Evaluating Effects of Water Withdrawals and Impoundments on Fish Assemblages in Connecticut Streams

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Executive Summary

The natural flow regime is regarded as a master variable in structuring the physical and biological condition of stream ecosystems. Anthropogenic activities of various forms have altered the natural flow regime. Large proportions of stream fishes are imperiled and of conservation concern worldwide and the protection of aquatic ecosystems under an expanding human population has become a global concern. Water withdrawals are a major anthropogenic activity that directly impacts the flow regime and are predicted to expand in frequency and magnitude as human population density increases. Water quantity is increasingly becoming a major issue for urbanizing landscapes in the northeastern United States, even though water has traditionally been considered plentiful. This perspective is particularly acute for the state of Connecticut, which is currently undertaking a process to revise instream flow regulations.

The purpose of this study was to evaluate the ecological effects of water withdrawals and impoundments on fish assemblage structure in smaller, wadeable streams in Connecticut (watershed sizes of approximately 5-80 km^2 draining streams of 2-4th order). We selected 33 stream reaches across Connecticut located directly downstream of water withdrawals for municipal and agricultural water supply. Study streams included two types of water withdrawals; one group had dams and associated impoundments (hereafter, impoundment sites), and the second were streams without such in-stream structures, i.e., unimpounded streams with pumping wells near the channel in the floodplain (hereafter, intake sites). Streams without withdrawals (hereafter, reference sites) were also sampled for comparison.

Fish data were collected between May and August in 2007-2008 by electrofishing a stream distance of 50 times the mean wetted-channel width at each study site. Local-scale habitat data (stream depth, velocity, etc.) were collected in the field, and watershed-scale habitat data (% forest cover, drainage area, etc) were calculated using existing digital maps and spatial data in program ArcGIS. A measure of water withdrawal magnitude, Withdrawal Index (WI), was defined as the maximum permitted daily withdrawal rate (in millions gallons per day, mgd) divided by the estimated seven-day, ten-year recurrence low flow (7Q10). Data were analyzed using a Generalized Linear Model (GLM) form of regression.

The results showed that water withdrawal magnitude was associated with several aspects of fish assemblage composition and structure. Stream sites with high withdrawal rates were characterized with lower proportions of fluvial dependent fishes (fish which need flowing water to complete a portion of their life history) and benthic invertivores (fish which feed on bottom-dwelling stream insects), and had a greater percent composition of macrohabitat generalists, particularly members of the family Centrarchidae (which includes the sunfish species and black basses). Increasing withdrawal rate generally resulted in an accelerated percent decrease in fluvial dependent individuals, benthic invertivore individuals, and white sucker individuals, with corresponding increases in macrohabitat generalist individuals, warmwater individuals, non-tolerant generalist feeders, tolerant individuals, and family Centrarchidae individuals. The effect was not necessarily linear; that is, fish assemblages showed less alteration when the withdrawal rate was small, and the effect accelerated with increasing withdrawal. For example, the proportion of benthic invertivores decreased from 27% to 24% as permitted withdrawal increased from 0 (no withdrawal) to 10; after which the percent was predicted to be 17% and 11%, respectively, as WI increased to 50, and 100.

Our results are consistent with ecological theory that alteration of the natural flow regime will impact stream biota. While the current study focused on fish assemblage composition and structure as a first step to describe general relationships, we note that the impact of water withdrawal may be species-specific and encourage further research. In summary, we found that water withdrawals have contributed to measurable alterations of fish assemblages and should therefore be considered in regulation and aquatic conservation.

Introduction

The natural flow regime as represented by the characteristic annual hydrograph is regarded as a master variable in forming and maintaining stream habitat and elements of biological assemblages (Poff et al. 1997). The condition of stream ecosystems then depends on the appropriate quantity, quality, timing and temporal variability of water flow, and aquatic species have evolved life history strategies to adapt to a natural flow regime (Poff et al. 1997; Bunn and Arthington 2002). The natural flow regimes may differ among watersheds and regions, but occur in a natural or least-altered landscape (Richter et al. 1996; Poff et al. 1997; Roy et al. 2005; Nelson and Palmer 2007). Anthropogenic activities of various forms have altered the natural flow regime (Bunn and Arthington 2002). Flow regime altering activities are rapidly expanding in developing areas and the protection of aquatic ecosystems under an expanding human population is a global concern (Baron et al. 2002; Brasher 2003; McIntosh et al. 2008).

Water withdrawals are a major anthropogenic activity that can directly impact stream flow. Reduced discharge has been shown to impact fish size (Walters and Post 2008) and occurrence of fluvial specialist fish species that require lotic habitats for at least part of their life history cycles (Armstrong et al. 2001; Freeman and Marcinek 2006). As a general pattern, stream discharge has been positively associated with species richness (Xenopoulos and Lodge 2006; Shea and Peterson 2007). Water reduction in the stream channel results in loss of habitat volume and can reduce connectivity (Labbe and Fausch 2000), and indirectly affects water quality and food resources (Lake 2003). When water withdrawals involve in-stream structures (i.e., dams and reservoirs), the ecological changes can be more complicated and severe than direct pumping or diversion of water from unimpounded streams (Freeman and Marcinek 2006). Either form of water withdrawal can be catastrophic, however, if the magnitude is great enough, and examples of ecosystem collapse through stream drying are numerous (Sophocleous 2007; Winter 2007).

Water quantity is increasingly becoming a major issue for urbanizing landscapes such as those in the northeastern United States, even though water has not traditionally been considered limited (Armstrong et al. 2001). This perspective is particularly acute for Connecticut, which is currently undertaking a process to revise instream flow regulations (CT DEP 2009). The use of fish assemblages for the determination of stream condition relative to aquatic life use support as required by the federal clean water act (i.e. biomonitoring) has been difficult in southern New England for a couple reasons. First, the Northeastern landscape was glaciated, and lacking well-connected refugia freshwater fishes had limited opportunities to recolonize. As a result, this region is characterized with a species-poor freshwater fauna relative to the Mid-Atlantic and southeastern United States. Second, extensive human occupation and landscape modification makes defining natural or reference watersheds difficult (Stoddard et al. 2006). For these reasons, quantifying fish assemblage responses to anthropogenic disturbances poses a substantial challenge, however, recent work on biological monitoring indicates that anthropogenic impacts can be associated with fish assemblage structure and composition (Bain and Meixler 2008; Kanno et al., in review).

The purpose of this study was to evaluate the ecological effects of water withdrawals and impoundments on fish assemblage structure in smaller, wadeable streams (watershed sizes of approximately 5-80 km^2 draining streams of 2-4th order). Connecticut does not allow drinking water supply abstraction from waters receiving treated wastewater discharge, so abstraction generally occurs upstream in smaller watersheds. Study reaches were located directly downstream of water withdrawals for municipal and agricultural water supply. Study streams included two types of water withdrawals; one group had dams and associated impoundments (hereafter, impoundment sites) and the other were streams without such in-stream structures, i.e., unimpounded streams with pumping wells near the channel in the floodplain (hereafter, intake sites). Streams without withdrawals were also sampled for comparison. Of course, fish assemblages are structured by other environmental factors besides water withdrawal and the presence of a dam; many of these factors are natural, such as stream channel gradient and elevation (Maret et al. 1997; Waite and Carpenter 2000). Drainage area and associated stream size is known to describe natural upstream-downstream assemblage differences (Sheldon 1968; Rahel and Hubert 1991; Walters et al. 2003). Surficial geology influences both channel geomorphology and the ability of groundwater to enter streams (Walters et al. 2003; Winter 2007). Prominent among the anthropogenic factors are landcover and landuse. In the eastern USA, the percent of the landscape developed has been shown to have strong effects (Wang et al. 2000; Wenger et al. 2008). Therefore, our evaluation of water withdrawal will necessarily involve these other factors known to affect fish assemblages.

Methods

Study stream selection

This study targeted 33 stream reaches across Connecticut for sampling (Figure 1; Table 1). Study streams included 16 impoundment sites and 11 intake sites, and we additionally sampled 6 sites that were not subject to water withdrawals (hereafter termed reference sites). Study sites were selected based on records of permitted and actual water withdrawals obtained from the Connecticut Department of Environmental Protection, Bureau of Water Protection and Land Reuse. The majority of the study streams were located downstream of water withdrawals for municipal water supply, but also included two streams in which water was abstracted for agricultural use (Freshwater Brook (INT 03), Hungary Brook (INT 05)). Reference sites were selected to be comparable in size and landcover characteristics and were not specifically intended to be considered "least-altered" or somehow indicative of streams with the greatest available biological integrity (Stoddard et al. 2006), although one site was identified as a least-disturbed stream in a recent regional evaluation (Whiting River (REF 03); Bellucci et al. 2009).

Field procedures

Field survey was conducted between May and August in 2007 and 2008. Fish were sampled with pulsed DC electrofishing with a Smith-Root (Vancouver, Washington) model LR-24 back-pack unit or a Smith-Root tote-barge electrofishing unit controlled by a Coffelt VVP-15 control unit and powered by a 3600-watt generator. Efforts were made to conduct sampling immediately downstream of water withdrawals (*sensu* Freeman and Marcinek 2006). This was not feasible for 3 sites (Wigwam Brook (IMP 10), Converse Pond Brook (IMP 12), Beacon Hill Brook (IMP 16)), and the sampled reaches were moved downstream but were in all cases within 2.5km of permitted withdrawals. The sampling distance extended 50 times the mean wetted-channel width to adequately represent species richness and composition. Stream reaches 40 times the mean wetted width during summer base flow conditions are recommended by the United State Environmental Protection Agency to capture the majority of fish species (Peck et al. 2006). However, this sampling distance may still underestimate true species richness in many streams (Kanno et al., *In Press*), so we chose to sample slightly longer reaches. On three occasions, shorter sections were sampled due to habitat conditions or logistical problems (Menunketesuck River (IMP 05), Mill River (INT 11), Whiting River (REF 03)). At each site, a crew of three people conducted one-pass electrofishing, starting from the downstream end and proceeding upstream by sampling all kinds of available microhabitats (e.g.,

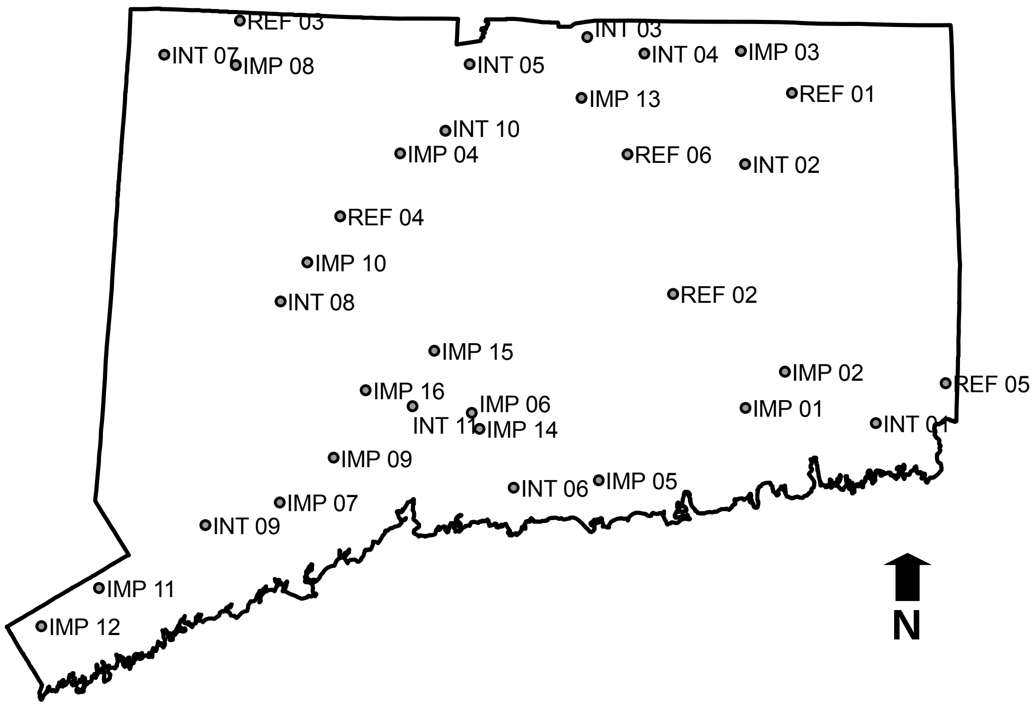

Figure 1. Location of study sites surveyed in 2007 and 2008 in Connecticut. Site abbreviations are: IMP = impoundment sites, INT = intake sites, and REF = reference sites (see Table 1 for site ID).

run, riffle, pool). Collected fish were identified to species, measured for length in the field and then were released back to the stream.

Local-scale habitat data were collected on the same day of fish survey, in order to provide descriptive statistics at each study site. Stream width, depth, and velocity were recorded at 8 transects spread across the study site. Stream depth and velocity were measured at three equally spaced points across each transect (0.25, 0.5, and 0.75 times wetted width). Velocities were measured with a Marsh-Mcbirney Flo-mate 2000 model current meter attached to a top-setting wading rod. Stream discharge was measured at a single transect where velocities were relatively smooth (Bain and Stevenson 1999) using the velocity-area current meter method as described by Gordon et al. (2004) with verticals spaced 0.5m or less depending on channel complexity. Water temperature was measured in the morning hours (9:00-11:00 am).

Data analyses

Site characteristics

Local-scale habitat data (stream width, depth, velocity, discharge, and water temperature) were compared between impoundment, intake and reference sites with analysis of variance (ANOVA). We used PROC MIXED in program SAS (version 9.1, SAS Institute Inc., Cary, NC). Data were tested for normality and

Table 1. Study sites used in the current study. For Site ID; INT = intake sites, IMP = impoundment sites, REF = reference sites.

SiteID	Name	Basin ID	Latitude	Longitude	Withdrawal Permit holder	Survey date
INT 01	Whitford Brook	2104	41.4127	-71.9596	Aquarion	7/9/2007
INT 02	Fenton River	3207	41.8114	-72.2226	University of Connecticut	7/25/2007
INT 03	Freshwater Brook	4003	42.0074	-72.5454	POLEK	8/17/2007
INT 04	Gulf Stream	4203	41.9816	-72.4285	CT Water Company	6/22/2007
INT 05	Hungary Brook	4320	41.9656	-72.7870	Imperial Nurseries	8/9/2007
INT 06	West River	5100	41.3167	-72.6976	CT Water Company	6/28/2007
INT 07	Moore Brook	6006	41.9786	-73.4147	Aquarion	7/19/2007
INT 08	Nonnewaug River	6802	41.6011	-73.1738	Watertown Fire District	7/24/2007
INT 09	Aspetuck River	7202	41.2576	-73.3241	Aquarion	8/6/2007
INT 10	Stratton Brook	4318	41.8635	-72.8373	Aquarion	7/25/2008
INT 11	Mill River	5301	41.4410	-72.9034	South Central CT Regional Water	7/23/2008
IMP 01	Latimer Brook	2202	41.4375	-72.2253	New London Water Department	8/7/2007
IMP 02	Stony Brook	3104	41.4929	-72.1441	Town of Norwich	7/12/2007
IMP 03	Roaring Brook	3104	41.9848	-72.2298	CT Water Company	6/15/2007
IMP 04	Nepaug River	4310	41.8286	-72.9301	MDC	8/22/2007
IMP 05	Menunketesuck River	5103	41.3274	-72.5246	CT Water Company	6/26/2007
IMP 06	Muddy River	5208	41.4310	-72.7825	Wallingford Water Department	8/15/2007
IMP 07	Farmill River	6025	41.2930	-73.1732	Aquarion	7/6/2007
IMP 08	Wangum Lake River	6202	41.9633	-73.2681	Aquarion	7/13/2007
IMP 09	Beaver Brook	6900	41.3617	-73.0640	South Central CT Regional Water	7/31/2007
IMP 10	Wigwam Brook	6910	41.6616	-73.1189	Waterbury Water Bureau	8/2/2007
IMP 11	Rippowam River	7404	41.1601	-73.5391	Aquarion	8/13/2007
IMP 12	Converse Pond Brook	7410	41.1001	-73.6550	Aquarion	6/25/2007
IMP 13	Broad Brook (E Windsor)	4206	41.9138	-72.5574	CT Water Company	8/30/2008
IMP 14	Farm River	5112	41.4066	-72.7669	South Central CT Regional Water	7/18/2008
IMP 15	Broad Brook (Cheshire)	5204	41.5262	-72.8591	Meriden Water Department	5/28/2008
IMP 16	Beacon Hill Brook	6918	41.4656	-72.9988	CT Water Company	5/22/2008
REF 01	Branch Brook	3203	41.9199	-72.1256	NA	6/17/2007
REF 02	Jeremy River	4705	41.6129	-72.3715	NA	7/16/2007
REF 03	Whiting River	6101	42.0313	-73.2600	NA	8/3/2007
REF 04	Rock Brook	6908	41.7324	-73.0521	NA	6/29/2007
REF 05	Green Fall River	1002	41.4727	-71.8164	NA	7/22/2008
REF 06	Tankerhoosen River	4503	41.8271	-72.4638	NA	7/14/2008

homogeneity of variance with PROC UNIVARIATE and PROC GLM, respectively, in SAS, and data were transformed prior to analysis when assumptions of normality and heteroscedasticity were violated. Landscape variables were similarly compared between the stream groups. For each stream site, upstream drainage area was delineated based on the 30-m resolution National Elevation Dataset using program ArcGIS version 9.2 and ArcHydro version 1.2 (ESRI, Redlands, California). The proportion of forested land within drainage areas was calculated using the 2001 National Land Cover Dataset (Homer et al. 2007). The proportion of forested land was highly correlated with the proportion of impervious surface (r = -0.745; P < 0.0001) and population density (r = -

0.561; *P* < 0.0007); therefore we only used percent of forest cover in subsequent analyses. The percent of upstream drainage area underlain by coarse-grained stratified drift was also calculated with an existing surficial materials GIS data layer available from CT DEP and is known to be correlated to the potential for groundwater inputs to streams (Cervione, Jr., et al. 1982).

The magnitude of water withdrawals needed to be estimated since stream flow was not gauged in the study streams and reporting of actual withdrawal amounts is not mandatory in Connecticut. Therefore, water withdrawal magnitude was indexed by potential withdrawal relative to stream size. Following Freeman and Marcinek (2006), a Withdrawal Index (WI) was calculated for each site as the maximum permitted daily withdrawal rate (in millions gallons per day, mgd) divided by the estimated seven-day, ten-year recurrence low flow (7Q10). Permitted withdrawal rates were used because actual withdrawal rates were not available from all study sites. As seen in Georgia (Freeman and Marcinek 2006), we hypothesized a strong correlation between permitted and actual rates. Actual withdrawals were known for some sites, and were provided by certain permit holders for the 3-year period prior to fish sampling. The 7Q10 was estimated using the regression method of Cervione, Jr., et al. (1982):

$$Q_{7,10} = 0.67 \times A_{sd} + 0.01 \times A_{till}$$

Where $Q_{7,10}$ is the 7-day, 10-year low flow (in cubic feet per second), A_{sd} is the upstream drainage area underlain by coarse-grained stratified drift (in square miles), and A_{till} is the upstream drainage area underlain by till-mantled bedrock (in square miles). The 7Q10 flow was then expressed in mgd, so that WI represented a fraction or multiple of the 7Q10 flow.

Water withdrawal and impoundment effects on fish assemblage characteristics

Proportional composition of ecological groupings was qualitatively investigated based on the characteristics of fish species collected (Appendix 2). Each fish species was classified by stream flow, thermal, tolerance, and trophic requirements based on regional references (Whitworth 1996; Halliwell et al. 1999; Armstrong et al. 2001). We constructed stacked bar plots of ecological guilds by stream group.

We quantified if fish assemblage metrics were affected by WI, relative to other natural and anthropogenic variables known to affect fish assemblages. Fifteen fish assemblage metrics were selected from 5 ecological classes: flow-related guild (% fluvial specialist individuals, % fluvial dependent individuals, and % macro-habitat generalist individuals), thermal guilds (% coldwater individuals, % coolwater individuals, and % warmwater individuals), trophic guilds (% benthic invertivore individuals, % non-tolerant general feeder individuals, and % top carnivore individuals), tolerance (% tolerant individuals, and % intolerant individuals), and indicator species/family (% white sucker individuals, % brook trout individuals, % family Cyprinidae individuals, and % family Centrarchidae individuals). In addition to WI, explanatory variables tested included upstream drainage area, percent of forested area, and percent of coarse-grained stratified drift.

Generalized linear models (GLM) were constructed to examine the water withdrawal impact on fish metrics. GLM are robust to assumptions of residual normality and variance homogeneity, and useful for analyzing response variables with a limited range such as proportional data (i.e., 0 – 1). All unique subsets of the five watershed-scale independent variables (i.e., the presence of impoundment, WI, drainage area, % forest, and % stratified drift) were combined to propose 32 regression models (we included a "null model" with only an intercept term). For each of the 15 fish metrics, we fitted the 32 multiple regression models, specifying a gamma distribution and logarithmic link. GLM were fit in program SAS using PROC GENMOD.

Support for candidate models was assessed using an information-theoretic approach (Burnham and Anderson 2002) in which a restricted, small all-subsets candidate set of models were considered. Ranking of candidate models was based on Akaike's Information Criterion corrected for small sample size (AIC_c). The most-supported model had the smallest AIC_c value, and competing models were identified as models with ΔAIC_c < 2 of the highest ranking model. Model selection uncertainty was addressed among competing models by multi-model averaging resulting in model-averaged estimates of regression parameter coefficients. The relative importance of variables found in the competing models was ascertained by summing the AIC_c weights (w_i) of each competing model in which a variable occurred (Burnham and Anderson 2002).

Effect sizes were predicted for those metrics in which WI was included as a parameter in the set of competing models. Simplified regression models were fitted using PROC GENMOD that included WI, an intercept, and a scale term. Coefficients of WI from these simplified models and model averaged coefficients varied < 0.001 and were virtually identical.

Results

Fish assemblage and habitat characteristics

A total of 25 species (excluding stocked salmonids) were recorded from our survey in 2007 and 2008 (Appendices 1 and 2). Blacknose dace *Rhinichthys atratulus*, white sucker *Catostomus commersonii*, longnose dace *Rhinichthys cataractae*, American eel *Anguilla rostrata*, and brook trout *Salvelinus fontinalis* were the five most common species in terms of occurrence among the 33 study sites. A single individual of banded sunfish *Enneacanthus obesus* (a species of special concern in Connecticut) was recorded from Whitford Brook (INT 01).

Watershed- and local-scale habitat variables were not statistically different among impoundment, intake, and reference sites (Table 2; Appendix 3). However, WI was significantly larger for reservoir sites than for intake sites (ANOVA: $F = 12.30$; $P = 0.0017$). The reported permitted withdrawal rate evidently exceeded the physical withdrawal capacities for two sites (Converse Pond Brook (IMP 12) and Farm River (IMP 14)). These sites were assigned a WI value of 130, which was slightly larger than the greatest observed WI value (120 for Wigwam Brook (IMP 10)). Actual water withdrawal records were available for 15 sites, and the permitted daily withdrawal rate had a strong relationship with the maximum daily withdrawal for the most consumptive month in the 3-year period prior to fish sampling ($r^2 = 0.91$; $P < 0.0001$). Therefore, we considered the use of WI based on permitted withdrawal rates was a reasonable index of actual water withdrawals at study streams.

Water withdrawal and impoundment effect on fish assemblages

The mean observed species richness was 8 species (range: 2 – 13) across all study sites. Reference sites had a mean of 7.8 species, impoundment sites had a mean richness of 7.0 and intake sites had a mean richness of 9.7. Qualitatively, assemblage composition tended to differ among stream groups (Figure 2). Impoundment sites were generally characterized by lower proportions of fluvial specialists compared to intake and reference sites, and reference sites had proportionally more Cyprinidae (minnow family) individuals than sites subject to withdrawals.

Withdrawal Index (WI) was included in the competing regression models for eight of the 15 metrics tested (Table 3). These metrics included % fluvial dependent individuals, % macrohabitat generalist individuals, % warmwater individuals, % benthic invertivore individuals, % non-tolerant general feeder individuals, % tolerant individuals, % white sucker individuals, and % Centrarchidae individuals (Table 3). The direc-

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Table 2. Summary of watershed- and local-scale habitat characteristics for impoundment, intake, and reference sites. Values shown are means (± standard deviation).

	Area (km2)	% Forest	% Stratified Drift	Withdrawal Index	Mean width (m)	Mean depth (cm)	Discharge (m3/s)	Water temp (°C)
Impoundment sites	22.97	71	11	53.17	4.5	19.4	0.054	18.8
(N =16)	(± 20.41)	(± 15)	(± 13)	(± 48.68)	(± 1.7)	(± 7.4)	(± 0.096)	(± 3.5)
Intake sites	30.15	70	20	1.69	6.8	20.0	0.100	19.3
(N = 11)	(± 15.62)	(± 15)	(± 16)	(± 1.38)	(± 3.6)	(± 5.4)	(± 0.074)	(± 2.1)
Reference sites	21.78	82	8	N/A	5.2	20.3	0.054	19.9
(N = 6)	(± 8.40)	(± 9)	(± 13)		(± 1.4)	(± 8.1)	(± 0.036)	(± 1.4)

Table 3. Model-averaged coefficients of generalized linear regression analyses for fifteen fish assemblage metrics selected to examine the relative importance of five watershed-scale variables. Blank space indicates that the variable was not included in the competing models used in model averaging.

	Intercept	Dam	Withdrawal index	Drainage area	Forest	Stratified drift
Flow-related guilds						
% Fluvial Specialist	-0.493			-0.005	0.359	
% Fluvial dependent	-1.690		-0.008	0.019		
% Macrohabitat generalist	-1.896		0.017	-0.021		
Thermal guilds						
% Coldwater	-2.168			-0.028	2.607	1.968
% Coolwater	-0.271					-0.723
% Warmwater	-1.475	-1.007	0.009			
Trophic guilds						
% Benthic Invertivore	-1.375		-0.009			
% Non-Tolerant Gen Feeder	-1.953		0.008	-0.023		-1.905
% Top Carnivore	-4.582				3.065	3.658
Tolerance						
% Tolerant	-0.461		0.002			-0.908
% Intolerant	-2.289			-0.028	2.647	2.052
Indicater species/family						
% White sucker	-1.927		-0.004	0.010		
% Brook trout	-1.759			-0.026	1.617	1.589
% Cyprinidae	-0.374					-1.787
% Centrarchidae	-2.499		0.020	-0.020	-1.337	1.616

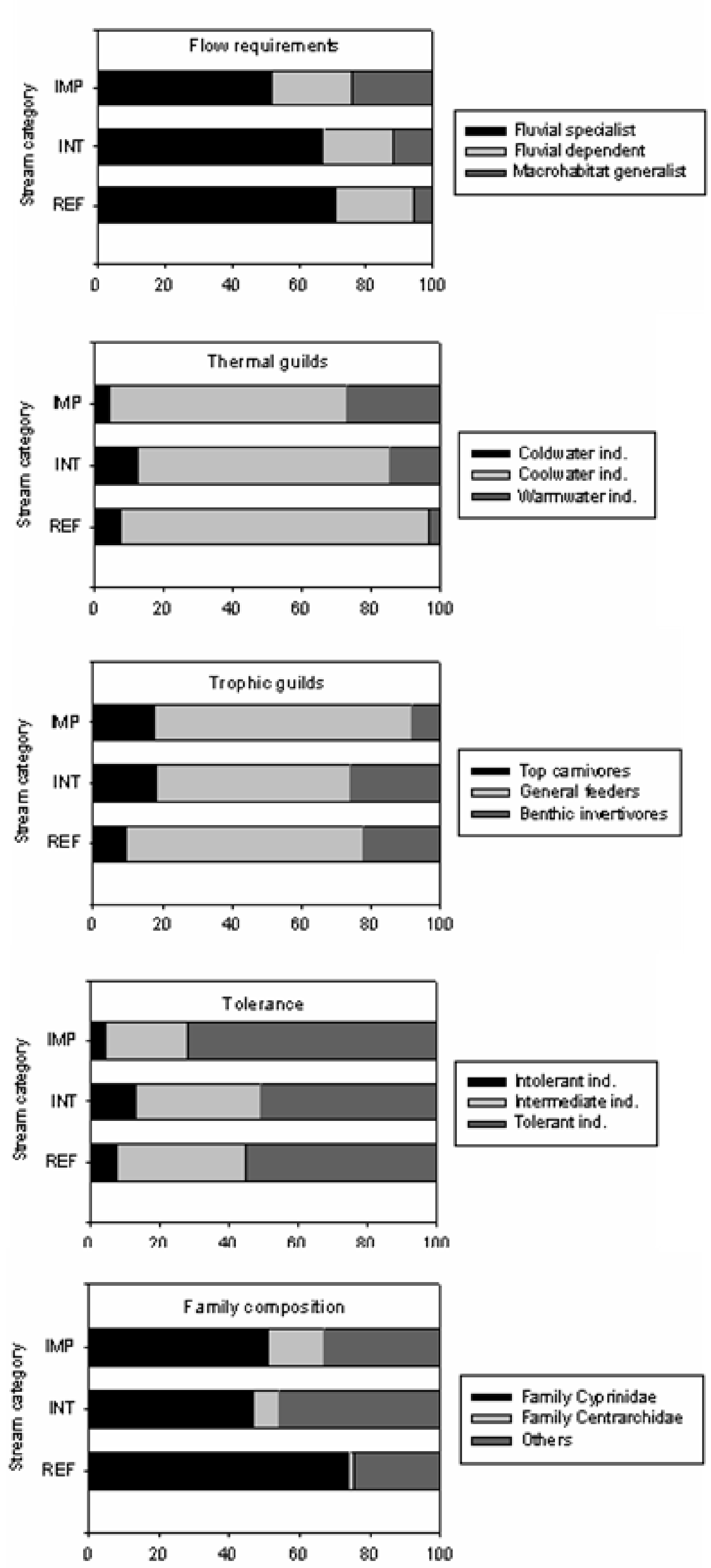

Figure 2. Proportional composition of ecological groupings among study site stream categories of impoundment, intake, and reference.

Table 4. Sums of Akaike Information Criterion (AIC_c) weights across competing generalized linear regression models for five explanatory variables. Larger values within table rows indicates relatively higher importance in explaining the data.

	Dam	Withdrawal index	Drainage area	Forest	Stratified drift
Flow-related guilds					
% Fluvial Specialist			0.13	0.09	
% Fluvial dependent		0.34	0.31		
% Macrohabitat generalist		0.66	0.41		
Thermal guilds					
% Coldwater			0.61	0.22	0.24
% Coolwater					0.11
% Warmwater	0.14	0.08			
Trophic guilds					
% Benthic Invertivore		0.27			
% Non-Tolerant Gen Feeder		0.06	0.26		0.18
% Top Carnivore				0.36	0.36
Tolerance					
% Tolerant		0.10			0.14
% Intolerant			0.65	0.26	0.30
Indicater species/family					
% White sucker		0.10	0.14		
% Brook trout			0.45	0.08	0.13
% Cyprinidae					0.41
% Centrarchidae		0.76	0.32	0.11	0.20

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tion of effects suggested that increasing withdrawal rate generally resulted in a proportional decrease in fluvial dependent species, benthic invertivore species, and white sucker (Table 3). A proportional increase was seen in macrohabitat generalists and the Centrarchidae (Figure 3).

The sum of AIC_c weights (w_i) across competing models indicated that WI was more important than other watershed-scale variables (impoundment, drainage area, % forest, and % stratified drift) for explaining the proportional abundance of fluvial dependent, macrohabitat generalist, benthic invertivore, and family Centrarchidae individuals (Table 4). Other watershed-scale variables were included in the competing models for some of the metrics tested (Table 3). Drainage area, forest and stratified drift were important variables explaining proportional abundance of several functional groups, such as % coldwater individuals, % top carnivores individuals, % brook trout individuals, % intolerant individuals, and % family Cyprinidae individuals (Table 4).

Predicted effect sizes for WI levels increased with the magnitude of WI (Table 5). Fish assemblages are predicted to remain less-altered when withdrawal rates are small (i.e., WI less than 10), but the alteration accelerated with increasing WI. The largest reductions were predicted for benthic invertivores and fluvial dependent species. The percent of benthic invertivores decreased from 27% to 24% as permitted with-

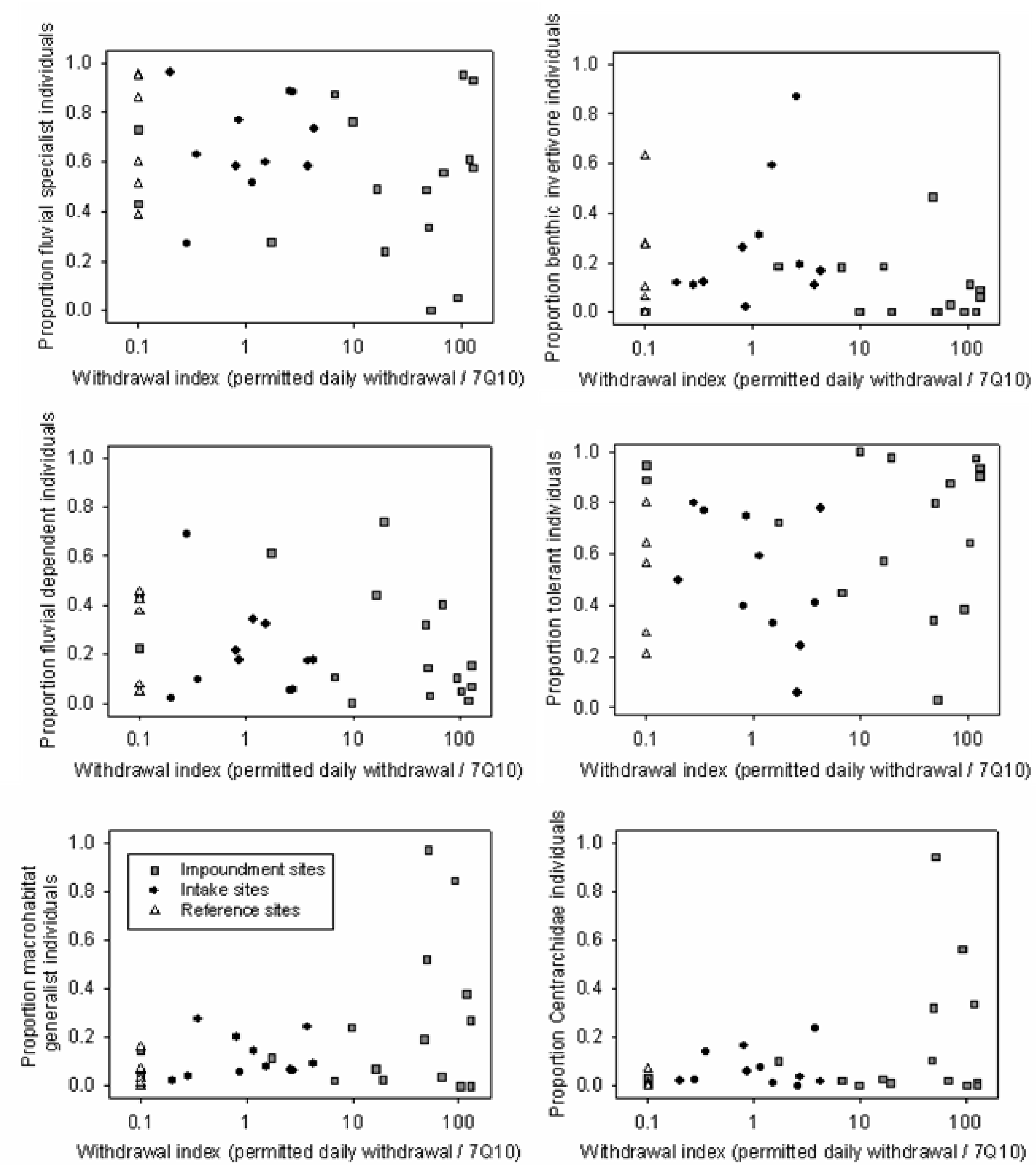

Figure 3. Example fish assemblage characteristic groupings plotted against Withdrawal Index (WI). A value of 0.1 was added to WI values to facilitate display in log transformed values.

Table 5. Effect size of Withdrawal Index (WI) on fish assemblage metrics which included WI in competing models. Effect size is based on a generalized linear regression model which included WI, intercept, and a scale term in the model. The WI value of 0 indicates no withdrawal.

	Withdrawal Index (WI)			
	0	10	50	100
Flow-related guilds				
% Fluvial dependent	28	25	18	12
% Macrohabitat generalist	10	12	25	64
Thermal guilds				
% Warmwater	16	17	25	40
Trophic guilds				
% Benthic Invertivore	27	24	17	11
% Non-Tolerant Gen Feeder	12	13	17	26
Tolerance				
% Tolerant	58	59	65	73
Indicater species/family				
% White sucker	17	16	14	12
% Centrarchidae	6	8	17	51

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drawal increased from 0 (no withdrawal) to 10 (10 × 7Q10). Benthic invertivores further decreased to 17% and 11%, respectively, as WI increased to 50 and 100. A corresponding proportional increase was recorded in other metrics. The largest proportional increase was predicted for macrohabitat generalists and family Centarchidae. The proportional abundance of macrohabitat generalists increased from 10% to 12%, 25%, and finally to 64% as permitted withdrawal increased from 0 to 10, 50, and 100 respectively.

Discussion

This study showed that the impacts of withdrawals and impoundments can be quantified even in a species-poor, highly-altered landscape such as the northeastern USA. Freeman and Marcinek (2006) demonstrated that fluvial specialist species richness was negatively impacted by withdrawal rate and impoundments in Piedmont streams in Georgia. In their study, fish collection from a single study site documented 20 fluvial specialist species while another site had 39 macrohabitat generalist species. This is in stark contrast with Connecticut streams, where the most speciose site recorded 13 total species with a mean of eight. Rather than species numbers, we investigated the proportional abundance of selected ecological groupings commonly investigated in biomonitoring. This approach allowed us to investigate similar questions, although it must be noted that changes in percent composition of ecological species groupings sensitive to flow alteration is different than the complete absence (or presence) of said species at withdrawal sites.

While fluvial specialist species richness decreased and macrohabitat generalist richness remained unaltered in response to increasing withdrawal rate in Georgia streams, the current study indicated an opposite pattern of constant fluvial specialist proportional abundance and increased macrohabitat generalist proportional abundance. The difference may have resulted from the mentioned difference in response variables (richness vs. proportion) or species classification schemes (i.e., the distinction between fluvial specialist and fluvial dependent). In Georgia, a diverse array of minnows, suckers, darters, and catfishes were observed to be sensitive to an altered flow regime (Freeman and Marcinek 2006). While Connecticut study sites had fewer flow-sensitive species, our results are interpreted as consistent with ecological theory that alteration of the natural flow regime will impact stream biota (Poff et al. 1997; Bunn and Arthington 2002).

Flow regime alteration has been documented to affect individual fish species and fish size structure (Wenger et al. 2008; Walters and Post 2008). Wenger et al. (2008) caution that all species in the same ecological grouping do not necessarily respond to environmental disturbances in a similar manner and fish grouping based on ecological traits might mask individual species' response to environmental disturbances. They observed that while some fluvial specialists were sensitive to flow alteration, other species showed no declines. We addressed this concern by including two indicator species for the region, brook trout and white sucker. Our results showed that an individual species approach might be worth pursuing as the percent of white sucker decreased with increasing WI, while the percent of tolerant fishes (a group to which white sucker belongs) increased.

Single-species approaches, however, may not be able to illuminate the generalized effects of environmental perturbations in species-poor regions such as the Northeast United States, as choices of indicator species are few, and their distributions are often not ubiquitous. We believe that ecological grouping is an appropriate first approach to understanding the impacts of water withdrawal, but suggest that more detailed research should be undertaken to better understand underlying relationships (*sensu* Wenger et al. 2008).

Depending upon the magnitude of withdrawal and the seasonal mode of operation, water withdrawal substantially reduce flow volume, and can have indirect consequences through physical and chemical habitat variables such as water temperature and oxygen content (Richter et al. 1996). The impact of water withdrawal may be especially important during low-flow periods (Fausch and Bramblett 1991; Poff and Ward 1989), and the use of 7Q10 in the current study is based on such an assumption. Low stream discharge tends to reduce riffle area (i.e., fluvial habitat) much more than pool area (Hakala and Hartman 2004), thereby potentially reducing habitat quality for species dependent on shallow fast habitats more so than for macrohabitat generalists. The proportional decrease of benthic invertivores (principally longnose dace and tessellated darter *Etheostoma olmstedi*) observed in our study fit this scenario, since they are associated with riffle habitat. Stream fish persistence under low-flow conditions is a complex problem which requires an understanding of stream connectivity at a broader spatial scale, including the presence of refugia and fish dispersal capability (Labbe and Fausch 2000; Magoulick and Kobza 2003).

It was surprising that presence of a dam was included in competing regression models for only one fish assemblage metric tested (% warmwater individuals). The scarcity of this variable in completing models may simply indicate that other variables were relatively more important in explaining patterns in the data. Withdrawal index, being a continuous measure of stressor magnitude, perhaps captured and explained more variation than the variable dam, which was coded simply presence or absence. In theory, water withdrawals with impoundments can exert additional impacts on flow regimes because they are capable of storing water and thus dampen temporal variability (Poff et al. 1997). The presence of impoundments, by itself, may create an inhospitable environment and act as a barrier for fluvial species (Skalski et al. 2008), and they may also function as source habitat of macrohabitat generalists. The current study did not directly examine if proportional increase in macrohabitat generalists at impoundments sites were caused by reduction of fluvial habitat downstream or the emigration of habitat generalists escaping from impoundments. We assume both mechanisms are plausible, but a better understanding of such ecological mechanisms requires more directed field work.

This study provided circumstantial evidence that groundwater was important for structuring fish assemblages. Groundwater potential was indexed by the percent of upstream drainage area underlain by coarse-grained stratified drift, and this watershed-scale variable was retained in many competing regression models (Tables 3 and 4). As expected, stratified drift was particularly important for coldwater and intolerant guilds, and the indicator species brook trout. Groundwater potential estimated at the watershed scale was similarly useful for characterizing fish distributions in Michigan, and brook trout was associated with small streams with the greatest groundwater input (Zorn et al. 2002). Behavioral thermoregulation at a fine spatial scale has also been observed for salmonids (Snucins and Gunn 1995; Biro 1998). Therefore, our results together with previous research suggests that groundwater protection is important especially for coldwater fishes, and may become even more important under future climate change scenarios (Power et al. 1999; Chu et al. 2008).

These results should be useful for water allocation management in Connecticut. Predicted withdrawal effect sizes suggested that withdrawal rate may have a relatively small effect on fish assemblage composition when WI ranged between 0 and 10. This range encompassed water withdrawal magnitudes at the intake sites sampled (Table 5). However, when WI increased to 50, proportional changes in functional guilds were noticeable, and generalist and tolerant forms were estimated to dominate when WI reached 100 (near the higher end of withdrawal magnitudes observed for impoundment sites). This suggests that the fish assemblage responses to withdrawal magnitude were not classically linear, and there may exist threshold points beyond which fish assemblages start to lose resistance or resilience to environmental stress (Allan 2004). However, precisely identifying this threshold is complex; and similar to Freeman and Marcinek (2006) we were not able to completely separate the effects of impoundments and withdrawal rate because impoundment sites withdraw significantly more water than intake sites. Another difficulty is that the threshold potentially differs among stream types (Utz et al. 2009). Kanno and Vokoun (2008) identified three lotic fish assemblages structured by a stream size gradient within wadeable streams (i.e., not including large rivers) in the region. They described brook trout dominated headwaters, blacknose dace and creek chub *Semotilus atromaculatus* dominated assemblages downstream, and even further downstream stream segments harbored more diverse fish assemblages including species such as longnose dace, fallfish *Semotilus corporalis*, common shiner *Luxilus cornutus*, and white sucker. Future research is warranted to examine if fish assemblage response to withdrawal magnitude differs among stream types.

In conclusion, this is the first study in the northeastern United States of which we are aware to describe the effect of water withdrawals and impoundments on fish assemblage compositions in streams. Our results suggest that water withdrawals have contributed to measurable alterations of fish assemblages and should therefore be considered in stream flow regulation and aquatic conservation. To measure these changes in assemblages we used ecological groupings commonly used to evaluate the biological condition of streams against a standard of naturalness or least-altered reference condition (*sensu* Stoddard et al. 2006). We note however, that this study used reference sites with comparable levels of watershed forest cover, surficial geology, stream widths, water depths and drainage area as the withdrawal sites. As such the fish assemblage changes described here represent differences from the prevalent biological condition, and are perhaps smaller than those that could be described if compared to a least-altered condition.

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Appendices

Appendix Table 1. Ecological characteristics of 25 fish species collected in this study. Stocked salmonids are not included in the list. Information is based on regional references (Whitworth 1996; Halliwell et al. 1999; Armstrong et al. 2001). Species are listed alphabetically by common name. Abbreviations are: C = coldwater, C-W = coolwater, W = warmwater; GF = general feeder, TC = top carnivore, BI = benthic invertivore, WC = water column insectivore; I = intolerant, M = intermediate, T = tolerant; FS = fluvial specialist, FD = fluvial dependent, MG = macrohabitat generalist; A = non-native, N = native.

Species name	Abbrev	Temp	Trophic	Tolerance	Flow	Origin
American eel *Anguilla rostrata*	AE	W	TC	T	FD	N
Banded sunfish *Enneacanthus obesus*	BS	W	WC	I	FD	N
Blacknose dace *Rhinichthys atratulus*	BL	C-W	GF	T	FS	N
Bluegill *Lepomis macrochirus*	BG	W	GF	T	MG	A
Brook trout *Salvelinus fontinalis*	BK	C	TC	I	FS	N
Brown bullhead *Ameiurus nebulosus*	BB	W	GF	T	MG	N
Chain pickerel *Esox niger*	CP	W	TC	M	MG	N
Common shiner *Luxilus cornutus*	CS	C-W	GF	M	FD	N
Creek chub *Semotilus atromaculatus*	CR	C-W	GF	T	MG	N
Creek chubsucker *Erimyzon oblongus*	CH	W	GF	I	FS	N
Cutlips minnow *Exoglossum maxillingua*	CM	W	BI	I	FS	N
Fallfish *Semotilus corporalis*	FF	C-W	GF	M	FS	N
Fourspine stickleback *Apeltes quadracus*	FS	C-W	WC	M	FD	N
Golden shiner *Notemigonus crysoleucas*	GS	W	GF	T	MG	N
Green sunfish *Lepomis cyanellus*	GR	W	GF	T	FD	A
Largemouth bass *Micropterus salmoides*	LM	W	TC	M	MG	A
Longnose dace *Rhinichthys cataractae*	LD	C-W	BI	M	FS	N
Northern pike *Esox lucius*	NP	C-W	TC	I	MG	A
Pumpkinseed *Lepomis gibbosus*	PS	W	GF	M	MG	N
Redbreast sunfish *Lepomis auritus*	RS	W	GF	M	MG	N
Redfin pickerel *Esox americanus americanus*	RF	W-B	TC	M	MG	N
Slimy sculpin *Cottus cognatus*	SC	C	BI	I	FS	N
Tessellated darter *Etheostoma olmstedi*	TD	C-W	BI	M	FS	N
White sucker *Catostomus commersonii*	WS	C-W	GF	T	FD	N
Yellow perch *Perca flavescens*	YP	C-W	TC	M	MG	N

Appendix Table 2. Number of individuals of 25 fish species collected at all sampling sites. See Appendix table 1 for fish species abbreviations (columns).

		AE	BS	BL	BG	BK	BB	CP	CS	CR	CH	CM	FF	FS
INT 01	Whitford Brook	88	1			1	2	17			1			
INT 02	Fenton River			201	4	23	1	1	23				14	
INT 03	Freshwater Brook	2		31	104		3						253	
INT 04	Gulf Stream	4		117		89								
INT 05	Hungary Brook			28	10	5			14				12	
INT 06	West River	36		30	2		1	1	11				5	
INT 07	Moore Brook			92	27	20				30				
INT 08	Nonnewaug River	1		319	1	20	1		5	43				
INT 09	Aspetuck River	11		60					6	20		23		
INT 10	Stratton Brook			24	6	83		1					5	
INT 11	Mill River			1		1								
IMP 01	Latimer Brook	4		2	9			8						
IMP 02	Stony Brook	1												
IMP 03	Roaring Brook			57	2		2	2					5	
IMP 04	Nepaug River			61		7	1		2	1				
IMP 05	Menunketesuck River	49		20										
IMP 06	Muddy River	29		159					90					
IMP 07	Farmill River	7		42	19		3	3		16				
IMP 08	Wangum Lake River			274						86				
IMP 09	Beaver Brook	2		78	2	4								
IMP 10	Wigwam Brook			115	62	2	4			4				
IMP 11	Rippowam River	5		3				9						
IMP 12	Converse Pond Brook			193						101				
IMP 13	Broad Brook (E Windsor)	167		14	27								19	
IMP 14	Farm River	1		138										
IMP 15	Broad Brook (Cheshire)	3		50	2	53								1
IMP 16	Beacon Hill Brook			37		15								
REF 01	Branch Brook			294		57			7				11	
REF 02	Jeremy River	13		47	1		2	3	73				42	
REF 03	Whiting River			87					102	49			1	
REF 04	Rock Brook			49		5	1		1	3				
REF 05	Green Fall River	3		47		42							3	
REF 06	Tankerhoosen River			34	4	11								
Total		426	1	2704	282	438	21	45	334	353	1	23	370	1

Appendix Table 2 expanded.

GS	GR	LM	LD	NP	PS	RS	RF	SC	TD	WS	YP	Total
		1	59		1				103			274
	3	7			5				7	30		319
3		8			29				65	101		599
					5			29		1		245
1		6	22		7		4		14	16		139
2	1		24			2				102	1	218
		2			2			27		21		221
		10	71						27	99		597
		12	22		10				47	84		295
				1			2		31	9		162
			59				9		61	7		138
		4			9						3	39
					32						1	34
6		1								61	8	144
		3	4							50		129
					1					13	1	84
		13	65		2		2		30	109	19	518
2					11	10				11	1	125
												360
1		2								23		112
		3								2		192
			35		6	8			28	38	3	135
1	1					4			34	60		394
		2	62		8		4		4	53		360
			10							10		159
		1	25						2	12		149
			7							3		62
	2									8		379
		1	61		1				16	18		278
			12						19	38		308
			25							6	2	92
			172				3		11	10		291
			6		3					42		100
16	7	76	741	1	132	24	24	56	499	1037	39	7651

Appendix Table 3. Watershed- and local-scale habitat characteristics for study stream sites.

SiteID	Name	Area (km^2)	% Forest	% Stratified Drift	Permitted withdrawal (mgd)	7Q10 (mgd)	Withdrawal Index (WI)	Mean width (m)
INT 01	Whitford Brook	32.37	78	16	1.46	0.96	1.53	6.7
INT 02	Fenton River	64.28	83	8	0.84	0.98	0.86	10.6
INT 03	Freshwater Brook	13.16	35	13	0.86	0.23	3.80	2.9
INT 04	Gulf Stream	7.77	85	4	0.01	0.07	0.20	2.4
INT 05	Hungary Brook	29.81	68	39	1.30	1.61	0.81	6.7
INT 06	West River	30.74	75	10	0.16	0.58	0.28	5.6
INT 07	Moore Brook	27.09	80	27	0.43	1.25	0.35	5.9
INT 08	Nonnewaug River	28.59	50	5	1.30	0.30	4.30	6.0
INT 09	Aspetuck River	36.44	82	8	0.65	0.56	1.16	6.4
INT 10	Stratton Brook	12.38	78	58	3.28	1.18	2.77	5.5
INT 11	Mill River	49.05	59	28	6.40	2.46	2.60	16.0
IMP 01	Latimer Brook	17.33	81	13	44.00	0.47	93.02	3.5
IMP 02	Stony Brook	6.22	83	2	2.12	0.04	52.40	2.0
IMP 03	Roaring Brook	14.45	90	1	0.00	0.05	0.00	7.7
IMP 04	Nepaug River	82.80	79	19	198.55	2.88	68.94	3.5
IMP 05	Menunketesuck River	26.39	83	7	8.34	0.43	19.50	4.6
IMP 06	Muddy River	25.54	46	16	12.20	0.74	16.53	3.9
IMP 07	Farmill River	17.40	52	14	24.00	0.48	50.05	3.6
IMP 08	Wangum Lake River	8.57	85	0	0.20	0.02	9.83	3.7
IMP 09	Beaver Brook	4.09	88	0	0.00	0.01	0.00	4.5
IMP 10	Wigwam Brook	47.22	63	1	25.40	0.21	120.11	5.4
IMP 11	Rippowam River	34.58	78	10	32.00	0.67	48.01	5.5
IMP 12	Converse Pond Brook	9.66	69	1	46.00	0.04	130.00*	2.8
IMP 13	Broad Brook (E Windsor)	40.82	45	50	6.15	3.57	1.72	8.2
IMP 14	Farm River	8.21	59	9	84.00	0.15	130.00*	3.7
IMP 15	Broad Brook (Cheshire)	12.33	56	29	5.00	0.74	6.75	3.9
IMP 16	Beacon Hill Brook	11.97	79	3	10.00	0.10	103.92	6.1
REF 01	Branch Brook	11.97	93	2	0.00	0.07	0.00	5.5
REF 02	Jeremy River	18.21	79	2	0.00	0.13	0.00	3.9
REF 03	Whiting River	36.67	83	2	0.00	0.20	0.00	5.1
REF 04	Rock Brook	22.33	75	4	0.00	0.21	0.00	7.8
REF 05	Green Fall River	17.66	92	7	0.00	0.26	0.00	4.0
REF 06	Tankerhoosen River	23.83	70	34	0.00	1.32	0.00	5.0

Appendix Table 3 expanded.

Mean depth (cm)	Discharge (m^3/s)	Water temp (°C)	Survey date
14.2	0.043	22.3	7/9/2007
16.2	0.106	17.0	7/25/2007
25.7	0.003	20.3	8/17/2007
16.1	0.048	15.7	6/22/2007
21.0	0.174	22.1	8/9/2007
23.9	0.062	21.1	6/28/2007
20.6	0.187	17.8	7/19/2007
26.5	0.165	17.9	7/24/2007
13.9	0.005	19.0	8/6/2007
28.7	0.210	17.8	7/25/2008
12.9	N/A	20.9	7/23/2008
12.3	0.007	22.0	8/7/2007
9.5	0.001	18.1	7/12/2007
26.4	N/A	21.2	6/15/2007
15.0	0.001	16.2	8/22/2007
14.0	0.001	17.5	6/26/2007
14.9	0.014	16.1	8/15/2007
20.5	0.018	22.7	7/6/2007
14.6	0.024	16.6	7/13/2007
18.9	0.063	22.4	7/31/2007
19.7	0.052	24.0	8/2/2007
35.1	N/A	18.4	8/13/2007
16.0	0.006	18.0	6/25/2007
32.4	0.328	18.9	8/30/2008
17.1	0.013	22.7	7/18/2008
15.5	0.016	14.8	5/28/2008
29.1	0.214	11.3	5/22/2008
15.3	0.030	20.8	6/17/2007
13.0	0.031	21.7	7/16/2007
33.1	0.072	20.1	8/3/2007
25.4	0.063	17.7	6/29/2007
13.0	0.015	19.8	7/22/2008
22.1	0.113	19.6	7/14/2008