



Evaluating effects of water withdrawals and impoundments on fish assemblages in southern New England streams, USA

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Abstract Balancing aquatic conservation and water supply is becoming a major global issue for urban landscapes. The purpose of this study was to evaluate the ecological effects of stream-flow alterations via water withdrawals and impoundments on fish assemblage structure. Electric fishing data were collected at 33 wadeable streams in Connecticut, located in the Southern New England region of USA. Fish sampling was conducted directly downstream of water withdrawals for municipal and agricultural water supply, and study sites differed in potential water withdrawal rates and the presence of impoundments. Regression analysis showed that water withdrawal rate was more important than other natural and anthropogenic factors (e.g. landcover and stream size) in explaining several fish assemblage metrics. Stream sites with high withdrawal rates were generally characterised by 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 in riffle habitat), and had a greater percent composition of macrohabitat generalists, particularly members of the family Centrarchidae. Some assemblage metrics responded linearly with increasing magnitude of water withdrawals, but others were non-linear. Results are consistent with ecological theory that alteration of the natural flow regime will impact stream biota.

KEYWORDS: fish assemblage, hydrologic alteration, Southern New England, stream, water supply, water withdrawal.

Introduction

The natural flow regime 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 depends on the appropriate quantity, quality, timing and temporal variability of water flow, to which aquatic species have adapted (Poff *et al.* 1997; Bunn & Arthington 2002). The natural flow regimes may differ among watersheds and regions because of stream size and geographic variation (e.g. climate and topography), and various forms of human activities can lead to stream flow alteration (Richter *et al.* 1996; Roy *et al.* 2005; Nelson & Palmer 2007).

Water withdrawals are a major anthropogenic activity that can directly impact stream flow. Reduced discharge impacts on fish size (Walters & Post 2008)

and occurrence of fluvial specialist fish species that require lotic habitats for most of their life history cycles (Armstrong *et al.* 2001; Freeman & Marcinek 2006). As a general pattern, stream discharge has been positively associated with species richness (Xenopoulos & Lodge 2006; Shea & Peterson 2007). Water reduction in the stream channel results in loss of habitat volume and can reduce connectivity (Labbe & Fausch 2000), and indirectly affects water quality and food resources (Richter *et al.* 1996; 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 & Marcinek 2006). Either form of water withdrawal can be catastrophic, however, if the magnitude is sufficient (Sophocleous 2007; Winter 2007).

The protection of aquatic ecosystems under an expanding human population is of global concern

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(Baron *et al.* 2002; Brasher 2003). Water quantity is becoming a major issue for urbanising landscapes such as those in the north-eastern USA, even though water has not traditionally been considered limited (Armstrong *et al.* 2001). Fish assemblages are structured by many environmental factors besides water withdrawal, and many of these factors are natural, such as stream channel gradient and elevation (Maret *et al.* 1997; Waite & Carpenter 2000). Surface geology influences both channel geomorphology and the ability of groundwater to enter streams (Walters *et al.* 2003; Winter 2007). Drainage area and associated stream size is known to describe natural upstream–downstream assemblage differences (Sheldon 1968; Rahel & Hubert 1991; Walters *et al.* 2003). Recent work on biological monitoring documented that general measures of anthropogenic disturbance (e.g. road density, number of point-source discharges, human population density) can be associated with fish assemblage structure and composition in this region (Bain & Meixler 2008; Kanno *et al.* 2009a). Land cover within a watershed (e.g. forested area, impervious surface area) can have strong effects on aquatic ecosystems (Wang *et al.* 2000; Wenger *et al.* 2008). Flow regimes in urbanised watersheds are typically characterised with increased surface runoff and flashiness (Roy *et al.* 2005; Nelson & Palmer 2007), while the effect of water withdrawal may manifest in other aspects of flow alteration such as reduced base-flow levels. Therefore, an evaluation of water withdrawal impact will necessarily include these other factors known to affect fish assemblages.

The purpose of this study was to evaluate the ecological effects of increasing water withdrawal rate and impoundment presence on fish assemblage structure in wadeable streams (watershed sizes of approximately 5–80 km² draining streams of 2–4th order) in Connecticut, located in the Southern New England region, USA. 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: (1) one group had dams and associated impoundments (hereafter, impoundment sites); and (2) 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 (hereafter, reference sites). The effect of water withdrawal and impoundments were compared with

other natural and anthropogenic factors known to affect fish assemblages.

Materials and methods

Study area

This study was conducted in Connecticut located in the southern New England region of north-eastern USA (Fig. 1). Freshwater fauna in the region is characterised by low species richness relative to the mid-Atlantic and south-eastern USA because of the lack of well-connected refugia that limited opportunities for freshwater fishes to recolonise after glaciation (Whitworth 1996). The region lies in an extensively modified landscape. In addition to current sprawling land development, extensive deforestation occurred shortly after European settlement (Foster 1992). Thus, pristine stream conditions are lacking in the landscape.

Fish sampling targeted 33 stream reaches across Connecticut (Fig. 1). Study streams included 16 impoundment sites, 11 intake sites and 6 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 from which water was abstracted for agricultural use. Most of the impoundments were simple spill-over type control structures. Reference sites were selected to be comparable in size and land cover characteristics and were not specifically intended to be considered least-altered or indicative of streams with the greatest biological integrity (Stoddard *et al.* 2006).

Field sampling

Field surveys were conducted during the summer base flow (May to August in 2007 and 2008). Each site was surveyed once in either 2007 or 2008 with pulsed DC electric fishing with a Smith-Root (Vancouver, Washington, DC, USA) model LR-24 back-pack unit or a Smith-Root tote-barge electric fishing unit controlled by a Coffelt VVP-15 control unit and powered by a 3600-W generator. Efforts were made to conduct sampling immediately downstream of water withdrawals (*sensu* Freeman & Marcinek 2006). This was not feasible for three sites where the sampled reaches were moved downstream but in all cases were within 2.5 km of permitted withdrawals. The sampling distance extended 50 times the mean wetted-channel width to

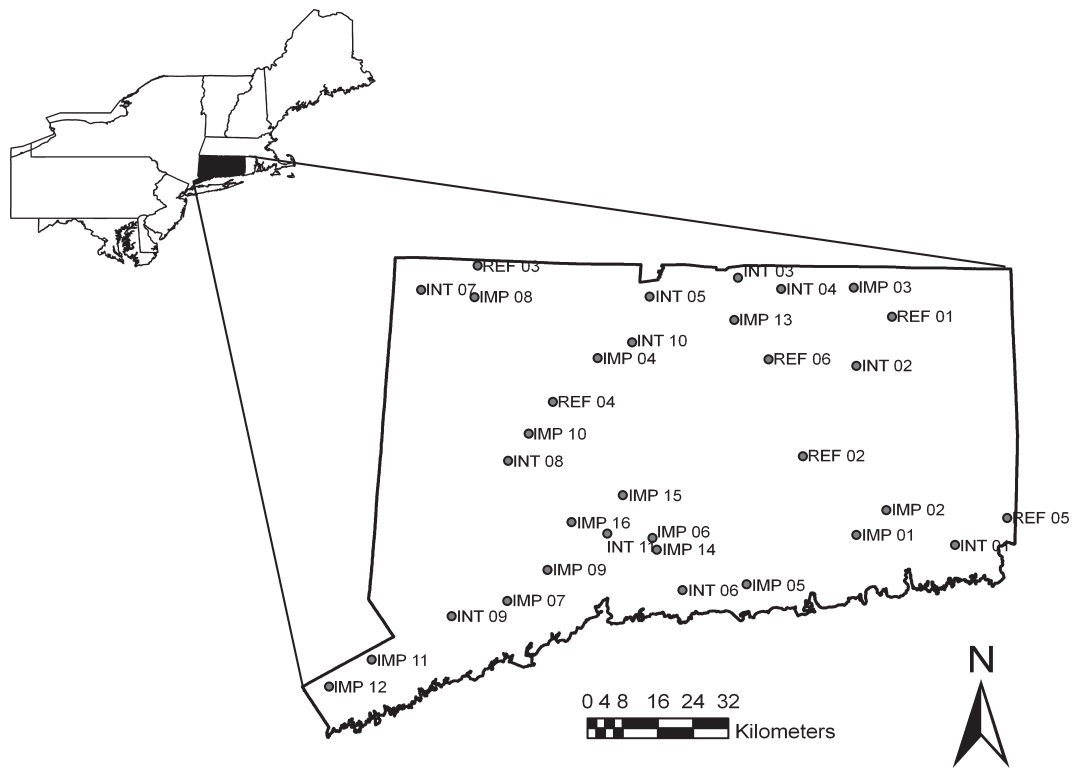


Figure 1. Map of the northeastern USA states with Connecticut highlighted in black and expanded in detail view showing spatial location of study sites surveyed in 2007 and 2008. IMP, impoundment sites; INT, intake sites; REF, reference sites.

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.* 2009b), so slightly longer reaches were sampled. At each site, one-pass electric fishing was conducted by three persons, starting from the downstream end and proceeding upstream by sampling all types of available microhabitat (e.g. run, riffle, pool). Fish caught were identified to species, measured for length (TL, mm) and released back to the stream.

Local-scale habitat data were collected on the same day fish were sampled to provide descriptive statistics at each study site. Stream width, depth and velocity were recorded at eight 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). Instantaneous velocities were measured with a Marsh-McBirney Flo-mate 2000 model current meter attached to a top-setting wading rod at six-tenths depth (0.6 D). Stream discharge was measured at a single transect

where velocities were relatively smooth (Bain & Stevenson 1999) using the velocity–area current meter method as described by Gordon *et al.* (2004) with verticals spaced 0.5 m or less depending on channel complexity. Water temperature was measured in the morning hours (9:00–11:00 AM).

Site characteristics

Local-scale habitat data (stream width, depth, velocity, discharge and water temperature) were compared between impoundment, intake and reference sites by Analysis of Variance (ANOVA) using PROC MIXED in program SAS (version 9.1, SAS Institute Inc., Cary, NC, USA). Data were checked for normality and homogeneity of variance, and transformed prior to analysis when these assumptions were not met. 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, CA, USA). 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, only percent of forest cover was used in subsequent analyses. The percent of upstream drainage area underlain by coarse-grained stratified drift was calculated with an existing surface materials GIS data layer available from the Connecticut Department of Environmental Protection. Stratified drift is known to be correlated with the potential for groundwater inputs to streams in the region (Cervione *et al.* 1982).

The magnitude of water withdrawals needed to be estimated because stream flow was not gauged in the study streams and reporting of actual withdrawal amounts is not mandatory in Connecticut. Water withdrawal magnitude was indexed by potential withdrawal rate 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 7-day, 10-year recurrence low flow (7Q10). Permitted withdrawal rates were used because actual withdrawal rates were not available from all study sites, but 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 at 15 sites for which actual withdrawal data were available ($r = 0.95$; $P < 0.0001$). The 7Q10 is a commonly used measure of base flow discharge, and it is often used by regulatory agencies in USA to set wastewater discharge criteria and minimum flow requirements (Freeman & Marcinek 2006). The 7Q10 was estimated using the regression method of Cervione *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 assemblages

The main aim was to quantify if fish assemblage metrics were affected by WI and the presence of an impoundment, relative to other natural and anthropo-

genic variables known to affect fish assemblages. Additional explanatory variables tested included upstream drainage area, percent of forested area and percent of coarse-grained stratified drift. Fifteen fish assemblage metrics were selected from five ecological categories: 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, *Catostomus commersonii* (Lacépède), individuals, % brook trout, *Salvelinus fontinalis* (Mitchill), individuals, % family Cyprinidae individuals and % family Centrarchidae individuals]. Ecological information on each fish species was based on regional references (Whitworth 1996; Halliwell *et al.* 1999; Armstrong *et al.* 2001) (Table 1). Fluvial specialist species require lotic habitats for most of their life history cycles and fluvial dependent species require lotic habitats to complete a portion of their life history, but otherwise are commonly found in lentic environment (Armstrong *et al.* 2001). Macro-habitat generalist species are commonly found in both lotic and lentic systems. The terms fluvial specialist species and macro-habitat generalist species are often used in the North American literature (Kinsolving & Bain 1993; Quinn & Kwak 2003; Freeman & Marcinek 2006) and they are synonymous to rheophilic species and eurytopic species, respectively, in European studies (Aarts & Nienhuis 2003; Poulet 2007). The term fluvial dependent species has been used in the study region (Armstrong *et al.* 2001) and is similar to partially rheophilic species in the European literature (Aarts & Nienhuis 2003).

Generalised linear models were constructed to examine the water withdrawal and impoundment impact on the 15 fish metrics. Generalised linear models are robust to assumptions of residual normality and variance homogeneity and are useful for analysing 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 (a null model with only an intercept term was also included). Withdrawal Index values depend on drainage area and percent stratified drift, but these three dependent variables were included in the regression analysis because neither pair was highly correlated (Pearson correlation coefficient: $|r| < 0.251$, $P > 0.160$). The 32

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 (Armstrong *et al.* 2001; Halliwell *et al.* 1999; Whitworth 1996). Species are listed alphabetically by common name

Species name	Temp	Trophic	Tolerance	Flow	Origin
American eel <i>Anguilla rostrata</i> (Lesueur)	W	TC	T	FD	N
Banded sunfish <i>Enneacanthus obesus</i> (Girard)	W	WC	I	FD	N
Blacknose dace <i>Rhinichthys atratulus</i> (Hermann)	C-W	GF	T	FS	N
Bluegill <i>Lepomis macrochirus</i> Rafinesque	W	GF	T	MG	A
Brook trout <i>Salvelinus fontinalis</i> (Mitchill)	C	TC	I	FS	N
Brown bullhead <i>Ameiurus nebulosus</i> (Lesueur)	W	GF	T	MG	N
Chain pickerel <i>Esox niger</i> Lesueur	W	TC	M	MG	N
Common shiner <i>Luxilus cornutus</i> (Mitchill)	C-W	GF	M	FD	N
Creek chub <i>Semotilus atromaculatus</i> (Mitchill)	C-W	GF	T	MG	N
Creek chubsucker <i>Erimyzon oblongus</i> (Mitchill)	W	GF	I	FS	N
Cutlips minnow <i>Exoglossum maxillingua</i> (Lesueur)	W	BI	I	FS	N
Fallfish <i>Semotilus corporalis</i> (Mitchill)	C-W	GF	M	FS	N
Fourspine stickleback <i>Apeltes quadracus</i> (Mitchill)	C-W	WC	M	FD	N
Golden shiner <i>Notemigonus crysoleucas</i> (Mitchill)	W	GF	T	MG	N
Green sunfish <i>Lepomis cyanellus</i> Rafinesque	W	GF	T	FD	A
Largemouth bass <i>Micropterus salmoides</i> Lacepède	W	TC	M	MG	A
Longnose dace <i>Rhinichthys cataractae</i> (Valenciennes)	C-W	BI	M	FS	N
Northern pike <i>Esox lucius</i> L.	C-W	TC	I	MG	A
Pumpkinseed <i>Lepomis gibbosus</i> (L.)	W	GF	M	MG	N
Redbreast sunfish <i>Lepomis auritus</i> (L.)	W	GF	M	MG	N
Redfin pickerel <i>Esox americanus americanus</i> Gmelin	W-B	TC	M	MG	N
Slimy sculpin <i>Cottus cognatus</i> Richardson	C	BI	I	FS	N
Tessellated darter <i>Etheostoma olmstedii</i> Storer	C-W	BI	M	FS	N
White sucker <i>Catostomus commersonii</i> (Lacepède)	C-W	GF	T	FD	N
Yellow perch <i>Perca flavescens</i> (Mitchill)	C-W	TC	M	MG	N

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.

multiple regression models were fitted for each of the 15 fish metrics, specifying a gamma distribution and logarithmic link. The objective here was to rank and select among competing regression models and estimate effect sizes and their precision (Anderson *et al.* 2000). Generalised linear models were fitted in SAS using PROC GENMOD.

Support for candidate models was assessed using an information-theoretic approach (Burnham & Anderson 2002) in which a restricted, small 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 $\Delta\text{AIC}_c < 2$ 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 & Anderson 2002).

Effect sizes were predicted for selected 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

Twenty-five species (excluding stocked salmonids) were recorded from field surveys in 2007 and 2008 (Table 1). *Rhinichthys atratulus*, *Catostomus commersonii*, *Rhinichthys cataractae*, *Anguilla rostrata* and *Salvelinus fontinalis* were the five most common species in terms of occurrence among the 33 study sites. The mean observed species richness was eight species (range: 2–13) across 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.

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

	Area (km ²)	% Forest	% Stratified drift	Withdrawal Index	Mean width (m)	Mean depth (cm)	Discharge (m ³ s ⁻¹)	Water temp (°C)
Impoundment sites (n = 16)	22.97 \pm 20.41	71 \pm 15	11 \pm 13	53.17 \pm 48.68	4.5 \pm 1.7	19.4 \pm 7.4	0.054 \pm 0.096	18.8 \pm 3.5
Intake sites (n = 11)	30.15 \pm 15.62	70 \pm 15	20 \pm 16	1.69 \pm 1.38	6.8 \pm 3.6	20.0 \pm 5.4	0.100 \pm 0.074	19.3 \pm 2.1
Reference sites (n = 6)	21.78 \pm 8.40	82 \pm 9	8 \pm 13	N/A	5.2 \pm 1.4	20.3 \pm 8.1	0.054 \pm 0.036	19.9 \pm 1.4

Watershed- and local-scale habitat variables were not statistically different among impoundment, intake and reference sites (Table 2). However, WI was significantly larger for impoundment 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. These sites were assigned a WI value of 130, which was slightly larger than the greatest observed WI value (120).

Water withdrawal and impoundment effect on fish assemblages

Withdrawal Index was included in the competing regression models for 8 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. The direction of effects suggested that increasing withdrawal rate generally resulted in a proportional decrease in fluvial dependent species, benthic invertivore species and white sucker (Fig. 2, Table 3). A proportional increase was seen in the other metrics (Fig. 2). The presence of impoundment was retained in a competing model for only one metric (% warmwater individuals).

The sum of AIC_c weights (w_i) across competing models indicated that WI was more important than other watershed-scale variables (impoundment,

Table 3. Model-averaged coefficients of generalised linear regression analyses for 15 fish assemblage metrics selected to examine the relative importance of five watershed-scale variables. Note that measurements units differ among the five variables and the direct comparison of values across the variables is not advised. Blank space indicates that the variable was not included in the competing models used in model averaging

	Intercept	Impoundment	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
Indicator species/family						
% White sucker	-1.927		-0.004	0.010		1.589
% Brook trout	-1.759			-0.026	1.617	-1.787
% Cyprinidae	-0.374					-1.787
% Centrarchidae	-2.499		0.020	-0.020	-1.337	1.616

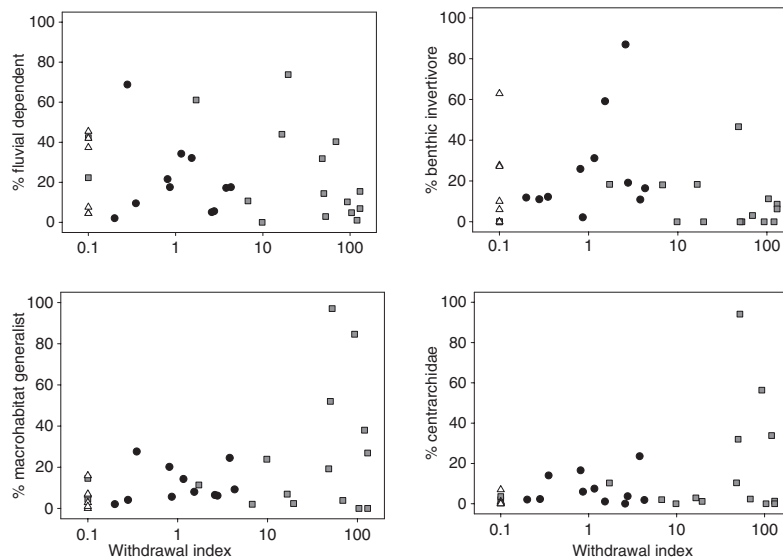


Figure 2. Example fish assemblage metrics and their relationship with Withdrawal Index (WI). Impoundment sites (□), intake sites (●) and reference sites (Δ). A value of 0.1 was added to WI values so that WI are shown in log-transformed values. As such, reference sites and other sites with discontinued withdrawal were assigned a WI value of 0.1.

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

	Impoundment	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
Indicator 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

drainage area, % forest and % stratified drift) in explaining the proportional abundance of fluvial dependent, benthic invertivore, macrohabitat generalist 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 responses to increasing WI levels somewhat differed among metrics (Fig. 3). The proportional

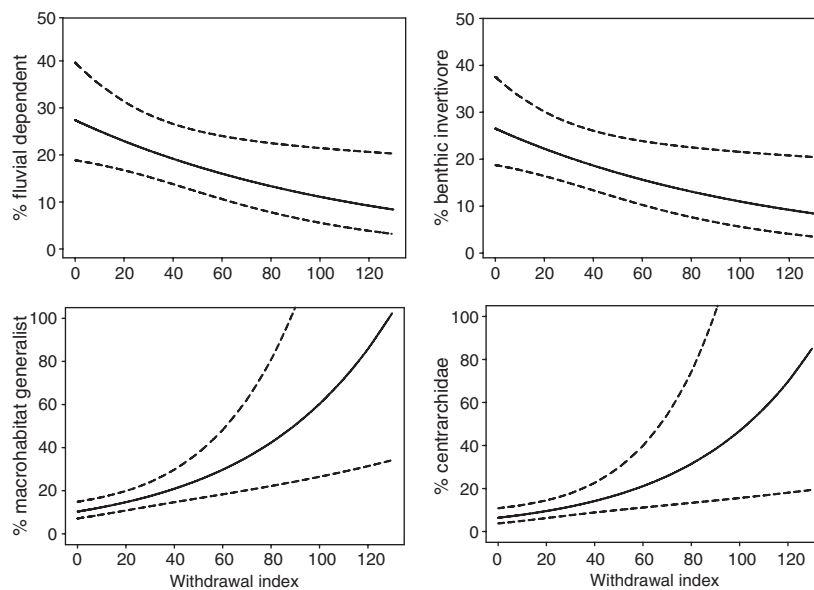


Figure 3. Predicted effect size of Withdrawal Index (WI) on example fish assemblage metrics. Predicted mean values (solid lines) are shown with 95% confidence intervals (dashed lines).

abundance of fluvial dependent and benthic invertivore individuals decreased rather linearly with increasing withdrawal rates. For example, the percent of benthic invertivores decreased from 27 to 24% as permitted withdrawal increased from 0 (no withdrawal) to 10 ($10 \times 7Q10$), and it further decreased to 17% and 11%, respectively, as WI increased to 50 and 100. However, predicted responses of macrohabitat generalists and family Centrarchidae were not linear, and the responses of these metrics were predicted to accelerate with increasing WI. The proportional abundance of macrohabitat generalists increased from 10 to 12%, 25% and finally to 60% as permitted withdrawal increased from 0 to 10, 50 and 100 respectively. All metrics had large 95% confidence intervals, indicating much variation in the data set and uncertainty associated with model predictions (Fig. 3).

Discussion

Water withdrawal can lead to reduced stream flow (Weiskel *et al.* 2007; Winter 2007), and the presence of impoundments can further impact temporal variability of stream flow through their water storage capacities (Poff *et al.* 1997). This study showed that the ecological impacts of water withdrawals can be quantified even in a species-poor, highly altered landscape such as northeast USA. In Piedmont streams in Georgia, USA, fluvial specialist species richness was negatively impacted by withdrawal rate and impoundments

(Freeman & Marcinek 2006). In that 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, the current study investigated the proportional abundance of selected ecological groupings commonly investigated in bio-monitoring. This approach allowed similar questions to be investigated, although it must be noted that changes in percent composition of ecological species groupings sensitive to flow alteration is different from the complete absence (or presence) of said species at withdrawal sites.

As another contrast to Georgia streams, the current study found constant fluvial specialist proportional abundance and increased macrohabitat generalist proportional abundance in response to increasing withdrawal rate. It may have resulted from the mentioned difference in response variables (richness vs proportion) or inherent differences in resilience to flow regime alteration among stream systems. In Georgia, a diverse array of minnows, suckers, darters and catfishes were sensitive to an altered flow regime (Freeman & Marcinek 2006). While Connecticut study sites had fewer flow-sensitive species, results are interpreted as consistent with ecological theory that alteration of the natural flow regime will impact stream biota (Poff *et al.* 1997; Bunn & Arthington 2002).

The impacts of water withdrawal may be especially important during low-flow periods (Poff & Ward 1989; Fausch & Bramblett 1991) 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 (Armstrong *et al.* 2001; Hakala & Hartman 2004), thereby potentially reducing habitat quality for species relying on shallow habitats with faster flows. The proportional decrease of benthic invertivores (principally *R. cataractae* and *E. olmstedii*) observed in this study fits this scenario, because they are associated with riffle habitat. Stream fish persistence under low-flow conditions is a complex problem that requires an understanding of stream connectivity at a broader spatial scale, including the presence of refugia and fish dispersal capability (Labbe & Fausch 2000; Magoulick & Kobza 2003).

Despite the general patterns of fish assemblage alterations in response to water withdrawals, predicting effect sizes of withdrawals precisely was confounded by large confidence intervals in regression analyses and model selection uncertainties, as similarly reported by Freeman and Marcinek (2006). This result should come as no surprise because stream fish assemblages are affected by multiple environmental factors operating at different spatial and temporal scales (Frissell *et al.* 1986). Other watershed-scale variables were included in competing regression models for many metrics, highlighting the complexity of stream ecosystems. In addition, some metrics showed a linear response to increasing withdrawal rate, while others had a non-linear response, suggesting that there may be a threshold point beyond which fish assemblages start to lose resistance or resilience to environmental stress for the latter group (Allan 2004). However, identifying this threshold precisely is complex; and similar to Freeman and Marcinek (2006) it was not possible to separate the effects of impoundments and withdrawal rate completely because impoundment sites withdraw considerably more water than intake sites. Another difficulty is that such a threshold potentially differs among stream types (Utz *et al.* 2009). Three lotic fish assemblages that were structured by stream size and thermal gradients were identified by Kanno and Vokoun (2008) within wadeable streams (i.e. not including large rivers) in the region. *Salvelinus fontinalis* dominated headwaters, *R. atratulus* and *S. atromaculatus* dominated assemblages downstream, and even further downstream segments harboured more diverse fish assemblages including species such as *R. cataractae*, *S. corporalis*, *L. cornutus* and *C. commersonii*. Future research is warranted to examine if fish assemblage

response to withdrawal magnitude differs among stream types.

Flow regime alteration can also affect individual fish species, as opposed to ecological groups (Wenger *et al.* 2008). Wenger *et al.* (2008) cautioned that all species in the same ecological grouping do not necessarily respond to environmental disturbances in a similar manner and grouping fish based on ecological traits might mask individual species' response to environmental disturbances. This concern was addressed by including two indicator species for the region, *S. fontinalis* and *C. commersonii*. Results showed that the percent of *C. commersonii* decreased with increasing WI, while the overall percent of tolerant fishes (a group to which white sucker belongs) increased. This disagreement suggested that an individual species approach might be worth pursuing. Single-species approaches, however, may not be able to illuminate the general effects of environmental perturbations. Ecological grouping is an appropriate first approach to understand the impacts of water withdrawal, but more detailed research should be undertaken to understand underlying relationships better (*sensu* Wenger *et al.* 2008).

It was surprising that the presence of an impoundment was included in competing regression models for only one fish assemblage metric tested (% warmwater individuals). The scarcity of this variable in competing models may indicate that other variables were relatively more important in explaining patterns in the data. Withdrawal Index, a continuous measure of stressor magnitude, perhaps captured and explained more variation than the variable impoundment, which was coded as presence or absence. In theory, water withdrawals with impoundments can exert additional impacts on flow regimes because they are capable of storing water, which can dampen temporal variability (Poff *et al.* 1997). The presence of an impoundment can create an inhospitable environment and act as a barrier for fluvial species (Skalski *et al.* 2008), and it may also function as source habitat for warmwater individuals (Herbert & Gelwick 2003). This study did not directly examine if proportional increases in warmwater individuals at impoundment sites were correlated to thermal regime alterations downstream or the emigration of individuals escaping from impoundments. Both mechanisms are potentially plausible, but a better understanding of such ecological mechanisms requires more directed field work.

This study provided circumstantial evidence that groundwater is important for structuring fish assemblages. Groundwater potential was indexed by the percent of upstream drainage area underlain by

coarse-grained stratified drift. This watershed-scale variable was retained in many competing regression models (Tables 3 and 4). Stratified drift was important for coldwater and intolerant guilds, and the indicator species *S. fontinalis*. Groundwater potential estimated at the watershed scale was similarly useful for characterising fish distributions in Michigan, USA, and *S. fontinalis* was associated with small streams with the greatest groundwater input (Zorn *et al.* 2002). Groundwater is typically cooler than surfacewater during summer and fish may take advantage of thermal refugia created by groundwater input (Snucins & Gunn 1995; Biro 1998). Therefore, the results together with previous research suggest 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). Groundwater may confer some degree of buffering capability to water abstraction, but an important question is to identify a withdrawal threshold beyond which ecological impacts become clear.

This is the first study to describe the effect of water withdrawals and impoundments on fish assemblage composition in streams in the northeast USA. 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, the current study 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). It should be noted, however, that this study used reference sites with comparable levels of watershed forest cover, surface 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 with a least-altered condition.

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