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Temporally stable and distinct fish assemblages between stream and earthen stormwater drain reaches in an urban watershed

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Abstract Streams and rivers have essential roles in landscape connectivity; however, urban watersheds are frequently modified to drain stormwater from urban areas. To determine whether an earthen stormwater drain in an urban landscape provides fish habitat temporally, we compared the fish assemblage among three reaches of a contiguous urbanized watershed in Kanata, Ontario, Canada. Watts Creek is connected to an earthen municipal surface stormwater drain (herein Kizell Drain), before discharging into the Ottawa River. We delineated transects in three reaches of the system, in Watts, in the Drain, and below their confluence (Main) and assessed the fish community using single-pass electrofishing repeated across eight months covering all seasons. Fish community composition was compared among reaches using non-metric multidimensional scaling (NMDS) and permutated multivariate analyses of variance (perMANOVA). Sign association tests identified indicator species driving assemblage patterns among reaches. Redundancy analysis (RDA) was used to assess the influence of physical characteristics of the transects on fish assemblage structure. Finally, fish assemblage measures were separated by month and temporal comparisons of fish assemblage were performed with NMDS and perMANOVA. Over the year, fish assemblages were distinct among the three reaches, and appear to be significantly influenced by temperature, undercut banks, and riparian vegetation type. Biotic homogenization in the Drain can be attributed to degraded physical features associated with channel modification in stormwater drains. Despite management and jurisdictional differences between streams and stormwater drains, evidence that earthen stormwater drains can maintain fish assemblages temporally demonstrates their biological potential and need to be considered as interconnected fish habitat elements within the overall watershed.

Keywords Stormwater management · Earthen surface drain · Urbanization · Non-metric multidimensional scaling · Redundancy analysis · Backpack electrofishing

Introduction

Rivers and streams are the hydrological highways that connect landscape elements (including riparian areas) and often serve as corridors within landscapes (Freeman et al. 2007). However, rivers and streams are frequently modified to suit human needs, and waterways may be altered to facilitate navigation, land use changes, or drainage (Brookes et al. 1983; Brooker 1985; Walsh et al. 2012). Physical modifications to rivers and streams can include channelization, dredging, or straightening. Consequences of these modifications can include reduction in riparian vegetation for bank stabilization and shading as well as destruction of natural riffle-pool-run sequences that oxygenate water and grade substrates (Daniels 1960; Hansen 1971; Keller 1978; Wang et al. 1997; Dudgeon et al. 2006; Urban et al. 2006).

Second to agriculture, urbanization is considered the most significant driver of stream alteration (Paul and Meyer 2001). With increasing demands on urban streams it is important to understand how the conversion of streams into stormwater management systems that are intended to convey water

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through urban areas may influence fish communities and fish habitat within urban watersheds. There are various types of stormwater drains in urban landscapes (e.g. manufactured or earthen) including surface (flows over the ground and is open to the surrounding environment) or subsurface (flows below ground and is closed to the surrounding environment; Djokic and Maidment 1991) drains. Subsurface drains tend to be pipes made of metal or concrete, whereas many surface drains can be more natural or earthen (i.e. ditches or swales) as a result of conversion from natural headwater streams. These altered drains remain part of the watershed and have the potential to retain biota and function as semi-natural watercourses (Kaushal and Belt 2012). Earthen stormwater drains have a direct impact on downstream watersheds by conveying runoff from impervious surfaces, resulting in altered flow regimes and increased turbidity and conductivity (Walsh et al. 2005). Physical, chemical, and biological habitat characteristics in streams are particularly sensitive to flow regimes, which can be altered in stormwater drains in urban areas. Natural variations in flow regimes are important for shaping fish community structure by influencing food conveyance, habitat availability, and stream hydrology (Poff and Ward 1989; Cunjak 1996; Marchetti and Moyle 2001; Konrad and Booth 2005).

Despite the considerable ecological change associated with human development and alteration, various types of drains (e.g., storm drains, municipal drains) still provide habitat and can support fish (Bayless and Smith 1967; Wichert and Rapport 1998; Stammler et al. 2008). Yet, there is a growing body of literature that demonstrates the effect of urbanization on stream fish assemblages including: increased inputs of sediment, changes in baseline flows, and increased discharge during storm events, all of which can reduce the amount and quality of habitat available to fish (Wichert 1994; Wang and Lyons 2003; Brown et al. 2005). Studies that compare fish communities both within and proximate to stormwater drains are therefore needed in order to understand the impacts of stream alterations on fish populations. Specifically, our objective is to determine whether municipal earthen stormwater drains provide suitable habitat for fish and whether they maintain distinct fish assemblages from interconnected reaches. Addressing these questions would demonstrate the ecological impacts of stormwater drains on stream habitats. If stormwater drains indeed provide fish habitat, then their role in the ecosystem may be more important than once thought, potentially necessitating management paradigm shifts. To assess the role of an earthen stormwater drain (which had been historically converted from a natural stream) in supporting the fish community of an urban watershed, we sampled fish from three reaches of an urban watershed, each with a different extent of urbanization. Multivariate analyses were used to test for differences in the fish assemblage among the reaches of the system, along physical environmental gradients, and through time. Temporal evidence of a fish assemblage being maintained within an earthen stormwater drain would demonstrate the biological potential for drains to contain habitat that supports fish throughout their life history.

Methods

Study area

We studied the fish assemblage native to the Watts Creek watershed (~2500 ha) in Kanata, Ontario, Canada (45°20′42″N, 75°52′19″W; Fig. 1). The land use within the Watts Creek watershed is approximately 47% agricultural, 35% developed, and 18% undeveloped (Stantec Consulting Ltd 2011). Watts Creek is a groundwater stream that originates in and flows through a residential community in Kanata before entering the Ottawa Greenbelt and eventually draining into the Ottawa River at Shirley’s Bay. The Ottawa Greenbelt contains both rural and agricultural lands that are protected and managed by the National Capital Commission (NCC). Our study area encompassed 4.6 km of Watts Creek (starting about 4.1 km upstream from the outflow into the Ottawa River) and a 1.5 km stretch of the Kizell Municipal Drain starting at its confluence with Watts Creek (Fig. 1). Kizell Drain is an earthen stormwater drain designated as a municipal drainage/watercourse under the Drainage Act in the Canadian Province of Ontario (R.S.O. 1990, c. D.17) and is managed by the City of Ottawa. It originates at Beaver Pond (a stormwater management pond near Walden Dr., Kanata, ON) and flows through residential Kanata before entering the Ottawa Greenbelt flowing into Watts Creek (Fig. 1).

We sampled three reaches within the Watts Creek watershed (Fig. 1), which were categorized as follows: Watts Creek upstream from the confluence of Watts and the Drain (herein referred to as Watts), Kizell Drain, and Watts Creek downstream of the confluence (herein referred to as Main). Twelve 100 m transects were established within the study area. Watts was the longest (~2.9 km), followed by Main (~1.7 km) and then the Drain (~1.5 km), therefore five, four, and three transects were established, respectively. The transect locations within each reach were selected based on their spread throughout the reach (toward the end and beginning of each reach) and in order to represent various riparian habitat types (e.g., forest, scrubland and agricultural). It is important to note that while there are few urban influences directly adjacent to much of our study area (except K3 which was adjacent to a road; Fig.1), there are heavy urban influences on the Drain located upstream from our sampling area. Also there were no natural or artificial barriers to movement within the study area.

The Main reach ran through a field primarily with riparian grass, sedge, and shrub, was the widest and deepest reach, and

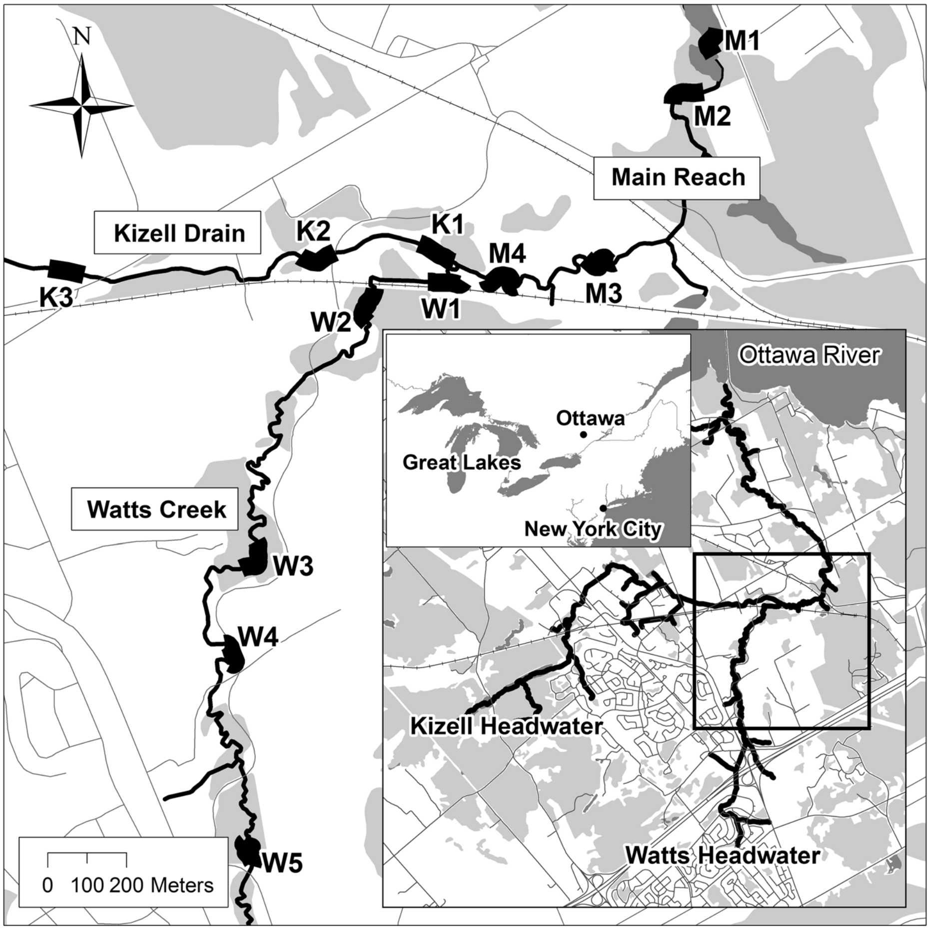


Fig. 1 Map of the Watts Creek study site with two insets. The first shows the location of Ottawa, in east-central North America and the second shows the position of the study site within Watts Creek showing the headwaters and connection to the Ottawa River. The direction of water flow is from west to east for Kizell Municipal Drain, and from south to north for Watts Creek. The sampling sites (*black polygons*) in each reach were numbered sequentially in an upstream direction. The confluence is located downstream of sites K1 and W1, and upstream from site M4. The thin grey lines are roads and pathways, while the thin grey hatched lines are train tracks. The light grey represents vegetated areas (primarily forest cover, but also dense riparian vegetation) with the dark grey representing water bodies

had considerable bank erosion (Table 1). Main also had the least amount of glide habitat while containing the most in-stream vegetation and structure (Table 1). Watts ran through more forested habitat with tree, grass, and shrub riparian habitat, had the lowest water temperature, coarser substrate, and the fastest water velocity (Table 1). Finally, the Drain traversed field habitat and was mostly bordered by grasses and shrubs, except for one transect that ran adjacent to an agricultural field. The Drain also lacked sinuosity, was mostly composed of fine substrate, and had the lowest velocity, highest mean water temperature and least in-stream structure (Table 1).

Community and habitat sampling

Fish communities were sampled monthly from each transect using single-pass backpack electrofishing (Model 12, Smith-Root, Vancouver, WA, USA) from 11 June 2012 to 17 December 2012 (except during September), and again from 22 April 2013 to 30 May 2013, for a total of 8 sampling events. An exception to our whole transect sampling was in August 2012 when the Ottawa area experienced drought conditions. Consequently, a large proportion of the Drain dried up

Table 1 Summary of the environmental variables for each reach. The values were averaged between the two sampling periods (September and May), except for sinuosity which was only measured once

Environmental Variable	Drain	Main	Watts
Channel Width (cm)	457	568	494
Channel Depth (cm)	55	87	94
Stream Width (cm)	315	380	309
Stream Depth (cm)	19	31	18
Temperature (°C)	19	15	17
Velocity (m/s)	0.12	0.14	0.16
Glide (%)	98	65	71
Pool (%)	1	10	10
Riffle (%)	1	25	19
Fine sediment (%)	85	61	57
Medium sediment (%)	11	23	28
Course sediment (%)	4	16	15
Overhang Cover (%)	19	18	40
In-stream vegetation (%)	23	32	4
In-stream structure (%)	21	45	27
Sinuosity	1.1	1.8	1.8

and we could only sample from small pools of water that remained. Flow returned to the Drain before the October 2012 sampling. Two dip nets were used to capture fish, which were held in large buckets and coolers with water from the area they were captured. Each fish was identified to species using Holm et al. (2009) and the total length (TL; mm) was measured before release. If fish could not be identified, vouchers were collected and preserved for later identification. Due to difficulties distinguishing Johnny (*Etheostoma nigrum*) and tessellated darter (*E. olmstedi*), they were grouped together as *Etheostoma* sp. In addition, two goldfish (*Carassius auratus*) that were captured during sampling were removed from the analysis because they are not native or established in the area (unlike common carp *Cyprinus carpio*, which are established) and could not be expected to provide relevant information about the habitat or species assemblage within the watershed. During the winter community assessment (11–17 December 2012), some transects had a layer of surface ice ranging from 1 to 13 cm thick. In order to access the water, the ice was manually broken and cleared. Regular electrofishing surveys were conducted 25 min after the ice was cleared to allow disturbed fish to return and the water clarity to improve. If a transect had no ice, we walked through it to mimic the disturbance caused by the ice breaking and again waited 25 min before sampling.

Qualities of the physical environment were measured to summarize the available habitats found within each reach. In order to capture variation in some of the variables (e.g. in-stream vegetation) the samples were measured twice: 24–28 September 2012 and 6–8 May 2013. Sinuosity was calculated by dividing the length of the reach (i.e. Watts, the Drain and Main) by the straight distance between the lower and upper limits of the reach in the study area. Channel (bankfull) and stream (wetted) widths were measured at 20 m intervals starting at the beginning of each transect for a total of six cross-sections. The cross-sections were further divided into three observation points (Stanfield 2005). The observation points were used to measure stream and channel depth, velocity and sample sediment composition. These measurements were averaged across the eighteen observation points covering the full transect. Sediment composition was determined by grabbing a sample from each observation point and estimating the proportion of different types (classified following the Wentworth scale for grain sizing; Wentworth 1922). The proportion of overhanging and in-stream cover was visually estimated within five 20 m subsections and averaged to determine the proportions for the entire transect. Overhanging cover included vegetation, under-cut banks, woody debris, and artificial structures (i.e., bridge). In-stream cover was separated into two categories: in-stream vegetation such as aquatic plants and macrophytes, and in-stream structure such as woody debris, coarse sediment (>64 mm), and detritus. Longitudinal measurements to the nearest centimeter were taken of habitat types (pool, riffle, or run). In addition four temperature loggers were deployed in two locations within the Drain, Main, and Watts (Fig. S1 in Online Resource 1) which recorded temperature every 255 min (5–6 times per day) from 7 May 2012 to 12 Oct 2012 and from 24 Oct 2012 to 16 Oct 2013.

Data analyses

Species diversity was measured by the Shannon-Weiner index for the three reaches using the diversity function in vegan (Oksanen et al. 2015). To explore the fish assemblage among the three reaches, our first approach was to sum fish abundance from all sampling events and run a nonmetric multidimensional scaling (NMDS) ordination comparing composite assemblages among the three reaches to determine whether the different sampling transects supported distinct fish assemblages. NMDS is an unconstrained ordination tool for delineating species assemblages that plots sampling locations in terms of species space measures; sites with similar characteristics are plotted more proximate to each other in the NMDS plot to evaluate relationships among sampling locations. The NMDS ordinations were based on Bray-Curtis matrices. NMDS was implemented using the metaMDS function in the vegan package and included rare species. To corroborate observations made during visual inspection of the NMDS plot, non-parametric permutated multivariate analyses of variance (perMANOVA) were performed to test the null hypothesis that the three reaches (Watts, Main, the Drain) had no difference in assemblage structure using 999 permutations with a pseudo-F test statistic. perMANOVA was carried out by the function *adonis* in the vegan package. In order to establish whether any species within the watershed were uniquely associated with particular reaches, indicator species analysis was carried out using a sign association test, implemented using the function *signassoc* in the indicspecies package (De Caceres and Legendre 2009). The sign association test produces p-values for each species (adjusted for multiple comparisons using Sidak's correction) to identify species whose abundance measures are driving a relationship among sites (i.e. indicator species). We used $\alpha = 0.10$ to assess indicator species for reaches; species that had significant relationships with a reach were considered to be indicator species driving relationships observed in NMDS and perMANOVA hypothesis tests.

To determine whether fish assemblage was driven by environmental parameters, we performed a redundancy analysis (RDA). RDA is a constrained ordination tool that combines regression and principle components analysis for modelling a multivariate set of response variables, such as fish community abundance, against external parameters that could contribute to the formation of an assemblage. RDA is particularly useful because it permits the testing of hypotheses, for instance hypotheses regarding drivers of species composition. Using RDA, we tested the influence of stream/drain width, depth, temperature, velocity, sinuosity, vegetation, cover, and structure, as well as the substrate composition (percentage of fine, medium, and coarse grain substrate), and the hydrological characteristics (percentage of glides, riffles, and pools) of each transect against the fish community. We pre-specified a model with undercut, riparian vegetation type (meadow, scrubland, and forest), channel width, channel depth, temperature, pool availability, fine substrate percentage, vegetation, structure, cover, and sinuosity as potentially important predictor variables; however, the model did not converge properly and we removed cover and sinuosity. The full model was retained as the final model. The response variables, species abundances, were transformed to Hellinger distance (Legendre and Gallagher 2001) with the function *decostand* in vegan. Non-parametric ANOVA-like permutation tests (function anova.cca in vegan) were performed to assess significance of environmental variables on the axes of the RDA using 1000 permutations to develop a global test of the RDA result using a pseudo-F statistic against a null hypothesis that the observations were attributable to random variation (Oksanen et al. 2015).

Finally, we tested for temporal differences in fish assemblage. We separated data into eight sampling months and conducted eight NMDSs to determine how species assemblage changed across time and used perMANOVA for quantitative comparison among reaches (see above). All analyses were carried out in R (R Core Development Team 2014) and statistical tests were assessed at $\alpha = 0.05$ unless otherwise specified.

Results

In total, 6719 fish representing 22 species were captured for this study. Banded killifish (*Fundulus diaphanus*) dominated the assemblage across months and reaches (relative abundance of 45.7%; Table 2). Although there were many different species identified in the watershed, most were rare occurrences, with 15 of the 22 species having relative abundance <5.0%. Species diversity was highest in Watts (1.78), lowest in the Drain (1.02), and intermediate in Main (1.31). Kizell Drain supported only a few species and the assemblage was distinct from Watts according to NMDS (Fig. 2) and the perMANOVA hypothesis test. The Main reach of Watts Creek (i.e., below the confluence of the other two upstream reaches) had a fish assemblage that was a mixture of those in the Drain and Watts (Fig. 2) and there was heterogeneity in species assemblage among the three reaches (perMANOVA test, pseudo-$F_{2,9} = 3.37$, $p = 0.02$). Creek chub (*Semotilus atromaculatus*) was the primary indicator species in the system and was associated with Watts (sign association test, $p = 0.02$).

Redundancy analysis characterized the Drain as a reach with fine substrates, abundant in-stream vegetation, and riparian meadow-type vegetation whereas Watts was associated with a larger percentage of pool habitat, deeper channels, and more bank undercuts (Fig. 3). The habitat quality]variables that were measured in each transect explained part of the variation in fish assemblage among transects (Fig. 3; pseudo-$F_{10,1} = 4.62$, $p = 0.09$). Three variables (undercuts: pseudo-$F = 8.03$, $p = 0.04$; temperature: pseudo-$F = 13.26$, $p = 0.01$; riparian vegetation category: pseudo-$F = 7.03$, $p = 0.04$) were found to individually contribute to the final RDA model.

Watts had the highest species diversity throughout the year whereas the Drain consistently had the lowest (measured using the Shannon-Weiner index; Fig. 4); the exception was during the month of June, when diversity was highest in the Drain and lowest in Watts. Separation of fish abundance measurements by month indicated that the fish community remained distinct among Watts, Main, and the Drain throughout the year (see Fig. 5 and associated p-values from perMANOVA). Interestingly, there were four main indicator species within the system, but their relative influenced varied among systems and months sampled (Table 3). Creek chub were the indicator species in Watts from June to November, and central mudminnow (*Umbra limi*) was an indicator for the Drain in October and April. Bluntnose minnow (*Pimephales notatus*) were an indicator for Main during June and May, and Banded killifish were indicators for Main and the Drain in November and April, respectively (Table 3).

Discussion

Habitat and fish assemblage within and proximate to a stormwater drain

The fish assemblage in Kizell Drain was less diverse than that in Watts. In The Drain the assemblage was predominantly composed of central mudminnow and banded killifish; however, it is important to note that in a simultaneous study Bliss et al. (2015) found using passive integrated transponder tracking that creek chub and white sucker moved freely in and out of the Drain. The fish assemblage in Main was a combination of the two upstream reaches (i.e. of Watts and the Drain), probably because Main retained many of the physical characteristics of the natural stream but received water from both Watts and the Drain. Finally, the more natural Watts reach supported a fish community predominantly composed of a few highly abundant species (i.e., creek chub, banded killifish and longnose dace; Table 1) as well as many rare species that were encountered at different times of the year and in different

Table 2 Life-history characteristics (thermal regime, and spawning guild: Coker et al. 2001) and the relative abundances for all 23 fish species found in Watts Creek, Kanata, Ontario. The spawning guild designations correspond with the following: Lithophil = Rock and gravel spawners, Phytophil = Plant spawners, Psammophil = Sand spawners, Ariandnophil = Gluemaking nesters, Speleophil = Cave spawners, Phytolithophil = Nonobligatory plant spawners, Polyphil = Miscellaneous substrate and material nesters (Balon 1981)

Scientific Name	Code	Common Name	Thermal Regime	Spawning Guild	Relative Abundance (%)			
					Drain	Main	Watts	Watershed
Fundulus diaphanus	FUDI	Banded Killifish	cool	Phytophil	61.61	56.23	26.32	45.71
Semotilus atromaculatus	SEAT	Creek Chub	cool	Lithophil	4.48	2.95	28.36	13.11
Rhinichthys cataractae	RHCA	Longnose Dace	cool	Lithophil	0.60	6.88	16.23	9.25
Pimephales notatus	PINO	Bluntnose Minnow	warm	Speleophil	2.84	12.10	8.06	8.69
Umbra limi	UMLI	Central Mudminnow	cool/ warm	Phytophil	13.52	9.83	2.95	7.90
Culaea inconstans	CUIN	Brook Stickleback	cool	Ariandnophil	12.70	5.73	6.41	7.38
Catostomus commersonii	CACO	White Sucker	cool	Lithophil	3.06	4.43	7.60	5.39
Luxilus cornutus	LUCO	Common Shiner	cool	Lithophil	0.15	0.58	3.07	1.46
Cyprinus carpio	CYCA	Common Carp	warm	Phytophil	0.00	0.29	0.12	0.16
Percina caprodes	PECA	Logperch	cool/ warm	Psammophil	0.00	0.29	0.08	0.15
Chrosomus eos	CHEO	Northern Redbelly Dace	cold/ cool	Phytophil	0.52	0.11	0.00	0.15
Lepomis gibbosus	LEGI	Pumpkinseed	warm	Polyphil	0.30	0.11	0.08	0.13
Etheostoma spp.	ETSp	Darter species	N/A	N/A	0.00	0.25	0.04	0.12
Etheostoma exile	ETEX	Iowa Darter	cool	Phytolithophil	0.07	0.14	0.08	0.10
Margariscus nachtriebi	MANA	Northern Pearl Dace	cold/ cool	Lithophil	0.07	0.00	0.19	0.09
Pomoxis nigromaculatus	PONI	Black Crappie	cool	Phytophil	0.00	0.00	0.15	0.06
Rhinichthys atratulus	RHAT	Blacknose Dace	cool	Lithophil	0.00	0.00	0.08	0.03
Notropis heterolepis	NOHE	Blacknose Shiner	cool/ warm	Psammophil	0.07	0.00	0.04	0.03
Pimephales promelas	PIPR	Fathead Minnow	warm	Speleophil	0.00	0.00	0.04	0.02
Notemigonus crysoleucas	NOCR	Golden Shiner	cool	Lithophil	0.00	0.00	0.04	0.02
Notropis hudsonius	NOHU	Spottail Shiner	cold/ cool	Psammophil	0.00	0.04	0.00	0.02
Perca flavescens	PEFL	Yellow Perch	cool	Phytolithophil	0.00	0.00	0.04	0.02

parts of the system. Although Watts consistently had high species diversity throughout the year, diversity was lower in Watts and higher in the Drain during June, a phenomenon that we cannot explain, except that the high diversity in the Drain during June was driven by just a few instances of rare species (i.e., northern redbelly dace and blacknose shiner) occurring in the Drain.

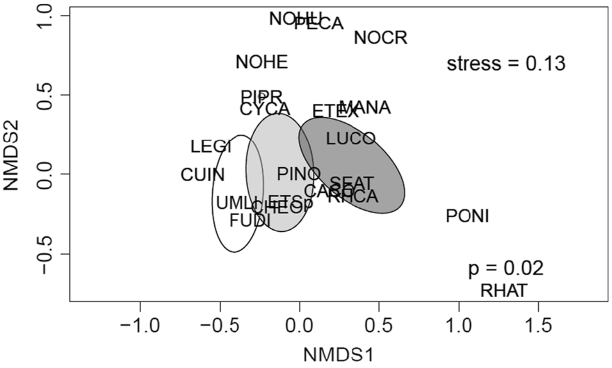
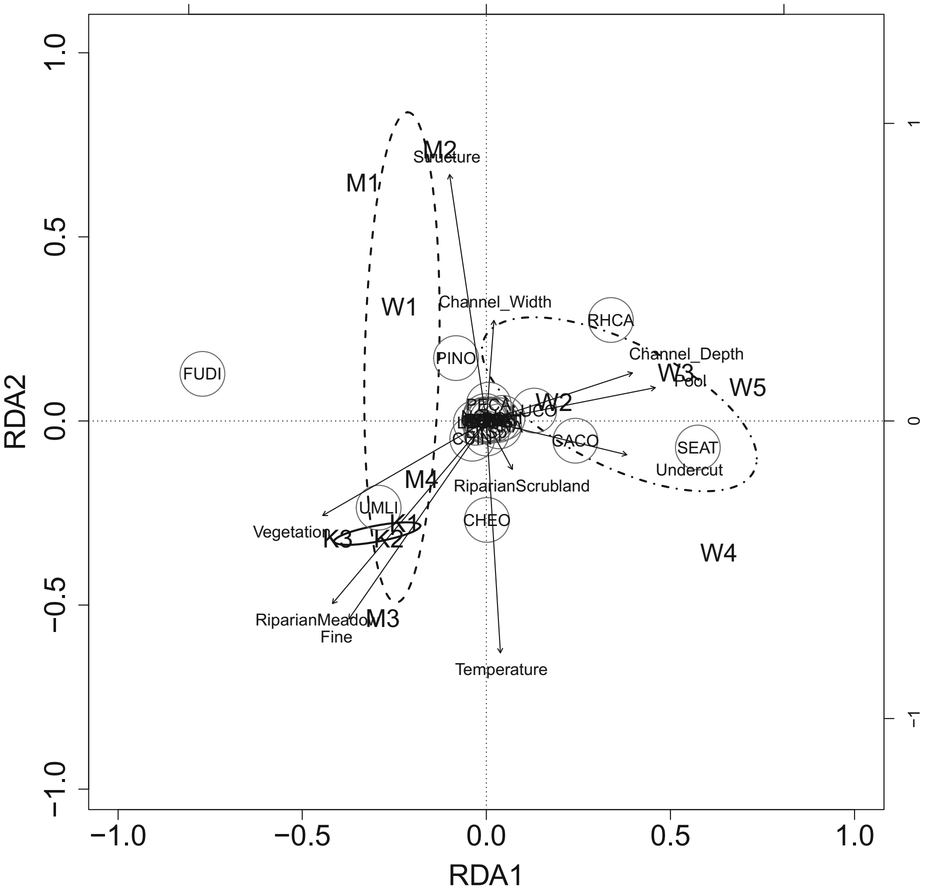


Fig. 2 Nonmetric multidimensional scaling of the species assemblage in the Watts Creek watershed. Shaded ellipses delineate the three reaches of the watershed compared in this study, with white representing Kizell Drain (*three transects*), dark gray representing Watts Creek (*five transects*), and light gray representing Main (*four transects*). The *p*-value is the result of a permutated MANOVA comparing the assemblage among the three reaches and stress is a goodness of fit metric. Refer to Table 2 for species codes

The physical environment in the watershed defined the species assemblage within the system, which was inhabited primarily by cool water fishes. Considering the Drain is less shaded (a common feature of urban streams because of encroachment into riparian habitat; Booth and Jackson 1997; LeBlanc et al. 1997) and a receiver of urban runoff directly from a stormwater management pond (which typically increases water temperatures; Van Buren et al. 2000), the drain tended to be warmer than Watts (Stantec Consulting Ltd 2011; also see Online Resource 1 for monthly temperature data). Temperature was found to be a significant predictor of the fish community and probably resulted in separation of the community into thermal guilds, with coolwater species often absent from the Drain. Increased scouring of stream bottoms from intensified storm flows, which can increase channel depth, is also commonly observed in urban streams; however,

Fig. 3 Redundancy analysis of the fish community in the Watts Creek watershed. Arrows demonstrate relationships between environmental variables, fish assemblage, and reaches of the watershed. Ellipses delineate the three reaches of the watershed compared in this study. The solid line surrounds Kizell Drain, the dashed line surrounds Main, and the hybrid dashed/ dotted line surrounds Watts. Refer to Table 2 for species codes. Structure and temperature are not pictured



the degree of sediment deposition varies among watersheds (Walsh et al. 2005). Higher pool availability was associated with Watts and negatively associated with the Drain (Fig. 3) likely because of sedimentation. Pool habitat is an important component of habitat heterogeneity that contributes to oxygenation and provides habitat for many aquatic invertebrates and fish (LeBlanc et al. 1997; Wood and Armitage 1997). Furthermore, lithophilic fishes such as white sucker, creek chub, and longnose dace, which use rock and gravel as spawning substrate, were predominantly found where there was higher habitat heterogeneity (more riffles and pools) and larger substrates as found in Watts and Main. Helms et al. (2005) found a similar trend in which lithophilic fishes were rare in urbanized watershed and Fitzgerald et al. (1999) suggested that creek chub can be a useful sentinel species in degraded watersheds. Conversely, phytophilic fishes such as banded killifish, northern redbelly dace (*Chrosomus eos*), and central mudminnow, which rely on aquatic vegetation for spawning, were associated with the Drain. These phytophils were scarce in Watts where there is minimal in-stream vegetation.

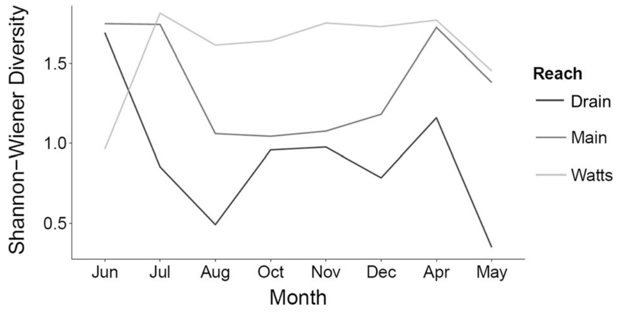


Fig. 4 Species diversity in the Watts Creek watershed by month. Species diversity is measured by the Shannon-Weiner index. June–December were sampled in 2012 whereas April and May were sampled in 2013; months are therefore ordered by sampling chronology

Urbanization generally causes biotic homogenization in which a few species become highly abundant, potentially excluding specialist species (Walsh et al. 2001; Helms et al. 2005; Meador et al. 2005; Walsh et al. 2005; Rahel 2010). Even though the Drain supported fish and therefore provided some necessary elements of fish habitat, species diversity in the Drain was low. The physical habitat characteristics in the Drain included high temperatures and homogenized substrate, which we attribute to the effects of urbanization of the stormwater drain. Channelization removes complex substrates and habitat structure that creates the riffles and pools found in Watts (Keller 1978). Ditches and drains receive runoff from urban areas, conveying chemical, thermal, and particulate pollutants that influence water quality (Hynes 1970). In addition, riparian vegetation can influence sedimentation rates of streams. Mostly, we focused on the physical alterations to

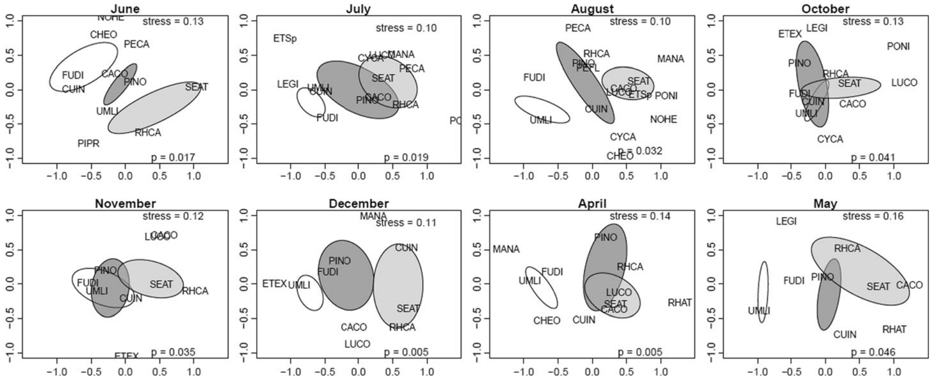


Fig. 5 Nonmetric multidimensional scaling of the species assemblage in the Watts Creek watershed, separated by month. Shaded ellipses delineate the three reaches of the watershed compared in this study, with white representing Kizell Drain (*three transects*), light gray representing Watts Creek (*five transects*), and dark gray representing Main (*four transects*). The *p*-values are the results of a permutated MANOVA comparing the assemblage among the three reaches and stress is a goodness of fit metric. June–December were sampled in 2012 whereas April and May were sampled in 2013; months are therefore ordered by sampling chronology. Refer to Table 2 for species codes

the Drain resulting from its conversion to a drain rather than the impacts of water chemistry changes. It can be difficult to separate the effects of urbanization, particularly as it pertains to water temperature and sedimentation given that high temperatures and turbidity in the Drain could both arise from urban runoff, decreased shading and flow, or bank erosion. Further research would be necessary to explicitly separate the physical alterations from the land use changes in urbanization that result in conveyance of pollutants; however, for the purposes of this study both have impacts on stream integrity and fish habitat that we captured in our analysis (Freeman et al. 2007).

Temporal variation in assemblage

Effects of urbanization have been documented both at various spatial (Weaver and Garman 1994; Stephenson and Morin 2009; Stanfield 2012) and temporal (Wang et al. 2001; Schweizer and Matlack 2005) scales. In the Watts Creek watershed, differences in the structure of the assemblage were consistent among months with distinct assemblages forming in each of the eight months that were sampled. There was limited overlap between the fish communities in the Drain and Watts but Main represented an intermediate assemblage between the two upstream reaches. Correspondingly, there was overlap between Watts and Main throughout most of the year whereas the fish community in the Drain and Main overlapped primarily in October and November. Stream fishes are highly mobile within their system and are capable of temporal shifts in assemblage, which likely relate to fish seeking optimal habitat that changes with the seasons. Generally, it is thought that stream fish shift habitats in the winter when foraging is less important than seeking shelter (Schlosser 1991; Brown et al. 2011). Furthermore, the assemblage differences that we identified were consistently driven by the same highly abundant species. Sign association tests indicated a small number of important species among the 22 identified within the watershed. Specifically, the monthly dynamics of banded killifish, central mudminnow, bluntnose minnow, and creek chub were important drivers of the differences that we observed among reaches within the Watts Creek watershed. This demonstrates that the spatial dynamics of rare species had limited effects on the establishment of assemblages within the watershed; however, there were many rare or uncommon species that were mostly identified in Watts or Main, which probably contributed to the differentiation of the reaches even if this was not clear from sign association tests.

The selection of overwintering habitat is critical for the survival of stream fishes (Cunjak 1996). Previous research has identified early winter as a stressful period for stream fish as temperatures decline and metabolism decreases, which affects energetic demands (Cunjak 1988). During the winter, fish economize energy expenditure and seek habitat where

Table 3 Indicator species in the Watts Creek watershed by month for 2012 and 2013. Indicator species were determined using the sign association test and *P*-values were calculated using Sidak's correction. June–December were sampled in 2012 whereas April and May were sampled in 2013; months are therefore ordered by sampling chronology. Refer to Table 2 for species codes

	Drain	Main	Watts
June		PINO*	
July			SEAT***
August			SEAT**
October	UMLI*		SEAT*
November		FUDI**	SEAT**
April	UMLI**, FUDI**		
May		PINO***	

* 0.10 > P > 0.05

** 0.05 > P > 0.01

*** P < 0.01

energetic costs are low and the environment is relatively predictable (Cunjak 1996). Overwintering areas are generally characterized by low velocity with suitable in-stream cover and increased habitat volume (Schlosser 1991; Cunjak 1996); however, habitat selected during the winter varies among species. For example, longnose dace and white sucker overwinter in deeper habitats with rubble cover even though these habitats tend to exhibit increased velocities (Cunjak and Power 1986). Watts and Main provide overwintering habitats with greater habitat complexity and instream cover, whereas the Drain was dominated by shallow glides with minimal structure, making it a less suitable overwintering location. Nonetheless, banded killifish and central mudminnow were found in the Drain in early winter. Bliss et al. (2015) found that PIT tagged central mudminnow were seldom detected leaving the Drain throughout the winter months. The morphology of these species may allow them to overwinter in the drain where oxygen supply could rapidly deplete during the winter because of low flow (Lewis 1970; Klinger et al. 1982). The flattened head and upturned mouth of banded killifish allows them to access oxygen at the air-water interface that species with rounded heads cannot (Lewis 1970). This adaptation may also allow banded killifish to access bubbles under surface ice or oxygen through cracks in the ice. Although central mudminnow lack the same adaptation, they have the ability to absorb more oxygen from bubbles trapped under surface ice or from cracks in ice and can extract oxygen from water taken into a highly vascularized gas bladder (Klinger et al. 1982). Regardless of the mechanism, it is clear that these species are able to exploit habitat in earthen stormwater drains all year round, including during harsh winter months of the temperate north.

Management implications

Regardless of whether drains support fish communities unique to neighbouring or connected watersheds, these systems are capable of providing habitat for some fish species. Nevertheless, the physical characteristics associated with physical modification, including finer substrate, lower flow, higher temperature, reduced hydrological complexity, and higher instream vegetation combined to create a homogenous habitat that did not attract the full suite of fish diversity associated with the watershed at large, meaning that the drain habitat was degraded. Reduced environmental complexity in the Drain corresponded with low diversity and high relative abundance of tolerant species, indicating that urbanization reduced habitat quality and excluded the majority of fish species. In contrast, Stammler et al. (2008) found that streams and drains in Southwestern Ontario had similar fish assemblages. The difference between our results could be attributed to the physical connectivity between the drain and stream. In the study by Stammler et al. (2008) their drains and reference streams were not physically connected, while in the present study the physical connection between Watts and the Drain, along with the availability of appropriate habitat in Watts, could mean that some fish species did not need to use the comparatively degraded habitat in the Drain. Essentially, since fish assemblages in all three evaluated reaches drew from the same pool of available species, the present study confirms that a comparatively degraded municipal drain does not provide the same type of habitat as a more natural creek.

Despite being a stream altered for stormwater conveyance, the Drain provided some suitable fish habitat but modifications contributed to a homogenized fish assemblage. An important caveat, however, is that we evaluated only one system across a single year, so our findings may not be transferable to all other urban streams or earthen stormwater drains. Further research is necessary to evaluate the observed trends across a larger landscape and over longer periods of time. More research is also needed to compare the relative effects of physical alterations to the water quality changes associated with runoff. Such information will help inform remediation and restoration efforts for enriching these habitats. It is also necessary to evaluate the potential for remediation of systems where headwaters remain modified (Booth and Jackson 1997). Many countries are already shifting their perception of stormwater management, transitioning from it being considered a nuisance and risk to human health and infrastructure to encouraging more sustainable management approaches. Enhanced understanding of the ecology of drains, including their fish assemblages, will assist with developing better management of systems impacted by urbanization (Roy et al. 2008). However, policies that regulate urban infrastructure and aquatic biological systems remain separated. In Canada, stormwater management falls under the jurisdiction of municipalities and is regulated provincially (i.e. Ontario has the Drainage Act [R.S.O. 1990, c. D.17]), whereas the management of natural freshwater systems falls under several jurisdictions depending on the property owner and is regulated under the federal Fisheries Act (R.S.C., 1985, c. F-14). Despite the regulatory differences, joint management of urban and natural systems is possible and would be beneficial (Roy et al. 2008). Earthen stormwater drains such as the Drain are only a fraction of the whole stormwater management system in urban areas, but the results of this study suggest that such altered streams need to be considered as inter-connected systems that provide habitat with the potential to support fish assemblages. Further research may identify ways in which remediation of drains can translate the physical connections with lower order streams into more healthy biological systems in urban areas.

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