entire- and sub-watershed-scale models was generally poor (low adjusted-r^2 values), survey data were explored further at the smaller pool-type (Table 2) scale in both watersheds. Except for the rare and invasive fishes noted above, only those BSR models with the lowest Mallow's CP scores and adjusted r^2 values ≥20% which were statistically significant in the SWR models (P≤0.06, because a number of the models had P-values of 0.05 to 0.06) were explored further. In the end, five spatial units were analyzed in the TCW (entire watershed, two sub-watersheds, Erie Canal and its tributaries, TCW tributaries combined), and the JCW data were analyzed at three spatial scales (entire watershed, two sub-watersheds) (Table 1). Both watersheds were analyzed in relation to the same four pool types (Table 2).

It is often difficult to balance statistical rigor and ecological meaning in relation to the potential for Type I (false indication of a significant difference) and Type II (false indication of no significant difference) statistical errors, especially when many statistical tests are used to analyze the same data set. BSR models suggested 80 significant habitat associations with 47 species and 33 assemblage variables among the spatial (N = 8) and pool (N = 4) units evaluated. Sixty-one SWR models were significant in the final analyses but at α = 0.06 the potential for a Type I error was 0.977 ($1 - 0.94^{61}$). We used the Bonferroni correction (cf. Tiemann et al., 2004; Freeberg, 2008, Etinger et al., 2009) to control for Type I errors; in this case the adjusted α was 0.001 (α / n tests = 0.06/61 significant models). Subsequently, only 12 SWR models for species or fish assemblage parameters were significantly related to one or more habitat variables. While the chances of making a Type II error rose substantially, we present only the 12 most robust (P ≤ 0.001) models from Wells (2009) here.

RESULTS

Cyprinidae was the most common family (23 spp.), comprising > 60% of all fishes recorded in the Tonawanda Creek Watershed, followed by Centrarchidae (16%, 11 spp.), Percidae (15%, eight spp.), and Catostomidae (7%, six spp.). These four families represented > 98% of all fishes recorded (Table 3). Only nine species comprised ≥ 9% of the total abundance over the nine spatial scales analyzed in the TCW; 40 of 64 total species (63%) sampled in the TCW comprised < 1% of the total abundance.

Cyprinidae was the most common family (15 spp.), comprising > 47% of fishes recorded in the Johnson Creek watershed, followed by Percidae (21%; six spp.), Centrarchidae (16%; six spp.), and Catostomidae (11%; four spp.). These four fish families accounted for > 97%

Table 2. Physical habitat variables observed and scored in Tonawanda Creek and Johnson Creek. Instream wood included standing or submerged timber (dead or alive) plus logjams, docks and pilings. Bank cover (natural or artificial) included overhead riparian canopy, overhanging bank vegetation, undercut banks, riprap and boulders, bridges and culverts. Aquatic vegetation included submergent, emergent and floating forms, excluding algae and mosses.

Variable	Observation	Determination	[1]Score	[2]Range
[3]**Pool Type (PT)**				
	Channelized reach	Lowest complexity	1	1.0 - 1.74
	Isolated pool or run	Some complexity	2	1.75 - 2.49
	Pool with run	Moderate complexity	3	2.50 - 3.24
	Pool with riffle	Highest complexity	4	3.25 - 4.0
Maximum Depth (MD)				
	Very shallow	0.5 m or less	1	1.0 - 1.74
	Mostly shallow	0.6 to 1.4 m	2	1.75 - 2.49
	Moderately deep	1.5 to 2.9 m	3	2.50 - 3.24
	Mostly deep	3.0 m or more	4	3.25 - 4.0
[4]**Substrate Size Score (SS)**				
	Very fine particles	mostly silt	1	1.0 - 1.49
	Fine particles	mostly sand	2	1.5 - 2.49
	Small course particles	mostly gravel	3	2.5 - 3.49
	Large course particles	mostly rock	4	> 3.5
% Instream Wood (IW) / Bank Cover (BC) / Aquatic vegetation (AV)				
	Absent	0%	1	0
	Present	5% or less	2	>0-2.49
	Moderate	6 to 25%	3	2.5-3.49
	Abundant	26 to 49%	4	3.5-4.49
	Dominant	50% or more	5	>4.5

[1]Score of 1 (lowest) to 5 (highest) were used to assign values for habitat complexity at each fish sampling site. [2]The range of scores used to quantify the amount of fish cover (an presumably better habitat) at each sampling site. [3]Type of pool was determined by the type of current (riffle or run) within or adjacent to a sampling site. [4]Substrate size score was the mean of estimated percent cover of each particle size group at a sampling site.

of all fishes recorded (Table 3). Only ten species comprised ≥ 9% of the total abundance over the seven spatial scales analyzed in the JCW (Table 3); 28 of 47 total species (60%) comprised < 1% of the total abundance.

Fish assemblage-habitat variable associations

Catch per unit effort accounted for 75% (9/12) of significant BSR associations ($P \leq 0.001$; see statistical analysis section of Methods); species richness and Simpson's diversity were not significantly associated with any of the six habitat variables measured (Table 4). CPUE was positively associated with habitat variables in 67% (6/9) of the 12 significant SWR models (Table 4), including type 3 pools (r^2 = 0.398) in the entire TCW, type 2 pools (r^2 = 0.217) in the entire JCW, sandy substrate (r^2 = 0.628) in the upper subwatershed of the JCW, and with low density of aquatic vegetation in the TCW's entire subwatershed (r^2 = 0.398), middle + upper subwatersheds combined (r^2 = 0.703), and in type 4 pools (r^2 = 0.409). CPUE was negatively associated with habitat variables in 33% (3/9) of the 12 significant SWR models (Table 4), including low-moderate density of instream wood in the JCW type 3 pools (r^2 = 0.997), moderate aquatic vegetation cover in JCW type 3 pools (r^2 = 0.997), and low-moderate density of bank cover in the JCW type 1 pools (r^2 = 0.589).

Fish species-habitat variable associations

Two fish species had significant associations with two (r^2 = 0.590). The round goby was positively associated with moderate depth (r^2 = 0.848) in type 3 pools in the entire JCW watershed.

DISCUSSION

Catch per unit effort-habitat variable associations

We attempted to control for natural and biased variability

Table 3. Number of fish sampled (N = 70 species) in the Tonawanda Creek (TCW) and Johnson Creek (JCW) watersheds. Species are listed in phylogenetic order from primitive to advanced.

Common name	Scientific name	TCW	JCW	Both
GARS	**LEPISOSTEIDAE**			
Longnose gar	*Lepisosteus osseus*		26	26
Bowfins	**Amiidae**			
Bowfin	*Amia calva*		33	33
Herrings	**Clupeidae**			
Alewife	*Alosa pseudoharengus*		24	24
Gizzard shad	*Dorosoma cepedianum*	1		1
Trouts	**Salmonidae**			
[1]Rainbow trout	*Oncorhynchus mykiss*		1	1
[1]Brown trout	*Salmo trutta*	36		36
Rainbow smelt	*Osmerus mordax*	1		1
Mudminnows	**Umbridae**			
Central mudminnow	*Umbra limi*	11	2	13
Pikes	**Esocidae**			
Grass pickerel	*Esox americanus vermiculatus*	7	1	8
Northern pike	*Esox lucius*	39	18	57
Chain pickerel	*Esox niger*	1		1
Minnows and carps	**Cyprinidae**			
Central stoneroller	*Campostoma anomalum*	297	431	728
[1]Goldfish	*Carassius auratus*	5		5
Redside dace	*Clinostomus elongatus*	15		15
[1]Common carp	*Cyprinus carpio*	269	183	452
Hornyhead chub	*Nocomis biguttatus*	477	32	509
River chub	*Nocomis micropogon*	1,146		1,146
Golden shiner	*Notemigonus crysoleucas*	150	28	178
Emerald shiner	*Notropis atherinoides*	342	381	723
Striped shiner	*Luxilus chrysocephalus*	1,854	172	2,026
Common shiner	*Luxilus cornutus*	80	5	85
Bigmouth shiner	*Notropis dorsalis*	23		23
Spottail shiner	*Notropis hudsonius*	4		4
Rosyface shiner	*Notropis rubellus*	467	12	479
Spotfin shiner	*Cyprinella spiloptera*	475	236	711
Sand shiner	*Notropis stramineus*	134		134
[2]Redfin shiner	*Lythrurus umbratilis*	6	57	63
Mimic shiner	*Notropis volucellus*	553	59	612
Bluntnose minnow	*Pimephales notatus*	2,518	784	3,302
Fathead minnow	*Pimephales promelas*	2,130	17	2,147
Longnose dace	*Rhinichthys cataractae*	95		95
W. blacknose dace	*Rhinichthys obtusus*	1,364	3	1,367
[1]Rudd	*Scardinius erythrophthalmus*	13		13
Creek chub	*Semotilus atromaculatus*	215	541	756

Table 2. Contd.

Fallfish	*Semotilus corporalis*	201		201
Suckers	**Catostomidae**			
White sucker	*Catostomus commersonii*	665	304	969
Northern hog sucker	*Hypentelium nigricans*	595	346	941
Silver redhorse	*Moxostoma anisurum*	19		19
Golden redhorse	*Moxostoma erythrurum*	160	54	214
Shorthead redhorse	*Moxostoma macrolepidotum*	27	9	36
Greater redhorse	*Moxostoma valenciennesi*	14		14
N. American catfishes	**Ictaluridae**			
Yellow bullhead	*Ameiurus natalis*		1	1
Brown bullhead	*Ameiurus nebulosus*	48	67	115
Channel catfish	*Ictalurus punctatus*	13		13
Stonecat	*Noturus flavus*	35	10	45
Tadpole madtom	*Noturus gyrinus*	1	3	4
Brindled madtom	*Noturus miurus*	21	9	30
Topminnows	**Fundulidae**			
Banded killifish	*Fundulus diaphanus*	1		1
New world silversides	**Atherinopsidae**			
Brook silverside	*Labidesthes sicculus*	5	2	7
Temperate basses	**Moronidae**			
White perch	*Morone americana*		1	1
Sunfishes	**Centrarchidae**			
Rock bass	*Ambloplites rupestris*	940	297	1,237
Green sunfish	*Lepomis cyanellus*	537	212	749
Pumpkinseed	*Lepomis gibbosus*	677	181	858
Bluegill	*Lepomis macrochirus*	189	23	212
[2]Longear sunfish	*Lepomis megalotis*	23		23
Smallmouth bass	*Micropterus dolomieu*	430	120	550
Largemouth bass	*Micropterus salmoides*	532	178	710
White crappie	*Pomoxis annularis*	7		7
Black crappie	*Pomoxis nigromaculatus*	16		16
Perches	**Percidae**			
Greenside darter	*Etheostoma blennioides*	144	86	230
Rainbow darter	*Etheostoma caeruleum*	108		108
Fantail dater	*Etheostoma flabellare*	178	34	212
Johnny darter	*Etheostoma nigrum*	2,075	820	2,895
Yellow perch	*Perca flavescens*	28	198	226
Logperch	*Percina caprodes*	249	27	276
Blackside darter	*Percina maculata*	491	141	632
Walleye	*Sander vitreum vitreum*	11		11

Table 2. Contd.

Drums	**Sciaenidae**			
Freshwater drum	*Aplodinotus grunniens*	2	12	14
Gobies	**Gobiidae**			
[1]Round goby	*Neogobius melanostomus*	63	37	100
Sculpins	**Cottidae**			
Mottled sculpin	*Cottus bairdii*	77		77
	Totals	**21,310**	**6,218**	**27,528**

[1]Non-native species: New York State Department of Environmental Conservation Fisheries Database CD (not online but see http://www.dec.ny.gov/docs/wildlife_pdf/oct11hlite.pdf for a description). [2]Native species listed as “Threatened” in New York State (http://www.dec.ny.gov/animals/7494.html). [3]Native species of “Special Concern” in New York State (http://www.dec.ny.gov/animals/7008.html)

Table 4. Multiple linear regression results from analyses of the Tonawanda and Johnson Creek watersheds. Criteria for including the results below: 1) variables in best subsets regression models had adj-r^2 values >20% and $P \leq 0.06$, and 2) variables in backward stepwise linear regression models had $P \leq 0.001$ (Bonferroni-corrected value of α).

[1]Scale	[2]Sites	[3]m-CP	[3]r^2	[4]Dep	[4]Ind	[4]Assoc	[5]Mean (SE)	[5]Habitat	[6]r^2	[6]P-value
TCW	68	2.7	0.398	CPUE	PT	pos	2.51(0.153)	Type 3	0.371	<0.001
JCW	40	3.2	0.280	CPUE	PT	pos	2.40(0.195)	Type 2	0.217	0.001
J-UWS	16	2.0	**0.628**	CPUE	SS	pos	2.1(0.138)	Sand	**0.628**	0.001
J-PT3	8	4.1	**0.997**	CPUE	IW	neg	2.5(0.189)	Low-mod	**0.997**	<0.001
J-PT1	14	0.4	**0.589**	CPUE	BC	neg	2.6(0.133)	Low-mod	**0.589**	<0.001
TCW	68	2.7	0.398	CPUE	AV	pos	2.21(0.137)	Low	0.371	<0.001
T-M+U	19	3.5	**0.753**	CPUE	AV	pos	2.4(0.325)	Low	**0.703**	<0.001
T-PT4	23	2.4	0.409	CPUE	AV	pos	2.5(0.280)	Low	0.409	<0.001
J-PT3	8	4.1	**0.997**	CPUE	AV	neg	3.1(0.441)	Mod	**0.997**	<0.001
JCW	40	3.4	0.364	Etni	PT	pos	2.87(0.196)	Type 3	0.348	<0.001
J-LWS	24	3.5	**0.590**	Etni	PT	pos	2.37(0.232)	Type 2	**0.590**	<0.001
J-PT3	8	1.3	**0.848**	Neme	MD	pos	3.00(na)	Mod	**0.848**	<0.001

[1]The watersheds were divided into spatial subunits as shown in Table 1. [2]Number of sites sampled per spatial scale as shown in Table 2.
[3]Results for best subsets regression (BSR): m (Mallow’s)-CP score and adjusted r^2 values (r^2). [4]Dependent (Dep) and independent (Ind) variables, and the direction of associations (Assoc) between them, in statistically significant backward stepwise regression models (SWR). Independent variables are pool type (PT), substrate score (SS), instream wood (IW), bank cover (BC), aquatic vegetation (AV) and maximum depth (MD). CPUE = catch per unit effort; Etni = johnny darter; Neme = round goby. [5]Mean habitat condition scores and their standard errors (SE) for sites sampled in each pool type (For example, 2 or 3) or spatial scale (e.g., TCW, J-PT3). Low and mod (moderate) vs. high are qualitative descriptors of the habitat scores. [6]Results for backward stepwise linear regression: adj-r^2 value (r^2) and the Bonferroni corrected value of α (P): original α (0.06) / 61 significant models = adjusted α ≤ 0.001.

of CPUE in three ways: 1) Segregating and evaluating data at spatial (watershed, sub-watershed, tributary) and habitat (pool type) scales, 2) Measuring and coding habitat variables in a consistent, semi-quantitative way (all done by the first author), and 3) Using the same electrofishing and seining methods and effort to collect fish at each site. Ultimately, however, habitat conditions and the different susceptibilities of species to sampling gears determine CPUE (Murphy and Willis, 1996; Boner et al., 2009). Fish are easier to sample in shallow water (Green, 1989; Murphy and Willis, 1996), which often limits access by larger fish (Butler and Fairchild, 2005; Gillette et al., 2005; Sharma and Jackson, 2007), resulting in disproportionate catches of smaller species. Extensive sampling by boat electrofishing (no seining possible) in the expansive and deep lower subwatershed of the TCW (Erie Canal) and JCW (drowned river mouth confluence with Lake Ontario) likely missed some fishes that were too deep or widely scattered for effective electrofishing. In contrast, CPUE increased as water depth decreased upstream.

In the TCW and JCW, CPUE was positively associated with type 3 and 2 pools, respectively. We often found greater fish abundance and species richness in the long and winding TCW where pools were associated with runs (type 3). In the much smaller JCW, isolated pools, mostly those in the lower subwatershed without adjacent runs or riffles (type 2), were associated with higher CPUE.

Substrate size influences CPUE because it is a primary component of habitat formation and alteration in flowing waters (Hunter, 1991; Gillette et al., 2005; Lau et al., 2006). It often dictates fish assemblage structure, especially in lotic systems (Talmage et al., 2002; Lau et al., 2006; Sharma and Jackson, 2007). Substrate size also can alter the effectiveness of certain gear types (Gillette et al., 2005; Van Snik Gray et al., 2005; Sharma and Jackson, 2007) which, due to high seining efficiency, may be why CPUE was positively associated with sandy substrate in the upper JCW sub-watershed.

Woody debris typically influences CPUE (Talmage et al., 2002; Powers et al., 2003; Lau et al., 2006) by diminishing the effectiveness of sampling, especially seining (Murphy and Willis, 1996; Flosi et al., 1998; Powers et al., 2003). However, large woody debris often provides optimal fish habitat in streams (Angermeier and Karr, 1984; Flosi et al., 1998), especially for young fishes (Trautman, 1981; Gregory and Bisson, 1997; Flosi et al., 1998). In type 3 pools of the JCW, instream wood had a negative influence on CPUE, possibly due to the difficulty of sampling deeper pools containing large woody debris.

Bank cover influences CPUE (Whitton, 1975; Madejczyk et al., 1998; Butler and Fairchild, 2005) by reducing sampling effectiveness when fish hide in hard to reach places (for example, undercut banks). Bank cover is important for creating microhabitats for stream fishes (Platts et al., 1983; Murphy and Willis, 1996; Talmage et al., 2002), especially as critical rearing habitat for young stream fishes (Trautman, 1981; Hunter, 1991) and a food source via terrestrial drop-ins (Hunter, 1991; Closs et al., 2004). CPUE and bank cover in type 1 pools of the JCW were negatively associated, likely due to poor habitat quality of type 1 pools (channelized, lowest complexity).

Aquatic vegetation often influences CPUE by reducing sampling effectiveness (Whitton, 1975; Ray et al., 2004; Van Snik Gray, et al. 2005). This problem is pronounced in areas of heavy weed growth, such as in-stream, slack water impoundments (for example, Lyndonville Pond, JCW; see Wells, 2009), which may impact foraging efficiency and reduce dissolved oxygen (Brazner and Beals, 1997), especially at night. However, aquatic vegetation provides important shelter and food for many aquatic organisms (Flosi et al., 1998; Van Snik Gray et al., 2005; Lau et al., 2006) and essential habitat for many fishes that require it for at least part of their life cycle (Whitton, 1975; Van Snik Gray et al., 2005; McGarvey and Hughes, 2008). In the typically turbid TCW, aquatic vegetation (usually emergents) had a positive influence on CPUE at three spatial scales (watershed, middle + upper subwatershed, and type 4 pools), likely due to the added cover along stream margins utilized by the fishes sampled. However, in the less turbid JCW submergent vegetation in type 3 pools had a negative influence on CPUE; reasons for this result are unclear.

Fish species-habitat variable associations

Stepwise regression suggested habitat associations for two stream generalists. The johnny darter occurs in many stream habitats (Scott and Crossman, 1973; Trautman, 1981; Smith, 1985), often adjacent to currents (Scott and Crossman, 1973; Miller and Robison, 1973; Knopf, 2002), and occasionally in pools near current edges (Miller and Robison, 1973). It is more tolerant of slow water than other darters (Scott and Crossman, 1973; Smith, 1979; Trautman, 1981). These conditions were common in the JCW, which is probably why the johnny darter was significantly associated with type 3 and 2 pools in the entire and lower subwatershed of the JCW, respectively.

The invasive round goby is a benthic habitat generalist (Jude et al., 1992; Lever, 1996; Hubbs and Lagler, 2004), ecologically similar to the mottled sculpin (*Cottus bairdii*) (Lever, 1996; Vanderploeg et al., 2002), and seemingly well adapted for life in North American streams (no air bladder). During the survey period in 2005, round goby abundance was low (Table 3), likely due to its recent invasion of the study area, where it had not been recorded before this study, from the Erie Canal upstream and Lake Ontario downstream. There was a negative association ($P = 0.003$; adj. $r^2 = 0.398$) with distance upstream from Lake Ontario in lower main stem of Johnson Creek.

Habitat modeling summary

Like Butler and Fairchild (2005), MLR models were interpreted as if fish species (and fish assemblages) were using the habitat they were captured in, thus species-habitat correlations were assumed for fish occupying sites with such habitat. Using a very conservative Bonferroni-corrected α-value of 0.001, we identified nine literature-supported habitat influences on fish assem-blage CPUE but only three associations for two of the 70 fish species sampled. Before Bonferroni correction, 80 habitat variables in 61 significant SWR models were associated with 43 species and 37 assemblage (CPUE, richness, Simpson's diversity) variables (Wells, 2009). The analyses reported here have a 0.05 study-wise risk of a Type I error and an Unknown, but likely high, risk of Type II error. With the necessary Bonferroni constraint, the power of MLR models in our study to predict associations of stream fish assemblage variables and species with specific habitat features was disappointingly limited.

Similar studies also have indicated scale-specific and habitat-specific relationships between fish species and fish assemblage parameters. Talmage et al. (2002) reported that local-scale factors, often reflective of past and present watershed disturbances (for example, farming, chan-nelizing), are important to fish communities. Van Holt et al. (2006) showed a positive impact of riparian forest on fish assemblages at multiple scales. Angermeier and Schlosser (1989) indicated that effects of site volume and complexity were significant for fish abundance in pools but species richness was related more to the nature of riffles. According to Ray et al. (2004), percent cover of aquatic vegetation explained much of the variation in fish diversity. Talmage et al. (2002) reported that relationships between fish communities and variability of instream habitat were positive and linear. Van Holt et al. (2006) found significance in pool variability and available cover, and suggested that their results showed variation of instream habitat structure was important. Also according to Van Holt et al. (2006), the variance explained differed among models by the spatial scale of analysis. In models for large watersheds, variance was associated with fish richness but in models at smaller scales variance was explained best by fish diversity (richness and evenness).

In contrast to the results reported here, Butler and Fairchild (2005) reported that individual species, not fish assemblages, were more associated with specific habitat variables. In addition, Van Holt et al. (2006) reported that no single model predicted fish assemblages well but that high statistical variability in a model predicted fish diversity and that there was a negative relationship between available cover and the number of intolerant species predicted (both as percentages). Others reported that fish assemblage composition is influenced by local habitat complexity (Gorman and Karr, 1978, Schlosser, 1982; Barko et al., 2004). Microhabitat specialization created through adaptive or opportunistic use of available habitats by stream fishes is probably a key component of a species' success (Barko et al., 2004; Rippe, 2005), which likely limits the predictive power of MLR models. Of the 39 fish species analyzed in this study, 67% are habitat generalists (Wells, 2009), presenting a substantial challenge to identifying statistically significant associations with specific habitat variables.

Research and management

Localized focus on species-specific management has shifted to a broader eco-region scale, and watershed analysis of fish assemblages is now common (Fausch et al., 2002; Guy and Brown, 2007, Bonar et al., 2009). Small-scale (this study was large scale, covering most of two watersheds) or short-term (a drawback of this two summer study) studies are largely ineffective for providing managers with information and tools at the scales needed to conserve stream fish populations and communities (Fausch et al., 2002). A lack of knowledge of many stream fishes and their habitats underline the importance of studying habitat heterogeneity on larger spatial and temporal scales in these linear aquatic habitats (Fausch et al., 2002).

Ono et al. (1983) remarked that every species may be necessary to keep an ecosystem intact. Notable declines of many native stream fishes in New York State (Carlson and Daniels, 2004; Carlson, 2005; Wells and Haynes, 2006) have increased awareness of the need to pursue conservation for lesser known or imperiled fishes. The conservation of stream fishes is an evolving science and requires assessment of entire fish assemblages on different spatial scales. With ever-increasing anthropogenic demands being placed on watersheds across the globe, a comprehensive and proactive approach to stream fish management is needed now more than ever to assess and protect important aquatic habitats and prevent further extirpation of native stream fishes. Focusing on riparian corridor restoration and management (particularly erosion control) will best enhance fish habitat in the agriculturally-influenced, warm water streams of western New York State.

ACKNOWLEDGMENTS

We thank Douglas M. Carlson (NYSDEC) for encouragement, advice, sharing of knowledge and editing. Ross Abbett gave extraordinary assistance in the field and laboratory. Hilary Richardson, Patrick Herbert, Coral Reina, Yorr Marchione, William Guenther, Crystal Kerr, Charles Mangan, Michael Koch, Brendan Farrell, Caleb Snyder, Linda Driscoll, Paul Wiedenmeyer, Nicholas Herbert, Todd Barton, Tracie Wells, Jordin Kelly, Cheryl Scott, Jevon Scott, Michael Wells, Christopher Wells, Thomas Kowaleski, Eric Kowaleski, Fenton Caster and Catherine Wells-Kelly provided assistance in the field. Thomas Bedard, Christopher Van Maaren and Robert D'Argenio (NYSDEC Region 6); Michael Wilkinson (NYSDEC Region 9) and Webster Pearsall (NYSDEC Region 8); Jason Becker (Michigan DPR) and Thomas Goniea (Michigan DNR); Michael Goehle, Michael Weimer, Elizabeth Trometer and Emily Zollweg (U.S. Fish and Wildlife Service); and William Snyder (SUNY Morrisville) provided additional advice and assistance. Special thanks go to the Tonawanda-Seneca Nation for allowing access to tribal lands, and for survey assistance provided by Linda Logan, Craig Jonathan and Roger Hill.

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