Seasonal Variation in Assemblage Structure and Movement of Small Stream Fish in an Urban Environment

by

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Dedication

I dedicated this thesis to my wonderful family! To my mom and dad for all their strong encouragement, support, and wisdom over the years. You have both played a huge role in shaping who I am as a person and I am deeply blessed to have such loving parents. To my sister (and niece) for her big heart and never-wavering love and belief in me. Although we don’t always see eye-to-eye we have a bond that can never be broken. And finally, to my husband Barry for going beyond anything I could have asked in order to support me through everything. I am incredibly fortunate to have a partner as passionate, caring, and understanding as you.

Abstract

Urban ecology is a discipline that has emerged in response to the unique changes associated with urbanization. Research on ecosystem impacts is still lacking, particularly on fish populations within urban stormwater drains. My research goal was to study the assemblage and movement of stream fish within Watts Creek, an urban stream, and a stormwater drain tributary (Kizell) in Kanata, Ontario. In chapter 2 I compared fish assemblage structures of Kizell and Watts and found they were relatively distinct. In chapter 3, using passive integrated transponder technology, I showed that the directionality of movements between Kizell and Watts had little variation. These findings demonstrated the connectivity between Kizell and Watts, and that stream fish are moving into, residing, and utilizing habitat within Kizell throughout the year (including during the winter). These findings suggest that stormwater drains are a functional component of urban stream systems and that drains and streams are interconnected systems.

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Co-Authorships

Chapter 2: Seasonal Variation in Fish Assemblage Structure in a North Temperate Urban Stream and Municipal Stormwater Drain. S.M. Bliss, J.D. Midwood and S.J. Cooke

Although this study is my own, the research was undertaken as part of a collaborative effort, and each co-author played a valuable role in its completion. The project was conceived by Bliss, Cooke and Midwood. Fieldwork was conducted by Bliss and Midwood. All data analysis was conducted by Bliss. Data were interpreted by Bliss, Cooke, and Midwood. All writing was conducted by Bliss. All co-authors provided comments and feedback on the manuscript. The manuscript is in preparation for submission to a peer-reviewed journal.

Chapter 3: Seasonal Movements of Small-bodied Fish in a North Temperate Urban Watershed Demonstrate Connectivity between a Stream and Stormwater Drain. S.M. Bliss, J.D. Midwood, K.M. Stamplecoskie, and S.J. Cooke

Although this study is my own, the research was undertaken as part of a collaborative effort, and each co-author played a valuable role in its completion. The project was conceived by Bliss, Cooke and Midwood. Fieldwork was conducted by Bliss, Midwood, and Stamplecoskie. All data analysis was conducted by Bliss. Data were interpreted by Bliss, Cooke, and Midwood. All writing was conducted by Bliss. All co-authors provided comments and feedback on the manuscript. The manuscript is in preparation for submission to a peer-reviewed journal.

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Glossary

ANOSIM: Analysis of similarities

df: Degrees of freedom

IBI: Index of biotic integrity

TL: Total length

NCC: National Capital Commission

SD: Standard deviation

SIMPER: Similarity percentages

Chapter 1. General Introduction

It has become increasingly important to understand the influence of urbanization on organisms and ecosystems as urban areas continue to expand. In 2010, 50.5% of the world's population resided in urban areas and it is predicted that the majority of the world's population growth will occur within these areas over the next four decades (United Nations 2011). As a result, urban ecology has developed into its own discipline in recognition of the unique features of urban ecosystems (Rebele 1994). Urban aquatic systems are valued for their many social and economic services, such as recreation, irrigation, drinking water, hydroelectric power, and stormwater management. However, anthropogenic activities contribute to changes in lotic environments, including altering water flows, stream morphology and habitats, as well as the addition of contaminants (Paul & Meyer 2001). These changes in streams usually cannot be attributed to a single activity or stressor, but rather the cumulative effect of various human activities (Konrad & Booth 2005). By recognizing the complex changes that accompany urbanization, researchers are beginning to formulate concepts, building upon the River Continuum Concept (Vannote et al. 1980), that include the unique features of urban aquatic systems. Walsh et al. (2005) reviewed and summarized the Urban Stream Syndrome concept, which describes common impacts and changes to streams as 'symptoms' of urbanization. These symptoms include a flashier stream hydrograph, altered channel morphology and stability, increased concentrations of pollutants and nutrients, and reduced biota richness usually due to the dominance of the most tolerant species (Walsh et al. 2005). Kaushal & Belt (2012), on the other hand, proposed the Urban Watershed Continuum which focuses more on the transfer and transformation of matter and energy. Their concept proposes that first order streams tend to be replaced by infrastructure (i.e. stormwater drains, pipes, or ditches), which modifies the retention of organic carbon and nutrients, leads to amplified downstream pulses of energy and material, and that leaky pipes, along with groundwater, influence the transportation of solutes to streams as a result of aging infrastructure (Kaushal & Belt 2012). The major common element between these concepts and their relationship to urban ecosystems is the addition of impervious surfaces and infrastructure to manage excess water (i.e. stormwater management systems).

Water management is a key component of both urban and agricultural systems, and involves the construction of ditches or drains for the purpose of removing excess water from the surrounding landscape. There are various types of stormwater drains (e.g. man-made or earthen) which are either characterized as surface drains (flows over the ground and is open to the surrounding environment) or subsurface drains (flows below ground and is closed to the surrounding environment; Djokic & Maidment 1991). While subsurface drains tend to be pipes underground made of metal or concrete, comparatively many surface drains can be more natural (also referred to as ditches or swales). Although these earthen surface drains are only a portion of the overall stormwater management systems, often they were once natural headwater streams that were converted for human-related services (Kaushal & Belt 2012). Such surface drains may still be connected to the overall watershed and have a direct impact on streams (Walsh et al. 2005). Within agricultural environments, drains are aimed at protecting crops from excess soil moisture and enhancing plant growth by replacing water with oxygen within the soil (Stammler, 2005). In urban environments drains are intended to remove excess water that collects from increased impervious surfaces (i.e. pavement, roads, buildings) which poses a threat to human safety and infrastructure (Roy et al. 2008). The concentrated water and contamination runoff from impervious surfaces ultimately alters the biochemical composition and flow regime of stream environments (Hatt et al. 2004). Although the differential impacts of urban and agricultural drains on aquatic biota has not yet been clearly defined, research suggests that fish community impairment occurs at a lower percentage of urban land use than agricultural land use. For example, in Wisconsin Wang et al. (1997) found that the index of biotic integrity (IBI) scores for fish declined when the landscape was between 10 – 20% urbanized, while a decline in IBI scores was only obvious within landscapes that were >50% agricultural. Thus, urban areas may have a disproportionately large influence on aquatic biota (Paul & Meyer 2001).

To date, studies have revealed that agricultural drains support multiple taxa (e.g. macrophytes, invertebrates, or amphibians; Herzon & Helenius 2008); however, the relationship of biota to urban stormwater drains is less understood. Studies of the impact of urbanization on streams have focused on stream morphological changes, transportation of contamination and sediment, and macroinvertebrates or fish community structure (Berkman & Rabeni 1987; Hatt et al. 2004; Sponseller et al. 2001). Less is known about fish dispersal and movement within urban stormwater drainage systems or how assemblage structure varies across seasons. Furthermore, most studies are conducted during the summer resulting in major gaps in our understanding of how fish in urban streams behave or are distributed throughout the entire year, including the winter months (Brown et al. 2011). The studies that have been conducted focus primarily on salmonids (Cunjak 1996), which comprise only a fraction of the freshwater fish families found in Ontario and Canada (Scott & Crossman 1998).

Ottawa, the capital city of Canada, is the second most populous city in Ontario (Statistics Canada, 2012a). The population has grown by 8.8% in the past 5 years (2006 to 2011; Statistics Canada, 2012b). Kanata is the largest suburban area within the Ottawa region and further growth and development is already underway (Water Environment Protection Program 2012). It is located on the far west end of Ottawa with a section of the National Capital Greenbelt dividing the two urban areas. The National Capital Commission (NCC) is a federal agency responsible for the management and conservation of the greenbelt. Currently in Kanata, stormwater runoff is collected and held in the Beaver Pond stormwater management facility and released periodically into the Kizell Municipal Drain. Kizell Drain is a highly channelized natural surface drain that undergoes persistent human activity (in the form of stormwater management and maintenance including the removal of debris and vegetation with heavy equipment) and is a tributary of Watts Creek. In contrast, Watts Creek is a comparatively protected system, which is predominately found within the greenbelt and drains directly into the Ottawa River via Shirley's Bay. While it is an urban stream, it experiences considerably less direct human activity (though not devoid of anthropogenic impacts) than Kizell Drain.

To meet the needs of a growing suburb, the Kanata Lakes North Development proposed to divert more water from another local watershed (Shirley's Brook) into Watts Creek (Water Environment Protection Program 2012). This raised many concerns, particularly within the NCC, regarding how a large addition of water would change the flow regime and ultimately impact the downstream aquatic environment. In an effort to better understand the existing conditions of Watts Creek I monitored the fish community over a one year period. The overarching research goal is to assess how small stream fish respond to persistent human activity in an urban environment. To accomplish this I evaluated the assemblage structure within Watts Creek and Kizell Drain (Chapter 2) and monitored movement patterns of small stream fish between these systems (Chapter 3). Monitoring and sampling occurred across all four seasons, including winter. The objective of chapter 2 was to compare fish assemblages in Watts Creek and Kizell Drain across seasons, and determine which species within the system were driving the observed patterns. The objective of Chapter 3 was to assess the connectivity between Watts Creek and Kizell Drain by evaluating the extent and timing of fish movements between these systems. For each season, I compared the direction of movements through the confluence among the four most common species: central mudminnow (Umbra limi), creek chub (Semotilus atromaculatus), longnose dace (Rhinichthys cataractae), and white sucker (Catostomus commersonii). I also characterized the diel movement patterns and compared the residency times between Watts Creek and Kizell Drain for the aforementioned species. Given that the physical connectivity of Watts Creek and Kizell Drain provides an excellent comparative opportunity, this research will help to understand whether fish are moving into and utilizing habitat in degraded areas, such as stormwater drains. This information will be valuable to fisheries biologists, conservation agencies, and land use planners in assessing priorities for sustainable ecosystem management.

Chapter 2. Seasonal Variation in Fish Assemblage Structure in a North Temperate Urban Stream and Municipal Stormwater Drain

2.1 Abstract

Urban watersheds often include tributaries that range from near-pristine to those that are heavily altered and function primarily as municipal storm drains. In this study I used non-metric multidimensional scaling to characterize the fish assemblage structure in an urban watershed across seasons including winter. I performed an analysis of similarities (ANOSIM) and similarity percentages (SIMPER) analysis to determine whether assemblages differed among three reaches of a contiguous urbanized stream and identify which species were driving the observed assemblage patterns. Sampling efforts focused on an urban stream in Kanata, Ontario (Watts Creek), an earthen municipal surface stormwater drain (Kizell Drain), and the area downstream of their confluence (herein termed Main). Based on abundance data collected from 23 species, Watts Creek and Kizell Drain were characterized by seasonally distinct assemblages in the summer, winter, and spring. The assemblage structure within Main overlapped with Watts and Kizell in the summer and winter, but in the spring was distinct from Kizell and similar only to Watts. In the fall there was no difference in the assemblage composition among the three reaches. Banded killifish (Fundulus diaphanous) contributed most to the observed dissimilarities. While distinct assemblages were identified throughout most of the year, there was some overlap in the species composition among the reaches. I conclude that earthen surface drains can provide habitat that supports fish communities across life stages and seasons. The biological integrity of aquatic ecosystems would

benefit if resource management agencies treated surface drains and the streams or rivers they flow into as an inter-connected systems.

2.2 Introduction

In recognition of the unique features of urban ecosystems, the discipline of "urban ecology" aims to understand the influence of urbanization on organisms and ecosystems (Rebele 1994). Streams and rivers are the hydrological 'highways' that connect various landscape elements (including riparian areas) and serve as corridors in urban areas (Walsh et al. 2005). However, anthropogenic alterations can cause disturbances in lotic environments that can reduce biodiversity and alter habitats (Dudgeon et al. 2006; Urban et al. 2006). The term "urban stream syndrome" has therefore been used to describe the ecological degradation witnessed in developed landscapes, and symptoms include flashier systems, increased nutrient and contaminant concentrations, and reduced biotic richness (Walsh et al. 2005). Typically such changes in streams cannot be attributed to a single activity or stressor, but rather the cumulative effects of various human activities (Paul & Meyer 2001; Konrad & Booth 2005).

A necessary component of urban landscapes is a stormwater management system, which helps funnel excess water from developed areas into natural bodies of water by means of drains. Of the various types of stormwater drains (man-made vs earthen and surface vs subsurface; Djokic & Maidment 1991) often earthen surface drains (also referred to as ditches or swales), which were once natural headwater streams that were converted for human-related services (Kaushal & Belt 2012), are still connected to the overall watershed. Consequently, these systems directly link human activities to neighbouring aquatic environments, especially streams. Klein (1979) observed reductions in aquatic diversity and stream quality with an increase in the proportion of impervious surfaces in the watershed. This was further refined by Wang and colleagues (2000) who suggested 10 percent imperviousness within the watershed as a threshold for major negative changes to aquatic environments. Changes in aquatic habitat and the dispersal of fish species in these environments are sensitive to flooding and drought, which can be exacerbated by altered flow regimes in urban areas (Brown et al. 2005). Variations in flow regimes are important in shaping fish community structure by influencing food availability, habitat, and temporal variations in stream hydrology (Poff & Ward 1989; Marchetti & Moyle 2001; Konrad & Booth 2005); perhaps most crucially during the winter when fish are already faced with the added physiological challenge of lower temperatures (Cunjak 1988). Indeed, most studies of fish assemblage focus on non-winter periods despite that winter is regarded as a significant selective force in north temperate regions (Cunjak 1996).

Despite their connection, streams and drains are governed differently in most jurisdictions. While streams are recognized for their ecological function as aquatic habitat, drains are typically not regarded the same way from a legislative perspective. Drains, usually governed by municipalities, may be regularly maintained (i.e. removal of blockages with heavy equipment) for optimum drainage performance, and are subject to channel straightening, widening, and even relocation as urban areas expand. Recent research in agricultural drains reveals that they are an important source of fish habitat (Stammler et al. 2008); however, similar research in urban systems is lacking. Indeed, although there is a growing body of literature that demonstrates the impacts of urbanization on stream fish assemblages, I am unaware of studies that have explicitly compared fish communities both within and proximate to surface stormwater drains across seasons.

While species richness can increase or decrease with increasing urbanization, fish assemblages are likely to be functionally less diverse in urban areas compared to nonurban areas (Walsh et al. 2005). For example, the same number of species could be present in both a degraded and a pristine stream, but the disturbance tolerance or spawning type composition of those communities may be different. Common tolerant-species increase in abundance while intolerant species decrease as a result of urbanization (reviewed in Meador 2005 and Walsh et al. 2005). This generally leads to the homogenization of assemblages, even at relatively low levels of urbanization (Wang et al. 2000). Homogenization is of concern because it represents the replacement of regionally unique species with wide-spread generalists (Rahel 2010). Such homogenization has been observed across various temporal (Wang et al. 2001; Schweizer & Matlack 2005) and spatial scales (Weaver & Garman 1994; Stephenson & Morin 2009; Stanfield 2012). With regard to urban streams, most effort directed at improving stream ecosystems has focused on physical variables (i.e., bank stabilization) with less focus on remediating or maintaining the stream biota (Paul & Meyer 2001). With regard to stormwater drains, some research has suggested that the fish community does not differ between surface drains and streams in an agricultural setting (Stammler et al. 2008), but little is known about whether the structure of fish assemblages between urban streams and surface stormwater drains are maintained across seasons.

Using the Watts Creek watershed in Kanata, Ontario, Canada, I studied the community ecology of stream fish in an urban stream and a contiguous municipal surface stormwater drain (Kizell Drain) using ordination techniques and an Analysis of Similarities (ANOSIM). My first objective was to compare fish assemblages among the stream (both upstream and downstream of the stormwater confluence) and stormwater drain across seasons. My second objective was to determine which species were driving the observed seasonal assemblage patterns.

2.3 Methods

2.3.1 Study area

The land use within the Watts Creek watershed (~2500 ha), located in Kanata, Ontario, Canada (45º20'42"N, 75º52'19"W), is approximately 47% agricultural, 35% developed, and 18% undeveloped (Stantec Consulting Ltd. 2011). Soils in this region are characterized as a layer of silt and clay overlying a layer of Precambrian and Paleozoic bedrock (Stantec Consulting Ltd. 2011). Watts Creek, an urban tributary of the Ottawa River, originates in the Katimavik-Hazeldean community of Kanata and flows through a residential area before entering the Ottawa Greenbelt, which is protected and managed by the National Capital Commission (NCC). The creek is predominately found within the greenbelt, which contains rural and agricultural lands (1000 ha). This natural watercourse is groundwater fed with stormwater inlets from the surrounding urban areas. Kizell Drain is an earthen surface drain designated as a municipal drainage/watercourse under the Drainage Act (R.S.O. 1990, c. D.17) and is governed by the City of Ottawa. It originates at Beaver Pond (a stormwater management area near Walden Dr., Kanata, ON) and flows into Watts Creek on NCC property (Fig. 2.1). The 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. 2.1). It is important to note that the section of Kizell Drain included in this study has not been cleaned with heavy equipment for many years (10+ years), but the section upstream from my study site likely underwent more persistent maintenance.

There are three reaches branching out from the confluence between Watts Creek and Kizell Drain, which were categorized as follows: Watts Creek upstream from the confluence (herein referred to as Watts), Kizell Drain upstream from the confluence (herein referred to as Kizell), and downstream Watts Creek (herein referred to as Main), which includes water from both upstream reaches (Fig. 2.1). Twelve transects measuring 100 m were established throughout the study area. The number of transects in each of the three reaches was selected proportionally to the stream length of that reach. Watts was the longest (~2.9 km), followed by Main (~1.7 km) and then Kizell (~1.5 km), therefore five, four, and three transects were established, respectively. The transects covered 19.4% of Watts Creek (17.1% of Watts reach and 23% of Main) and 19.6% of Kizell. The transect locations within each reach were selected based on their spread throughout the reach (toward the end and beginning of each reach), accessibility, and in order to represent various riparian habitat types (e.g., forest, scrubland and agricultural).

2.3.2 Community and habitat sampling

Fish communities were sampled from each transect once per month 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. It is important to note that the Ottawa area experienced drought from July to September 2012. In August, these drought conditions led to reduced flow and drying of Kizell, leaving only small pools of water in some sections of my transects. The August 2012 community sampling occurred during this time and fish in the remaining pools were found within transects. Flow returned to Kizell before the October 2012 sampling. 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. During the winter community assessment (11 – 17 December 2012), some transects had a layer of ice across the surface ranging from 1 – 13 cm thick. In order to access the water, the ice was broken and cleared. Regular electrofishing surveys were conducted 25 minutes after the ice was cleared to allow disturbed fish to return and the water clarity to improve. If a transect had no ice, I walked through it to mimic the disturbance caused by the ice breaking and again waited 25 minutes before sampling.

I collected several environmental variables to summarize the available habitats found within each reach. In order to capture seasonal variation in some of the variables (i.e. 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 by the straight distance between the lower and upper limits of the reach in the study site. The average overhanging, in-stream, and substrate composition was determined as the proportion (%) of a 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 macrophyte, and in-stream structure such as woody debris, course sediment (>64 mm), and detritus. Sediment type was classified as loose or consolidated clay (<3.9 µm), silt (3.9 - 62.5 µm), sand (62.5 µm – 2 mm), gravel (2 - 64 mm), cobble (64 - 256 mm), boulder (>256 mm), and bedrock following the Wentworth scale for grain sizing (Wentworth 1922). Longitudinal measurements to the nearest centimeter were taken of habitat types (pool, riffle, or run). Finally, the dominant riparian type (defined here as a width of 30 m; Wenger 1999) was described visually in the field as forest (dominated by trees with at least 10 cm diameter), scrubland (small trees of less than 10 cm in diameter and shrubs with dispersed grasses and sedges; a transition between meadow and forest), meadow (dominated by grasses and sedges; no woody vegetation present), or cultivated (manicured lawns or actively farmed lands) as outlined in the Ontario Stream Assessment Protocol (Stanfield 2005). For two transects I had to create a fifth category referred to as modified scrubland, which incorporates the presence of human alterations within approximately 50% of the riparian area. The specific alterations are as follows: the W1 transect (see Fig. 2.1) had an elevated train track running parallel to it along one of its banks and the W4 transect had a gravel utility road, an underground sewage holding tank, a farm, and a paved bike path with regularly mowed edges all along one of the banks.

2.3.3 Data analysis

Sample events were defined as the month in which a community assessment occurred. The sampling events were designated to each season as follows: summer (June, July, and August), fall (October and November), winter (December), and spring (April, May). The Shannon-Wiener index was chosen to describe diversity within the system because it takes into account species richness and evenness (McCune & Grace 2002). Three transects per reach were randomly chosen for the diversity index. Considering there was only a single sampling event for winter (December), I only included July, October, December, and April to represent each season in order to keep the sample size equal among the reaches and seasons. The July, October, December, and April sampling events were specifically chosen to represent the midpoint of each season, with the exception of winter which could only be sampled early in the season. All of the dominant species sampled were spring or spring/summer spawners (Scott & Crossman 1998; Coker et al. 2001), and so the different spawning types were summarized as the proportion of each spawning type found in Kizell, Main, and Watts for spring and summer only. The habitats for each reach were summarized by averaging all the samples collected from each transect in that reach during both sampling periods. For example, the measurements for all three transects in Kizell from both September and May were included in the average.

To evaluate seasonal community composition among transects I used nonmetric multidimensional scaling (NMDS) ordinations. NMDS is a useful graphical tool for analyzing community data because its use of rank ordered ecological distances to arrange sample units allows for easier interpretation and it does not assume linearity among variables (McCune & Grace 2002). Preliminary ordinations were made for each sampling event based on abundance data in 2-dimensions (2D) and 3-dimensions (3D). The general patterns and clustering of transects did not differ greatly (through visual inspection) for sample events within a season; for example among June, July, and August. Based on these preliminary evaluations and considering I was only able to sample once during winter (December), I used one sample event from each season to further assess seasonal variation. As with the Shannon-Wiener index, July, October, and April were chosen as the midpoints of summer, fall, and spring, respectively. 2D ordinations were chosen given their relatively low stress values and similar output patterns to the 3D ordinations. Stress is a goodness-of-fit statistic that measures the scatter of data points from the best-fit monotonic regression between the observed dissimilarity and ordination distances (Clarke 1993). This value ranges from 0 to 1 and the lower the stress value the better the fit (McCune & Grace 2002). Preliminary ordinations with and without rare species (defined as <1% of the whole system relative abundance; Table 2.2; Walser & Bart 1999) were also performed which resulted in very similar outputs. Therefore, I included rare species in the ordinations since their influence was minimal to the overall NMDS and because my objective is to assess whole community structure, so the removal of rare species would be inappropriate (McCune & Grace 2002).

An analysis of similarities (ANOSIM) on Bray-Curtis dissimilarity matrices with 999 permutations was used to determine whether the reaches significantly differed in their assemblage composition (Oksanen 2013). A similarity percentages (SIMPER) analysis was used to identify the Bray-Curtis dissimilarity between each reach (group) pair and the proportion that each species contributes to the dissimilarity (Clark 1993). Each analysis was performed using the functions “anosim” and “simper”, respectively. All NMDS ordinations were performed using the function “metaMDS”, which first square root transformed and applied a Wisconsin double standardization because of the large range of values in the data, and used the Bray-Curtis distance metric to create a distance matrix (Oksanen 2013). Ellipses, representing the 95% confidence limits of the weighted averages around the centroid of each reach, were calculated and drawn on each ordination using the function "ordiellipse". All functions used are in the Community Ecology Package "vegan" (Oksanen et al. 2013). All analyses were performed in the program R for Statistical Computing (v-2.14.1; R Development Core Team 2013).

2.4 Results

In terms of habitat, Kizell was the narrowest and shallowest reach, dominated by runs and fine sediments (Table 2.1). In-stream vegetation and structure were present, but there was very little overhanging cover (typically less than 20%). The riparian zone was primarily grasses, sedges, and small shrubs, with the most upstream site (K3) running through a farm that had cultivated lands on both sides of the waterway. Most of Kizell has been channelized in the past as evident by the low sinuosity (1.1; Table 2.1). Both Watts and Main were more sinuous (1.8 for both; Table 2.1) than Kizell, though small sections of the stream had been modified when a train track and bike path were built in the area. Watts and Main had higher habitat complexity with a mix of runs, small and deep pools, and riffles; there was even a small cascading waterfall just upstream from one of the transects in Watts (W3). While there was still a high proportion of clay and silt, medium and course sediment were more prevalent in Watts than in Kizell. In addition there were more occurrences of in-stream structure. Watts, more so than Main, frequently had overhanging cover (40% vs. 18%, respectively). The riparian of the Watts reach was mainly forest with some areas of mixed trees, grasses, and shrubs. The upstream sites of

Main had primarily grasses, sedges, and shrubs in the riparian zone while the downstream sites were mainly forest covered.

A total of 6,721 fish representing 23 species was captured, of which only 8 were considered common (>1% relative abundance within the whole system) and Fundulus diaphanous (banded killifish) was the most abundant (relative abundance of 45.71%; Table 2.2). The common species also included Semotilus atromaculatus (creek chub), Rhinichthys cataractae (longnose dace), Pimephales notatus (bluntnose minnow), Umbra limi (central mudminnow), Culaea incostans (brook stickleback), Catostomus commersonii (white sucker), and Luxilus cornutus (common shiner). Only a single individual was captured for Pimephales promelas (fathead minnow), Perca flavescens (yellow perch), Notemigonus crysoleucas (golden shiner), and Notropis hudsonius (spottail shiner). With the exception of two species (Cyprinus carpio (common carp) and Carassius auratus (goldfish), representing less than 0.2% of the total catch) all species captured are native to Ontario. The majority of species caught are considered cool water species (Table 2.3; Coker et al. 2001). Among the reaches and seasons the mean catch ranged from 16 - 331 and species richness from 7 - 13 (Table 2.3). There was no clear pattern in the mean total catch among the three reaches. When comparing among seasons, summer had the greatest mean catch followed by fall, spring, and then winter with the lowest. With the exception of spring, Kizell typically had the lowest species richness with Main and Watts generally having greater species richness (Table 2.3). The species richness was highest in the summer and lowest in the spring and winter for Main and Watts, respectively. Within Kizell the species richness was highest in fall and lower during all other seasons. The Shannon-Wiener diversity was highest in Watts and lowest in Kizell across seasons, except in spring when Main was more diverse than Watts; however, there was no clear pattern in the diversity among seasons (Table 2.3).

Across all seasons the Kizell sites clustered closer together (more homogeneous) than the sites within Main and Watts in the two-dimensional NMDS ordinations, particularly along the horizontal axis (Fig. 2.2). The Main sites had the greatest ordination spread along both axes during the summer, along the horizontal axis in winter, and along the vertical axis in fall and spring. In winter the Watts sites were dispersed roughly the same as the Main sites, with a slightly wider spread along the vertical axis (Fig. 2.2c). However, the Watts sites had the greatest spread along the horizontal axis in fall (Fig. 2.2b). The overall assemblage composition among the reaches differed significantly in summer (ANOSIM, p=0.04; Table 2.4), winter (ANOSIM, p=0.003), and spring (ANOSIM, p=0.003). Kizell and Watts were the most dissimilar in summer and winter while Main positioned between and overlapped with both groups, but was more similar with Kizell than Watts (Table 2.4; Fig. 2.2). In the spring however, Main and Watts were more similar to each other (Table 2.4) and both were highly dissimilar to Kizell as indicated by the lack of overlap between the 95% confidence ellipses (Fig. 2.2d). During the fall the assemblage composition did not significantly differ among reaches (ANOSIM, p=0.11) and the ellipses of all three reaches overlapped (Fig. 2.2).

Among the 8 most abundant species, only common shiner was not found to contribute greatly to the dissimilarity among the reaches, while the other species had variable contributions across the seasons (Table 2.4). Banded killifish had the highest proportional contribution to the dissimilarity between all reach pairs across all the season (21 - 54.9%; Table 2.4), except for between Main and Watts in spring (contribution of

12.6%) when creek chub was the highest contributor (28%). Banded killifish were most abundant in Kizell during summer, winter, and spring, and least abundant in Watts throughout the year (Table 2.4). Central mudminnow was also highly abundant in Kizell throughout the year. In contrast, creek chub were highly abundant in Watts compared to Main and Kizell (Table 2.2) and contributed to the dissimilarity between all pairs with Watts (14.2 - 28%; Table 2.4) and between Kizell and Main in fall and spring (7.9% and 10.1%, respectively). Longnose dace were also more abundant in Watts and contributed to all comparisons between Main and Watts, and the comparison between Kizell and Watts during the summer and spring.

For the spawning types of the most dominant species within the system, there are clear associations with specific reaches occurring in spring and summer. Adriadnophilic fishes (gluemaking nesters that build nests from twigs and leaves; Holm et al. 2009) were found in somewhat similar proportions among the reaches during both seasons (Table 2.5). Lithophilic fishes (rock and gravel spawners) were highly associated with Watts during spring and summer (71.8% and 73.6%, respectively). Phytophilic fishes (obligatory plant spawners) were more associated with Kizell in spring (52.3%) and Main in summer (46.9%). And lastly, Speleophilic fishes (hole nesters that tend to spawn on the underside of large stones or rocks and wood; Holm et al. 2009) were highly associated with Main in the spring (76.9%), but almost equally with Main and Watts in summer (52.7% and 45.7%, respectively).

2.5 Discussion

2.5.1 Habitat and assemblage structure within and proximate to a stormwater drain

The objective of this study was to determine whether the fish assemblage structure within the Watts Creek watershed differed between the creek and an earthen stormwater drain tributary. Overall, a gradient was evident within the study system whereby the habitat and assemblage structure in Watts and Kizell were distinct, while Main was a mix of the two upstream reaches. Kizell had the lowest habitat complexity, lowest diversity, and most homogenous assemblage, primarily composed of banded killifish, central mudminnow, and brook stickleback. Despite their prevalence throughout the entire system, fewer banded killifish were found in Watts throughout the study period. Watts had the highest diversity and a more heterogeneous assemblage primarily composed of banded killifish, creek chub, longnose dace, bluntnose minnow and white sucker. The diversity in Main was intermediate to that of Kizell and Watts, and the assemblage appears to be composed of aspects of both upstream reach assemblages, making it the most heterogeneous reach. Longitudinal gradients in streams and rivers have been well documented, but tend to focus on species richness, which generally increases from upstream to downstream (Grenouillet et al. 2004). In a study by Ostrand and Wilde (2002) they demonstrated that diversity increased and assemblage composition changed from upstream to downstream in the upper forks of the Brazos River in Texas. While I did not see the same pattern in diversity, as Watts was more diverse than Main, the changes observed in the assemblage from Kizell and Watts into Main appear to be additive (Matthews 1986). However, Ostrand and Wilde (2002) also demonstrated that environmental variables could have more influence on assemblage structure rather than longitudinal gradients. Likewise, in this study there were similar patterns between assemblage composition and habitat complexity among the reaches suggesting that environmental variables could be strongly influencing assemblage structure in the Watts Creek system.

While the species composition of Kizell and Watts differed significantly in this system, Stammler and colleagues (2008), who looked at the fish assemblages in agricultural drains, suggested that at a regional scale there was no difference between the assemblages of drains and streams. Based on their results one might assume that the assemblage would be similar between Watts and Kizell, especially considering Watts and Kizell are physically connected, creating the potential for movement between these systems (explored in Chapter 3). Instead, I observed that some fishes are clearly partitioning into separate reaches. It should however be noted that some of the drain and reference pairs did not cluster together within the biplots (Fig. 2) from Stammler et al. (2008), suggesting that assemblage similarities or differences between drains and streams may be related to catchment variation (Stephenson & Morin 2009) and is likely to differ among watersheds. Nonetheless both the results of my study and Stammler and colleagues (2008) provide compelling evidence that surface drains, whether agricultural or municipal, are capable of providing habitat to support fish communities.

2.5.2 Seasonal variation in assemblage structure

Differences in the structure of the assemblage were consistent among summer, winter and spring, with the exception of the Main reach in the spring, which (like Watts) differed significantly from Kizell (Table 2.4). Only during the fall were there no significant differences in species composition among reaches. The consistent structure of the assemblages among most of the seasons could suggest that environmental variability is driving the assemblage pattern in Watts Creek; however, at the reach scale I may have not been able to detect finer scale spatial changes (e.g. shifts from overwintering to spawning habitats within a reach). It is generally thought that stream fish shift habitats between summer and winter as optimal feeding locations in the summer become less important in the winter (Schlosser 1991; Brown et al. 2011). Considering only about 20% of each reach was sampled in this study, it is plausible that some fish may have shifted habitats across the seasons outside of the sampling areas. Other researchers have also suggested that spatial rather than temporal variables are stronger indicators of assemblage structure (Ostrand & Wilde 2002), but this can vary greatly among taxa and regions (Hatzenbeler et al. 2000). Given that I documented relatively similar composition of the dominant species in each reach between the summer, winter, and spring, and that >80% of the cumulative contribution to these patterns are from species that are found in the system year-round (Table 2.4), it is likely that each reach provided sufficient habitat and resources to support these species throughout the year.

The homogeneous pattern (Fig. 2.2b) noted among the reaches in the fall is largely driven by the distribution of banded killifish (Table 2.4). The average abundance of banded killifish was highest during the fall; however, unlike the other seasons more were found in Main than Kizell. Bearing in mind that this study took place during extreme environmental conditions (drought), I surmise that their broad distribution throughout the system was in response to drought conditions and the lack of water in

Kizell in August, 2012. Supra-seasonal droughts such as this tend to be unpredictable and consequently aquatic organisms have a low to moderate resilience to them (Lake 2003). Some banded killifish were trapped in small pools of water during the drought, and the shift in the highest average abundance from Kizell to Main leads me to believe that some fish moved downstream to find refuge (Lake 2003). An increase in the abundance of banded killifish in the lower sites (W1 & W2; Fig. 2.1) of Watts was also observed (data not shown). However, recovery to pre-drought assemblage patterns occurred rapidly as evident by the similar assemblage patterns between the summer and winter (Fig 2.2). It should be acknowledged that sampling would be needed in other years to determine whether the homogenization in the assemblages of these reaches during the fall was an anomaly; this would help support my conclusion that drought was the driver behind reach assemblage homogeneity.

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 when combined with high stream discharges can increase energetic demands (Cunjak 1988). Habitat selection was not explored in this study, so based on previous research I am assuming that during the winter the fish are concentrated in habitats that minimize energy expenditure and the probability of encountering adverse winter conditions (Cunjak 1996). Overwintering areas generally exhibit reduced velocity with suitable in-stream cover and increased habitat volume (Schlosser 1991; Cunjak 1996); however, this is a general criterion applied to stream fish and it is probable that habitat selected during the winter varies among species. For example, although longnose dace and white sucker have been shown to overwinter in deeper habitats with high rubble cover, these habitats tended to exhibit increased velocities (Cunjak & Power 1986). This general criterion provides insight into the use of Watts and Main as overwintering habitat, but it does not explain the occurrence of fish in Kizell, which is dominated by shallow runs with minimal rubble and in-stream structure. It is possible that species-specific adaptations allow banded killifish and central mudminnow to survive winters in unfavourable habitats where oxygen supply may be depleted. Indeed, Klinger et al. (1982) demonstrated that 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. Similarly, central mudminnow have the ability to absorb more oxygen from bubbles trapped under surface ice and are able to gulp water into a gas bladder (Klinger et al. 1982). Regardless of the mechanism, it is clear from these results that these species are able to exploit habitat in surface drains during the winter.

The assemblage structure in spring was similar to that of summer and winter, except that Main was more similar to Watts than to Kizell (Fig. 2.2d). I also found that species-specific spawning substratum preference led to associations with particular reaches. For example, lithophils (white suckers, creek chub, and longnose dace) were highly associated with Watts, but also found in Main, where medium to course sediment and riffles could provide spawning habitat. An increase in the abundance of white suckers during the spring also occurred, particularly in Main (Table 2.4), because a run of adult white suckers migrate into Watts Creek each year to spawn before returning to the Ottawa River. Phytophils (banded killifish and central mudminnow) on the other hand were proportionally less abundant in Watts, which had minimal in-stream vegetation suitable for spawning. This clear segregation of spawning types among reaches in this study further reinforces my conclusion that, regardless of stream status (drain or creek), habitat within the system is the main determinant of species assemblage.

2.5.3 Management implications

This is the first study that I know of that has specifically explored the fish assemblage of a municipal surface stormwater drain as part of an urban cool/warm water stream system. Regardless of whether surface drains support fish communities unique to neighbouring streams, these systems are capable of providing habitat for some fish species. I therefore echo the conclusions of Stammler et al. (2008) that earthen municipal surface drains should be recognized as providing important fish habitat.

With growing evidence that some stream fish can utilize habitat in surface stormwater drains it will be necessary to change how society views these systems in order to balance the development of urban areas with sustaining local biota. Fortunately, many countries are currently shifting their perception of stormwater runoff, transitioning from it being considered a nuisance and risk to human health and infrastructure to encouraging more sustainable management approaches as we better understand the impacts of stormwater drainages on local aquatic systems (see Roy et al. 2008). Unfortunately, policies that regulate urban infrastructure and aquatic biological systems are rather segregated. For example, 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 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, researchers such as Roy and colleagues (2008) are advocating the joint management of urban and natural systems. I agree that viewing urban stormwater drains as separate from streams and rivers would be detrimental to aquatic communities. Although earthen surface drains similar to Kizell Drain are only a fraction of the whole stormwater management system in urban areas my results support the notion that to maintain the biological integrity of aquatic habitats in urban environments, streams and surface drains need to be considered as inter-connected systems.

2.6 Tables

Table 2.1. Summary of the environmental variables for each reach. The values were averaged between the two sampling periods (September and May), except for sinuosity.

Environmental Variable	**Kizell**	**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
Run (%)	98	66	72
Pool (%)	1	10	10
Riffle (%)	1	25	19
Fine sediment (%)	86	61	57
Medium sediment (%)	11	24	29
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

Table 2.2. Life-history characteristics (Coker et al. 2001) and the relative abundances for all 23 fish species found in Watts Creek, Kanata, Ontario. The spawning type 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 Type	Relative Abundance (%)			
					Kizell	Main	Watts	Whole System
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

Scientific Name	Code	Common Name	Thermal Regime	Spawning Type	Relative Abundance (%)			
					Kizell	Main	Watts	Whole System
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.	ETspp	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
Carassius auratus	CAAU	Goldfish	warm	Phytophil	0.00	0.04	0.04	0.03
Pimephales promelas	PIPR	Fathead Minnow	warm	Speleophil	0.00	0.00	0.04	0.015
Notemigonus crysoleucas	NOCR	Golden Shiner	cool	Lithophil	0.00	0.00	0.04	0.015
Notropis hudsonius	NOHU	Spottail Shiner	cold/cool	Psammophil	0.00	0.04	0.00	0.015
Perca flavescens	PEFL	Yellow Perch	cool	Phytolithophil	0.00	0.00	0.04	0.015

Table 2.3. Summary statistics for Kizell, Main, and Watts reaches across seasons. The catch, species richness, and cool water species were averaged by the number of sampling events per season. The Shannon-Wiener index was calculated from three transects per reach for July (summer), October (fall), December (winter), and April (spring).

Reach	Season	Sampling events	Mean total catch	Average species richness	Shannon-Wiener diversity	Average cool water species (%)
Kizell	Summer	3	201	7	1.11	64
Main	Summer	3	331	11	1.81	51
Watts	Summer	3	280	13	1.96	59
Kizell	Fall	2	178	8	1.21	67
Main	Fall	2	301	10	1.37	66
Watts	Fall	2	172	9	1.68	78
Kizell	Winter	1	20	7	1.13	71
Main	Winter	1	17	8	1.37	75
Watts	Winter	1	16	8	1.75	75
Kizell	Spring	2	47	7	1.30	45
Main	Spring	2	46	7	1.77	79
Watts	Spring	2	53	9	1.72	76

Table 2.4. One-way ANOSIM (R values and significance levels) results on the assemblage composition among reaches within each season and the SIMPER results indicating overall between-group dissimilarity along with the proportional contribution of the most influential species up to a cumulative contribution of >80%. Group A and Group B corresponds with the first and second reach, respectively, in the between-group comparisons.

Season / **Between-group comparisons**	**ANOSIM R (significance)** / **Overall between-group dissimilarity (%)**	**Species**	**Average abundance**		**Contribution to dissimilarity (%)**
			Group A	**Group B**	
Summer	R=0.33 (p=0.04)*				
Kizell & Main	63.0	Banded killifish	58.3	25.0	31.6
		Central mudminnow	12.0	32.0	22.1
		Brook stickleback	30.7	19.0	18.2
		Longnose dace	0.0	13.0	12.6
Kizell & Watts	79.8	Banded killifish	58.3	11.4	36.0
		Brook stickleback	30.7	8.2	17.8
		Creek chub	1.0	23.0	14.2
		Longnose dace	0.0	21.2	13.2
Main & Watts	72.0	Banded killifish	25.0	11.4	21.0
		Central mudminnow	32.0	8.2	20.9
		Creek chub	1.5	23.0	14.6
		Longnose dace	13.0	21.2	14.6
		Brook stickleback	19.0	8.2	13.0
Fall	R=0.18 (p=0.11)				
Kizell & Main	42.2	Banded killifish	91.3	133.5	54.9
		Bluntnose minnow	9.0	24.5	14.5

Season **Between-group comparisons**	**ANOSIM R (significance)** **Overall between-group dissimilarity (%)**	**Species**	**Average abundance**		**Contribution to dissimilarity (%)**
			Group A	**Group B**	
		Central mudminnow	16.7	8.5	8.0
		Creek chub	8.7	10.0	7.9
Kizell & Watts	71.5	Banded killifish	91.3	25.8	50.2
		Creek chub	8.7	36.4	19.3
		Central mudminnow	16.7	1.2	9.1
		Bluntnose minnow	9.0	8.6	7.0
Main & Watts	74.8	Banded killifish	133.5	25.8	53.9
		Creek chub	10.0	36.4	16.6
		Bluntnose minnow	24.5	8.6	9.6
		Longnose dace	8.8	13.2	6.5
Winter	R=0.43 (p=0.003)*				
Kizell & Main	43.5	Banded killifish	13.0	10.8	44.3
		Central mudminnow	4.0	0.8	21.4
		White sucker	1.7	1.3	9.8
		Brook stickleback	0.3	1.5	9.0
Kizell & Watts	78.2	Banded killifish	13.0	2.8	37.7
		Creek chub	0.7	6.0	18.2
		Central mudminnow	4.0	0.0	16.4
		White sucker	1.7	1.6	7.8
Main & Watts	70.8	Banded killifish	10.8	2.8	41.2
		Creek chub	0.8	6.0	21.9
		Brook stickleback	1.5	1.8	8.9
		White sucker	1.3	1.6	7.8
		Longnose dace	0.3	2.4	7.3

Season Between-group comparisons	ANOSIM R (significance) Overall between-group dissimilarity (%)	Species	Average abundance		Contribution to dissimilarity (%)
			Group A	Group B	
Spring	R=0.47 (p=0.003)*				
Kizell & Main	70.2	Banded killifish	18.3	3.5	45.5
		White sucker	1.0	3.5	10.9
		Brook stickleback	4.7	1.0	10.5
		Bluntnose minnow	0.3	3.3	10.3
		Creek chub	5.3	1.3	10.1
Kizell & Watts	69.2	Banded killifish	18.3	1.8	40.7
		Creek chub	5.3	10.0	18.4
		White sucker	1.0	4.8	9.9
		Longnose dace	0.0	4.0	9.7
		Brook stickleback	4.7	3.6	7.9
Main & Watts	64.4	Creek chub	1.3	10.0	28.0
		White sucker	3.5	4.8	14.6
		Longnose dace	1.3	4.0	13.0
		Banded killifish	3.5	1.8	12.6
		Brook stickleback	1.0	3.6	12.3

* denotes when the ANOSIM was significantly different ($p<0.05$)

Table 2.5. The total abundance of each spawning type (Balon 1981) and the proportion that were found in Kizell, Main, or Watts during spring and summer.

Reach	Ariadnophils	Lithophils	Phytophils	Speleophils
		Spring		
Kizell	34.2%	7.1%	52.3%	3.9%
Main	17.1%	21.1%	36.6%	76.9%
Watts	48.7%	71.8%	11.1%	19.2%
Total abundance	41	294	199	52
		Summer		
Kizell	38.7%	2.9%	24.4%	1.6%
Main	33.4%	23.5%	46.9%	52.7%
Watts	27.9%	73.6%	28.7%	45.7%
Total abundance	323	898	1823	258

2.7 Figures

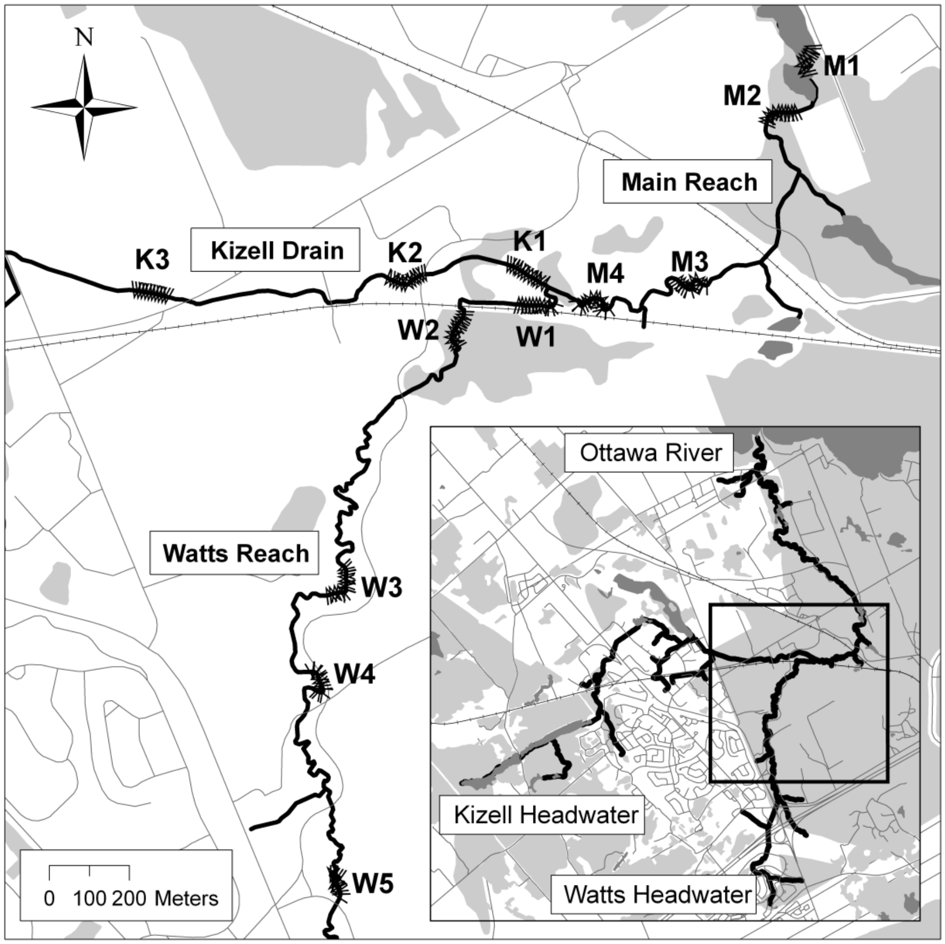

Figure 2.1. Map of the Watts Creek study site and an inset with its position within Watts Creek showing the headwaters and connection to the Ottawa River. The direction of water flow is from west to east for Kizell Municipal Drain, and from south to north for Watts Creek. The sampling sites (black hatches) 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 heavily vegetated areas with the dark grey representing water bodies.

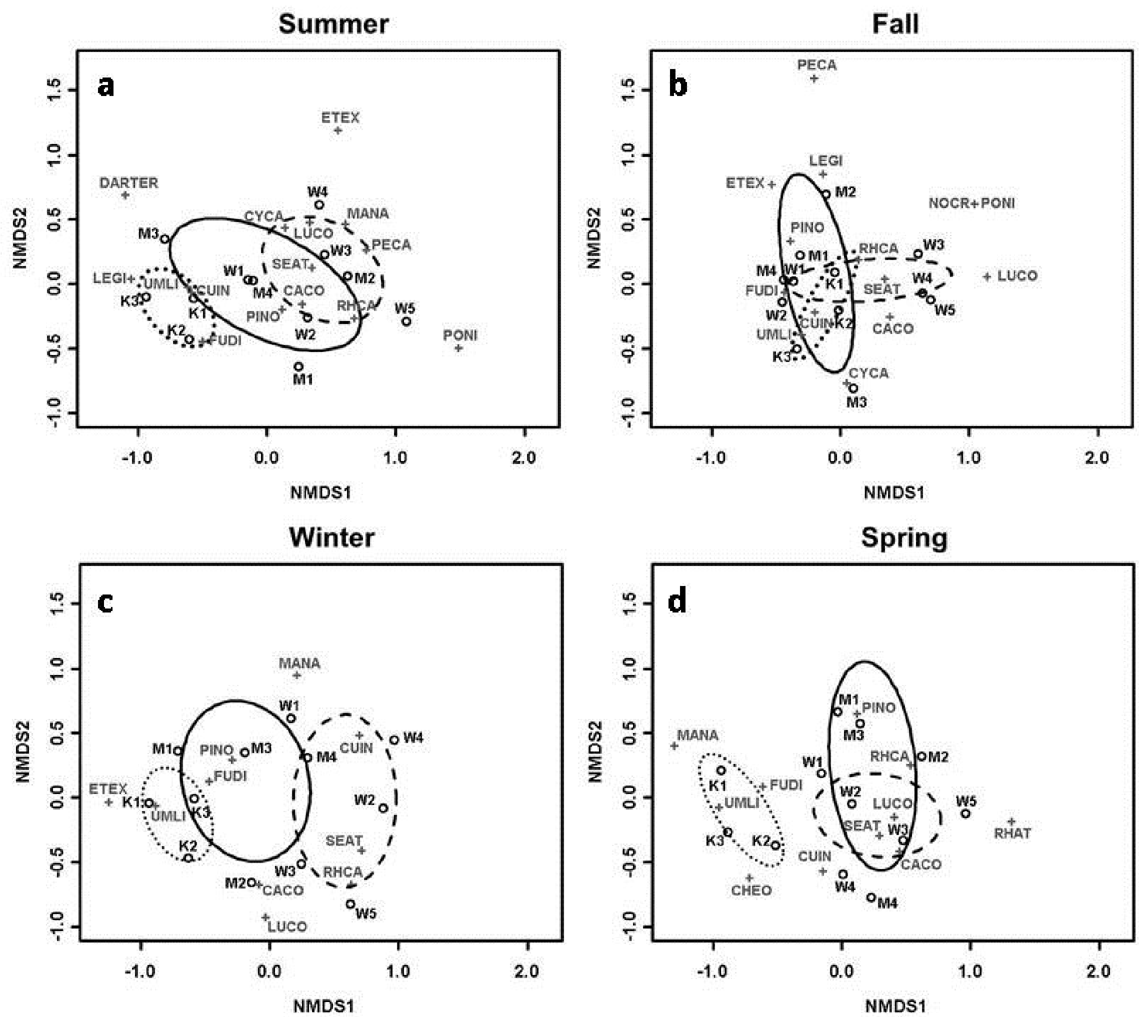

Figure 2.2. Non-metric multidimensional scaling (NMDS) of community composition for each season based on species abundance. Summer (a) is represented by the July sampling event, fall (b) is from October, winter (c) is from December, and spring (d) is from April. Ordination stress values for summer, fall, winter, and spring were 0.10, 0.13, 0.11, and 0.14, respectively. Species are represented as the (+) symbols and transects as the open circles (o). The ellipses are the 95% confidence limits around the centroid (weighted average) of each reach. Kizell, Main, and Watts are each represented by the dotted, solid, and dashed ellipse, respectively.

Chapter 3. Seasonal Movements of Small-bodied Fish in a North Temperate Urban Watershed Demonstrate Connectivity between a Stream and Stormwater Drain

3.1 Abstract

Fish move throughout aquatic ecosystems for a variety of reasons, including for foraging, spawning, to escape predators, or even to find refuge from environmental disturbances. A better understanding of stream fish movement is vital to understanding their potential responses to human-induced environmental change. The goal of this study was to evaluate the interconnectedness of a natural urban stream (Watts Creek) and an adjoining surface stormwater drain (Kizell Drain) located in Ottawa, Canada, from the perspective of fish movements over a one-year period. By using a stationary passive integrated transponder (PIT) array, I quantified and compared the direction of movements among Watts Creek, Kizell Drain, and the area downstream of their confluence (herein termed Main). I also determined the residency time (percentage of days spent) within each of these reaches generated from two datasets: 1) from individuals detected on the PIT array, and 2) data combined from the array and data collected from individuals that were recaptured or detected using a portable PIT backpack reader. I limited quantitative analyses to creek chub, central mudminnow, longnose dace, and white sucker, for which there were sufficient data. The directional movements among reaches generally did not vary across species and seasons, with a few notable exceptions. In the winter, creek chub moved more often between Kizell and Watts, while in the summer, longnose dace moved more often from Main into Watts. Diel patterns of movement also varied among species. Creek chub and white sucker were detected at all hours of the day, although white

suckers moved slightly more at night. Central mudminnow and longnose dace movements were almost exclusively nocturnal (occurred from 6pm - 6am). Residency time for creek chub was only significantly greater in Watts using the combined dataset. Conversely, white sucker resided in Kizell more often than Watts when only considering the PIT array dataset. Both datasets revealed that central mudminnow resided more in Kizell and Main, while longnose dace were found in Watts significantly more often. I conclude that there is a high degree of connectivity between Watts Creek and Kizell Drain and fish do move quite freely through the system, though populations seemed to contain both mobile and comparatively sedentary individuals. In summary, this study provides strong evidence for considering earthen stormwater drains as a functional component of urban watersheds and for adjusting their management accordingly to reflect their value as fish habitat.

3.2 Introduction

Streams and rivers are the hydrological 'highways' that connect various landscape elements (including riparian areas) and serve as corridors in urban areas (Walsh et al. 2005). Indeed, stream corridors provide opportunities for fish and wildlife to move about otherwise fragmented habitats (Puth & Wilson 2001). However, intensification and expansion of urban centers means that it is important to understand the influence of land use change on individuals, populations and ecosystems. Impervious surface cover and stormwater management systems designed to efficiently drain water runoff out of cities is an inevitable part of the land use changes in urban landscapes (Paul & Meyer 2001). Studies on the impacts of stormwater management on streams have primarily examined stream morphological changes, transportation of contamination and sediment, and changes in macroinvertebrate or fish community structure (Berkman & Rabeni 1987; Hatt et al. 2004; Sponseller et al. 2001). Less is known about fish dispersal and movement within streams and associated stormwater drainage systems in urban areas. From the perspective of a fish, many questions exist regarding the extent to which they move among reaches exposed to different stressors within urban watersheds.

The study of fish movement has been essential to ecologists for years because it describes the mechanism fish use to respond to various changes in environmental conditions, such as foraging and predator avoidance (Rodriguez 2002). Understanding fish movement in streams enables researchers to determine the habitat needs of various species, define spatial boundaries of populations, identify migration patterns and corridors, and characterize the effects of physical barriers and environmental disturbance (Freeman 1995; Lonzarich et al. 2000; Lucas & Baras 2000; Belanger & Rodriguez 2002; Stuart & Jones 2006). Stream fish in both warmwater and coldwater systems have been studied for decades, initially with simple mark-recapture approaches (e.g., Funk 1957). However, with the advent of telemetry it became apparent that mark-recapture studies were biased against the detection of movement (Gowan et al. 1994). Although there have been many telemetry studies on stream fish, nearly all of them have focused on salmonids given that they are rather large by stream-fish standards and socio-economically important as gamefish. Innovations in passive integrated transponder (PIT) technology provide new opportunities for the study of entire stream fish communities given that tags are inexpensive, small, and can last for years (Gibbons & Andrews 2004; Cucherousset et al. 2010). PIT tagged fish can be located manually with mobile tracking systems

(Zydlewski et al. 2001; Cucherousset et al. 2010), monitored as part of traditional mark-recapture studies (Dieterman & Hoxmeier 2011) or tracked using automated stationary PIT detection arrays (Teixeira & Cortes 2007). PIT systems can also be used in winter including under ice (or through ice) which is particularly relevant to streams in north temperate regions (Roussel et al. 2004). Despite its limitations, such as tag orientation sensitivity (orthogonal vs. parallel), reasonably small detections ranges, and in the case of stationary detection arrays, its reliance on the tagged animal moving through a known corridor (Zydlewski et al. 2001; Cucherousset et al. 2010), PIT telemetry serves as a very useful tool for monitoring fish movements. In addition, it provides unique opportunities for studying small-bodied stream fish species on a seasonal basis.

As noted above, the impacts of human development on fish have been well documented (Klein 1979; Schlosser 1991; Roy et al. 2008), but most of this work has been focused on changes in stream fish assemblages (Wichert & Rapport 1998) rather than fish movements. Researchers that have studied movement have focused on migration barriers (culverts) and changes in flow and temperature (Scott et al. 1986; Marchetti & Moyle 2001; Norman et al. 2009). I am unaware of any studies that have explored the connectivity between earthen surface drains and streams from the perspective of fish movements. Earthen surface drains (also known as ditches or swales) are only a portion of the various types of stormwater drains (man-made vs earthen and subsurface vs surface; Djokic & Maidment 1991) which are open to the surrounding environment, flow above ground, and may be physically connected to streams. Despite physical connectivity in most jurisdictions drains and streams are treated quite differently. Drains are subject to cleaning (i.e., use of heavy equipment to maintain an open channel) as needed (or sometimes at regular intervals – e.g., every 5 years) and are typically not regarded as fish habitat in a regulatory context. In general, most storm drains are comparatively simple channels, often with less horizontal and vertical sinuosity relative to natural streams. However, it is important to recognize that historically many surface drains were natural stream environments prior to their channelization and management for stormwater conveyance (Kaushal & Belt 2012). Knowledge of the extent to which fish use earthen surface drains and the level of connectivity between streams and drains on a seasonal basis would be useful in identifying the ecological value of these systems. Furthermore, most studies are conducted during the summer resulting in major gaps in our understanding of how fish in north temperate urban streams behave or are distributed throughout the entire year including the winter months (Cunjak 1996; Brown et al. 2011).

Using the Watts Creek watershed in Kanata, Ontario, Canada I assessed the movements of small stream fish between a protected urban stream (Watts Creek) and an adjoining earthen municipal surface stormwater drain (Kizell Drain). Watts Creek and Kizell Drain are distinct in their substrate composition, legislative management, and origin. Relative to Watts Creek, the aquatic habitat in Kizell Drain is more disturbed from being straightened and homogenized through channelization (see Chapter 2). This field study focused on the confluence between Watts Creek and Kizell Drain, which had three reaches that branched out: Watts (upstream Watts Creek), Main (downstream Watts Creek), and Kizell. My objective was to compare the extent of fish movement among these reaches using a stationary PIT detection array over a one year period (including under the ice). I predicted that the degradation of the habitat in Kizell would result in proportionally less movement into the drain. Given the availability of 24-hr movement data from the stationary PIT array, I explored diel activity patterns of common species to provide a more detailed understanding of their behaviour. Finally, I compared the residency time (in days) among the reaches and did so using data generated in two ways: from the fixed detection array, and using the array supplemented with additional fish positions obtained via mark-recapture of PIT tagged fish and infrequent manual tracking outside of the area where the array was installed. My quantitative analyses were focused on four of the most abundant species: creek chub (Semotilus atromaculatus), central mudminnow (Umbra limi), longnose dace (Rhinichthys cataractae), and white sucker (Catostomus commersonii). My findings are discussed in the context of urban stream ecology and the interconnectedness of streams and storm surface drains.

3.3 Methods

3.3.1 Study area

Watts Creek (45°20’42”N, 75◦52’19”W) is a tributary of the Ottawa River and located in Kanata, Ontario, Canada, the largest suburb in the Ottawa region. Precambrian and Paleozoic bedrock overlaid with a layer of silts and clays makes up the soil composition of the Watts Creek watershed (~2500 ha), and about 47% of the land used is agricultural, 35% is developed, and 18% is undeveloped (Stantec Consulting Ltd 2011). The creek originates in the Katimavik-Hazeldean community of Kanata and flows through a residential area, including under a major highway before entering the Ottawa Greenbelt. The Greenbelt is protected and managed by the National Capital Commission (NCC) and contains rural and agricultural lands. Watts Creek is groundwater fed with stormwater inlets from surrounding areas including Kizell Drain, an earthen municipal surface drain originating at the Beaver Pond stormwater management pond (near Walden Dr., Kanata, ON). The confluence between Watts Creek and Kizell Drain is the focal point of this study in which there are three reaches branching out (Fig. 3.1). These reaches are herein referred to as Watts (~2.9 km of the creek upstream from the confluence), Kizell (~1.5 km of Kizell Drain upstream from the confluence), and Main (~1.7 km of the creek downstream from the confluence).

3.3.2 Fish sampling and tag implantation

Fish were sampled using single-pass backpack electrofishing (Model 12, Smith-Root, Vancouver, WA, USA) from 26 March 2012 to 8 November 2012, and again from 22 April 2013 to 30 May 2013 throughout Watts Creek and Kizzel drain. Sampling frequency and location varied throughout the study. Most of the sampling and tagging occurred within twelve 100 m transects approximately once per month throughout the study period (for a total of 9 occasions). Additional sampling between transects occurred within the first 5 months to increase the number of tagged fish. This included 8 occasions where specific sections were targeted and 1 occasion where I sampled the whole system to include areas that were infrequently sampled (each step increase in the number of tagged fish in Fig. 3.2 represents an occasion of active sampling).

Each fish was identified to species, the total length (TL) was measured, and they were tagged with a uniquely coded 12 mm (12 x 2.15 mm) or 23 mm (23 x 3.65 mm) HDX PIT tag (Oregon RFID, Portland, OR, USA). The size of tag selected depended on the species and total length of the fish. In general fish of approximately 70 mm – 130 mm

(TL) were tagged with 12 mm tags, while fish of approximately 120 mm and larger were tagged with 23 mm tags. The minimum size of fish tagged varied depending on species. For example, central mudminnows were generally large enough to receive a tag at 70 mm, while some longnose dace below 75 mm were considered too small (varied by individuals). Also, I observed that some species experienced higher tagging mortalities (i.e. bluntnose minnows and logperch); therefore I did not continue tagging these species throughout the study. Using a 12-gauge needle or scalpel, a small puncture (< 3 mm) for 12 mm tags or an incision (< 5 mm) for 23 mm tags was made to the side of the ventral midline anterior to the pelvic girdle and tags inserted into the coelomic cavity. Air exposure and handling time was minimized (< 1 min) and tagged fish were kept in a recovery bucket for a short period (< 30 min) before being returned to the creek. No anesthetic was needed given that most fish were still in a sedated state as a result of capture by electrofishing and that anesthesia was not needed to restrain fish for this simple procedure. A GPS coordinate was used to indicate the location fish were tagged and released.

3.3.3 Water temperature monitoring

To assess water temperature among the reaches and whether fish detection on the PIT array paralleled changes in temperature, loggers were installed throughout the study site. Water temperature was recorded every 255 minutes (5 - 6 times per day) from 7 May 2012 to 12 Oct 2012 and from 24 Oct 2012 to 16 Oct 2013 in 2 locations within Kizell, Main, and Watts (ibutton DS1921Z; Maxim Integrated, San Jose, CA, USA; Fig. 3.1).

3.3.4 Tracking and observations

To monitor fish movements between Watts Creek and Kizell Drain, three pass over antennas were installed in Main (1.3 m x 3.25 m; length x width), Watts (0.84 m x 2.1 m), and Kizell (1 m x 2.5 m) approximately 5 to 7 m from the confluence center (Fig. 3.1). The width of the antennas corresponded to the width of the stream where they were located. The antennas were secured to the stream floor with large, heavy rocks that were placed between two sheets of diamond mesh polyethylene fencing material with 12 awg THHN electrical wire tied along the perimeter. The antennas were tuned manually with remote tuner boxes and connected to a MultiAntenna HDX Reader with Twinax cable (equipment obtained from Oregon RFID, Portland, OR, USA).

For each sampling occasion I scanned all fish captured that were a minimum of 70 mm (TL) for the presence of a tag. Any fish that had already been tagged was considered a recapture. A portable HDX Backpack PIT Reader with attached antenna pole (Oregon RFID, Portland, OR, USA) was infrequently (3 occasions between May & July 2012) used to scan the whole system within my study area for tagged fish. The operator swept the antenna from bank to bank across the surface of the water while moving downstream. In an attempt to improve detection potential in deep pools the antenna was submerged below the surface to a maximum of 30 cm. Scanning efficiency within this system was low because the streambed was predominantly composed of fine sediments (clay and silt) which slowed the pace in which the operator walked. Fish were frequently observed swimming faster than the operator could move despite previous work demonstrating the efficiency of tracking in a downstream direction (Cucherousset et al. 2010); therefore scanning was discontinued. Most of the previous work using this method focused on salmonids, which may have a tendency to exploit structural complexity and hide, enabling PIT detection, rather than attempting to escape.

3.3.5 Data analysis

I defined movement between two reaches as the detection of a tag from one antenna to another with a minimum of 30 seconds between detections. Six possible directions of movement between reaches were identified as follows: Kizell to Main, Kizell to Watts, Main to Kizell, Main to Watts, Watts to Kizell, and Watts to Main. Specific data analyses on the extent and timing of detections as well as residency were conducted on the four species with suitable tagging and detection sample sizes ($n > 50$). The extent of movement was calculated as the movement detected for each of the six possible directions as a percentage of the total movement observed within a given time period. For example, during summer 58 movements for creek chub were observed among the reaches, and of that 13 occurred in the direction of Main to Kizell. Therefore, 22.4% of the movement during summer was in the Main to Kizell direction for creek chub. Chi-square goodness-of-fit tests were used to compare the observed distribution of movement among reaches with what would be expected if all movement among the reaches were equally distributed. Diel patterns of tagged fish detections on the PIT array were determined by plotting the number of records by the hour of day for the entire study period. Residency time was determined by counting the number of days an individual fish spent within each reach relative to the total number of days the fish was available for detection after being tagged. The location of a fish was determined by tracking its movement through the PIT antenna or using other locations identified with mark-recapture or a portable backpack PIT reader. These calculations were converted into percentages because the total number of days each fish was available for detected varied across the study period. Two datasets were also generated and determined by: 1) tracking the movement among the three antennas on the stationary PIT array only (herein referred to as the PIT array dataset), and 2) using data from the PIT array in combination with data from recaptured and portably detected fish (herein referred to as the combined dataset). It is important to note that the sample sizes between the datasets used differed because fish that were recaptured or detected with the portable antenna were not necessarily detected on the stationary PIT array. A Kruskal-Wallis analysis of variance was used to test if residency time among the reaches differed significantly, and Tukey-Kramer HSD comparisons determined specific group differences following a significant result. All statistical analyses were deemed significant at $p < 0.05$ and performed using JMP statistical software (version 7.0.1; SAS Institute Inc., NC, USA).

3.4 Results

Sixteen species comprising 1,283 fish were tagged, of which 430 fish (33.5%) across 10 species were either detected or recaptured. A total of 340 fish were detected on the stationary PIT array alone (26.5%); however, the average number of fish detected per day was 0.41% (ranged from 0 – 3.76%; Fig. 3.2). The majority of the fish tagged (93%) and detected (94%; Table. 3.1) were made up of the four common species (creek chub, white sucker, longnose dace, and central mudminnow). Creek chub and longnose dace were primarily caught and tagged in Watts (80.2% and 74.7%, respectively). Central mudminnow was caught and tagged in Main mostly (58.5%) and similar numbers of white sucker were tagged in Main and Watts (48.4% and 44.0%, respectively). The total length of fish tagged ranged from 70 – 580 mm, and the minimum length of fish detected corresponded with the minimum length tagged for all the species detected except common shiner (Table 3.1). Although we did not quantify array performance, the detection efficiency was presumably lowest during times of high flows or on stormy days (Aymes & Rives 2009); however I am assuming that because the antennas were close to each other that the efficiency among them was relatively similar.

Overall the difference in mean daily temperature between Kizell, Main, and Watts was very small. Although some variation in temperature among the reaches occurred each day (Fig. 3.3) the overall average temperatures over the course of the entire study for Kizell, Main, and Watts were 11.8ºC, 11.2 ºC, and 11.1 ºC, respectively. In general there did not appear to be a pattern between water temperature and the number of fish detected (visually inspected).

The directional movements of creek chub, central mudminnow, longnose dace, and white sucker did not vary significantly among reaches or seasons ($p>0.05$; Fig. 3.4), except for creek chub and longnose dace during the winter and summer, respectively. In winter, creek chub moved significantly more between Kizell and Watts and less between Main and Watts ($\chi^2 = 23.91$, df = 2, $p<0.001$). In contrast, during the summer longnose dace moved significantly more between Main and Watts and there were no movements from Watts into Kizell and from Kizell into Main ($\chi^2 = 82.29$, df = 2, $p<0.001$). Nevertheless, creek chub, central mudminnow, and white sucker appeared to move more often into Kizell, while longnose dace movements were most often toward Watts Creek (Fig. 3.4).

In terms of diel movement, creek chub and white suckers were detected at every hour of the day (Fig. 3.5) and while there was no discernable diel pattern for creek chub, there were slightly fewer records for white suckers mid-day. In contrast, almost all records for central mudminnow and longnose dace occurred between 6pm and 6am (i.e., nocturnal activity).

A Kruskal-Wallis test on the residency of creek chub using the PIT array dataset suggested that there was a significant difference among reaches ($H = 7.39$, $df = 2$, $p = 0.02$); however, post hoc comparisons (Tukey HSD) revealed no significant differences among the reaches. Still Kizell did have a higher mean value (38%) and Main had the lowest (26.5%; Fig. 3.6). By including data from recaptured and portably detected fish there was a clearer difference among the reaches with creek chub residing in Watts significantly longer than Kizell and Main (Kruskal-Wallis, $H = 62.01$, $df = 2$, $p < 0.0001$). Both central mudminnow and white sucker spent significantly more days in Kizell than Watts (Kruskal-Wallis, $H = 12.22$, $df = 2$, $p = 0.002$; $H = 12.33$, $df = 2$, $p = 0.002$) when considering the PIT array dataset. Using the combined dataset central mudminnow still spent significantly more time in both Kizell and Main than in Watts (Kruskal-Wallis, $H = 15.25$, $df = 2$, $p < 0.001$). Using the combined dataset, there was no difference in residency among reaches for white suckers (Kruskal-Wallis, $H = 3.40$, $df = 2$, $p = 0.18$). Finally, both datasets revealed that longnose dace resided in Watts significantly longer than both Kizell and Main (PIT array: $H = 15.24$, $df = 2$, $p < 0.001$; combined: $H = 114.66$, $df = 2$, $p < 0.0001$).

3.5 Discussion

3.5.1 Movement of stream fish between Watts Creek and Kizell Stormwater Drain

To my knowledge, this is the first study to evaluate the movements of non-salmonid stream fish between an earthen surface stormwater drain and urban stream including over winter months in a north temperate urban system. Assuming the habitat in Kizell Drain was more degraded from persistent anthropogenic activity (e.g. channelization) relative to Watts Creek, I had predicted fish would move proportionally less into Kizell. Contrary to this prediction, throughout most of the year the movements among Kizell, Main, and Watts did not differ significantly for creek chub, central mudminnow, and white sucker (Fig. 3.4). Longnose dace was the only species that fit in with my prediction relative to the other species; however, the sample sizes for dace were small. Most of the movements of longnose dace were into Watts, but not exclusively. There were also, to some degree, dace that moved into Kizell or Main. The overall movement patterns for all four species (though to a lesser degree for longnose dace) strongly suggest a high level of connectivity between Kizell Drain and Watts Creek, because fish moved quite freely between the two systems. Although, it should be noted that Kizell Drain has not undergone cleaning (with heavy equipment) for many years (10+ years) prior to this study and so it is possible that these results could differ shortly after such an event. Future research is necessary to evaluate how fish may respond to cleaning activity within stormwater drains. Also the use of one PIT antenna at the entrance of each reach only allowed for the evaluation of whether a fish entered a reach, not how far they travelled into each reach. In the future, use of a paired antenna system which includes an additional antenna approximately 10 - 20 m away from the reach entrance would allow us to evaluate how far into a reach fish are moving. Though, by determining residency time it is evident that fish were moving into and remaining within Kizell for long periods of time even with a single antenna at the entrance of each reach (Fig. 3.6).

The proportion of movements among the reaches did not differ significantly across seasons for central mudminnow and white sucker, while there was a difference in the movements of longnose dace and creek chub in the summer and winter, respectively. There were, however, slight variations in the directionality of movements across seasons for all four species. More upstream movements from Main into Kizell (creek chub, central mudminnow, and white sucker) or Watts (longnose dace) were detected during the summer, while in the fall and spring there was a less discernable pattern across species. Winter, on the other hand, differed significantly for creek chub, which had more movement between Kizell and Watts, particularly into Kizell. Central mudminnow and white sucker also appear to have moved into Kizell more during the winter. This is particularly interesting because overwintering areas are generally understood to have suitable in-stream cover and increased habitat volume with reduced velocity (Schlosser 1991; Cunjak 1996); however, Kizell was predominantly shallow with barely any in-stream cover. Although most winter research has focused on salmonids, Moshenko and Gee (1973) also described the overwintering habitat for creek chub as deep (>50 cm) sheltered pools. There is one pool in Kizell that fits this description (>50 cm deep with some in-stream woody shelter from a fallen tree found in the K1 transect; Fig. 3.1), and so it is possible that the fish moving into Kizell could have exploited the few areas that were suitable for overwintering. The quantity and quality of overwintering habitats in drains would need to be further investigated. Interestingly, Brown et al. (2001) demonstrated that white sucker and common carp would move long distances in response to winter flooding and ice break-up. Whether fish were moving into Kizell to utilize habitat or in response to changes in the condition of habitats in Main or Watts is unknown; however, this study still demonstrates that earthen surface drains are capable of supporting different fish species throughout the year, including during winter.

In general the diel activity of stream fish is highly variable and complex both inter-specifically and intra-specifically (Reebs 2002). For example, lake chub (Couesius plumbeus) have been shown to exhibit clear circadian rhythms under constant conditions unlike other cyprinids (Reebs 2002). On the other hand, various cyprinids (including lake chub) display plasticity in their diel activity in response to external variables, such as seasons, prey availability, or predator avoidance (Reebs 2002). In this study, the differing diel activity among creek chub, central mudminnow, longnose dace, and white sucker further support the importance of understanding species-specific patterns. Creek chub exhibited no clear diel pattern, while longnose dace were more active at night across the study period (Fig. 3.5). Although central mudminnow are from a different family than longnose dace, these species appeared to have similar diel activity. White sucker, in contrast, were only slightly more active at night but detected at any time of the day. Reebs et al. (1995) attained different results when they looked at the diel activity of juvenile white suckers and blacknose dace (Rhinichthys atratulus) using baited and unbaited nets. They found that both species were more active during the day than at night. Steffensen et al. (2013) found that creek chub and white sucker move actively through a nature-like fishway almost exclusively overnight. Among the results of Reebs et al. (1995), Steffensen et al. (2013), and this study three different diel behavioural patterns have been reported for white suckers. Therefore diel patterns not only vary across species but also within a species. This emphasizes the need to monitor movement over 24-hours so the different movement behaviours can be accounted for within a study. By monitoring fish movement in real-time using PIT telemetry I was able to acquire a substantial amount of data on certain species that may have been missed if monitoring only occurred during the daytime or if we relied solely on mark recapture.

3.5.2 Residency patterns of stream fish

It is increasingly apparent that stream fish populations are composed of sedentary and mobile individuals (Gatz & Adams 1994; Knaepkens et al. 2004; Hilderbrand, 2011) or even individuals that may switch between these behavioural modes (Harcup et al. 1984; Knaepkens et al. 2004). This could partially explain the movement dynamics and spatial ecology of the fish populations within the Watts Creek watershed. In the case of creek chub, a small proportion of the fish tagged were detected moving through the array (21%), with more movement toward Kizell (Fig. 3.4). In addition, by referring only the PIT array dataset, these individuals did not display a particular reach preference in terms of residency (Fig. 3.6), despite that 80% of creek chub were tagged in Watts (Table 3.1). It is possible that the behaviours of the primarily mobile individuals within the creek chub population are being over-represented in the PIT array dataset. When I supplemented this information with data from recaptured/portably detected individuals there was a notable shift in in the residency times (Fig. 3.6). It became apparent that creek chub resided within Watts significantly more than Kizell and Main, because some of the more sedentary individuals (fish tagged in Watts but never passed the array) were represented within this dataset. I saw a similar scenario with white suckers where the PIT array data suggested fish resided in Kizell significantly longer than Watts; however, by taking into account recaptured individuals an increase in the residency times for Main and Watts was evident, which better reflects the proportion of fish captured and tagged in those reaches. Conversely, almost half (47%) of the central mudminnow tagged were represented on the PIT array and slightly more than half (56%) when recaptured/portably detected fish were considered, meaning a larger portion of the population sampled is represented in this study. Although most of the mudminnow tagged were in Main, both Kizell and Main had higher residency levels (and more movement between them) relative to Watts. This could suggest that a larger proportion of this population may be mobile compared to creek chub or white sucker. In contrast, a very small proportion of the longnose dace tagged were detected on the PIT array (7%), and in this case more individuals were recaptured or portably detected (17%). This leads me to believe that longnose dace may be more sedentary, and have a preference for Watts given that more were tagged and the residency time was significantly greater in Watts. This highlights the usefulness of combining methods, because the stationary array in this study could have placed a stronger emphasis on mobile individuals within some species, while mark-recapture studies have a tendency to bias toward sedentary individuals (Gowan et al. 1994). By combining both of these approaches I was able to attain a better representation of the entire population compared to using one of these methods.

3.5.3 Tagging implications

To my knowledge, this may be the first study that used PIT tags on several stream fish species including longnose dace, central mudminnow, banded killifish, bluntnose minnow, and logperch. I found that of the species tagged, bluntnose minnow and logperch appeared to be intolerant to the procedure. I observed numerous mortalities < 30 minutes after tagging and did not recapture or detect fish on the array for both species. In addition, this study is the first I know of to tag common shiner as small as 73 mm; however, the smallest size detected was 102 mm (Table 3.1). Therefore, for common shiner it may be advisable to tag fish no smaller than 100 mm until tagging effects for this species have been further investigated. As PIT technology improves and size of tags shrink, studies on smaller-bodied stream fish has been increasing. It is important to define species-specific tagging thresholds (e.g. minimum size for receiving tags), especially considering that the response and tolerance presumably varies greatly among species (Stakenas et al. 2009; Burdick 2011). Although I was not directly testing the effects of tagging, and bearing in mind some of the smallest fish tagged were detected on the PIT array (Table 3.1), I have shown that PIT telemetry studies can be useful and applied to adult and juvenile stream fish, some (i.e. banded killifish, creek chub, central mudminnow, longnose dace, and white sucker) as small as 70 - 72 mm (TL).

3.5.4 Implications for management

Movement is costly to fish due to energy expenditure (Boisclair & Tang 1993) and the risk of predation (Belica & Rahel 2008). When fish do move, it is done with the notion that it will improve their fitness. A part of this is avoiding habitats that do not provide any energetic gains (Facey & Grossman 1992). Therefore, based on the movements and residency time observed in this study, fish are moving into and gaining something from the habitat in Kizell, whether it was for foraging, overwintering, or spawning habitats. So, in urban environments where habitat degradation and loss is quite prominent, earthen stormwater drains could provide additional habitat that fish can exploit and eventually colonize. Stammler et al. (2008) and my own research (Chapter 2) has demonstrated that fish do colonize surface drains which could lead to similar or distinct assemblages from that found in streams. Though, it is also important to remember that most these types of drains were once natural waterways prior to their conversion into drainage infrastructure. In terms of management, this mean that surface drains will need to be considered a part of and connected to the aquatic ecosystem with which they drain into. As well, more research is providing support that the management of targeted home ranges for stream fish should be increased to account for complex movement behaviours (Smithson & Johnston 1999), and connected drains will need to be included to maintain local biota within a dynamic urban landscape. In summary, this study provides strong evidence that the management of urban aquatic ecosystems needs to consider earthen surface stormwater drains as a functional component of urban watersheds to reflect their value as fish habitat.

3.6 Tables

Table 3.1. Species tagged along with the mean (± standard deviation) and minimum total lengths (TL) tagged and detected, tagging location (reach), and the number of tags that were detected or recaptured over the course of the entire study.

Species	TL (mm)	Minimum TL (mm) tagged	Minimum TL (mm) detected	Number tagged in Kizell	Number tagged in Main	Number tagged in Watts	Tags detected (%)
Brown bullhead Ameiurus nebulosus	89	-	-	0	1	0	0
Black crappie Pomoxis nigromaculatus	80 ± 4	76	-	0	3	1	0
Banded killifish Fundulus diaphanus	76 ± 4	71	73	1	1	2	2 (50)
Bluntnose minnow Pimephales notatus	82 ± 5	72	-	0	9	1	0
Common carp Cyprinus carpio	141 ± 50	82	82	0	5	2	3 (43)
Creek chub Semotilus atromaculatus	118 ± 28	71	72	34	44	315	120 (31)
Central mudminnow Umbra limi	84 ± 9	70	70	39	120	46	115 (56)
Common shiner Luxilus cornutus	105 ± 18	73	102	2	9	19	9 (30)
Longnose dace Rhinichthys cataractae	90 ± 9	72	72	1	58	174	55 (24)
Logperch Percina caprodes	96 ± 9	86	-	0	6	0	0
Northern pearl dace Margariscus nachtriebi	105 ± 19	84	84	0	0	6	6 (100)

Species	TL (mm)	Minimum TL (mm) tagged	Minimum TL (mm) detected	Number tagged in Kizell	Number tagged in Main	Number tagged in Watts	Tags detected (%)
Northern redbelly dace Chrosomus eos	101	-	-	1	0	0	0
Pumpkinseed Lepomis gibbosus	95 ± 9	83	84	2	7	2	4 (36)
Spottail shiner Notropis hudsonius	78	-	-	0	1	0	0
White sucker Catostomus commersonii	182 ± 113	71	71	28	178	162	114 (31)
Yellow perch Perca flavescens	109 ± 19	90	90	0	3	0	2 (67)

3.7 Figures

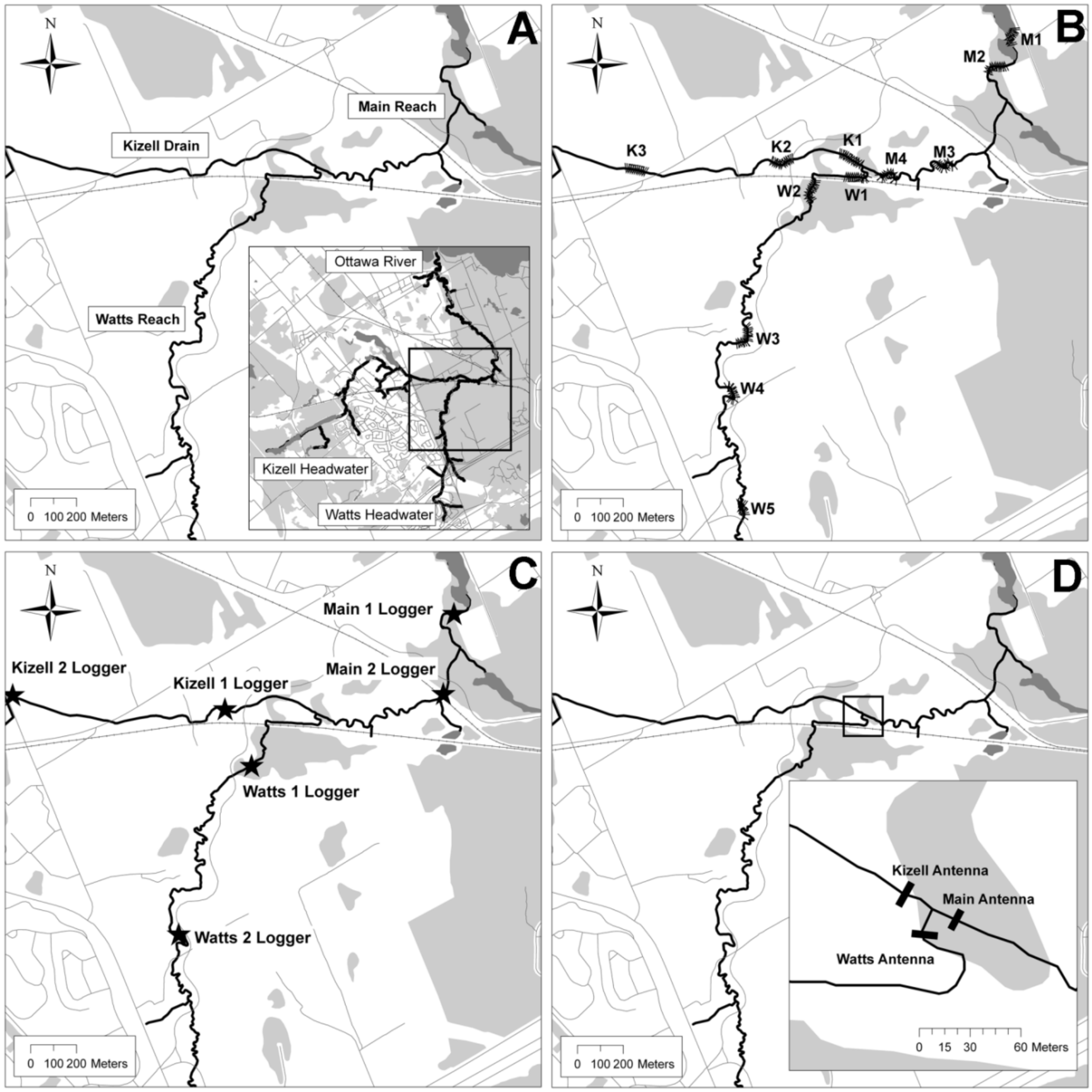

Figure 3.1. Map of the Watts Creek study site showing the A) location of site within the watershed (inset) and location of the reaches, B) transects where regular sampling occurred, C) locations of the temperature loggers, and D) the PIT array set-up around the confluence of the stream and stormwater drain. The direction of water flow is from the east to west for Kizell and from the south to north for Watts and Main. The sampling sites and temperature loggers in each reach are 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 hatched lines are train tracks. The lighter grey shaded area represents heavily vegetated areas with the dark grey representing water bodies.

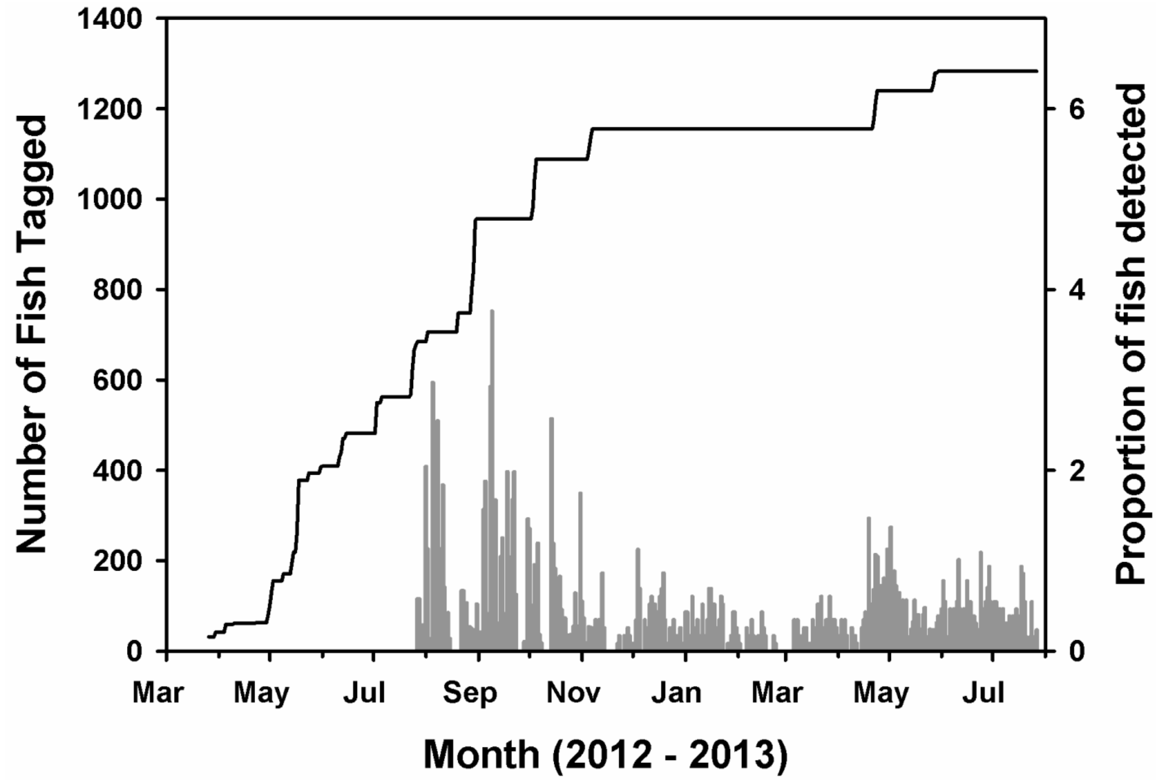

Figure 3.2. The number of fish tagged (black line, left axis) and the proportion of tagged fish detected (dark grey bars, right axis) each day over the course of the study. Tagging initiated on 26 March 2012, the PIT array was active on 27 July 2012, and the study ended on 27 July 2013. Every step increase in the number of fish tagged represents an event when active sampling and tagging occurred (18 in total). Sampling frequency was higher in the beginning of the study in order to increase sample sizes.

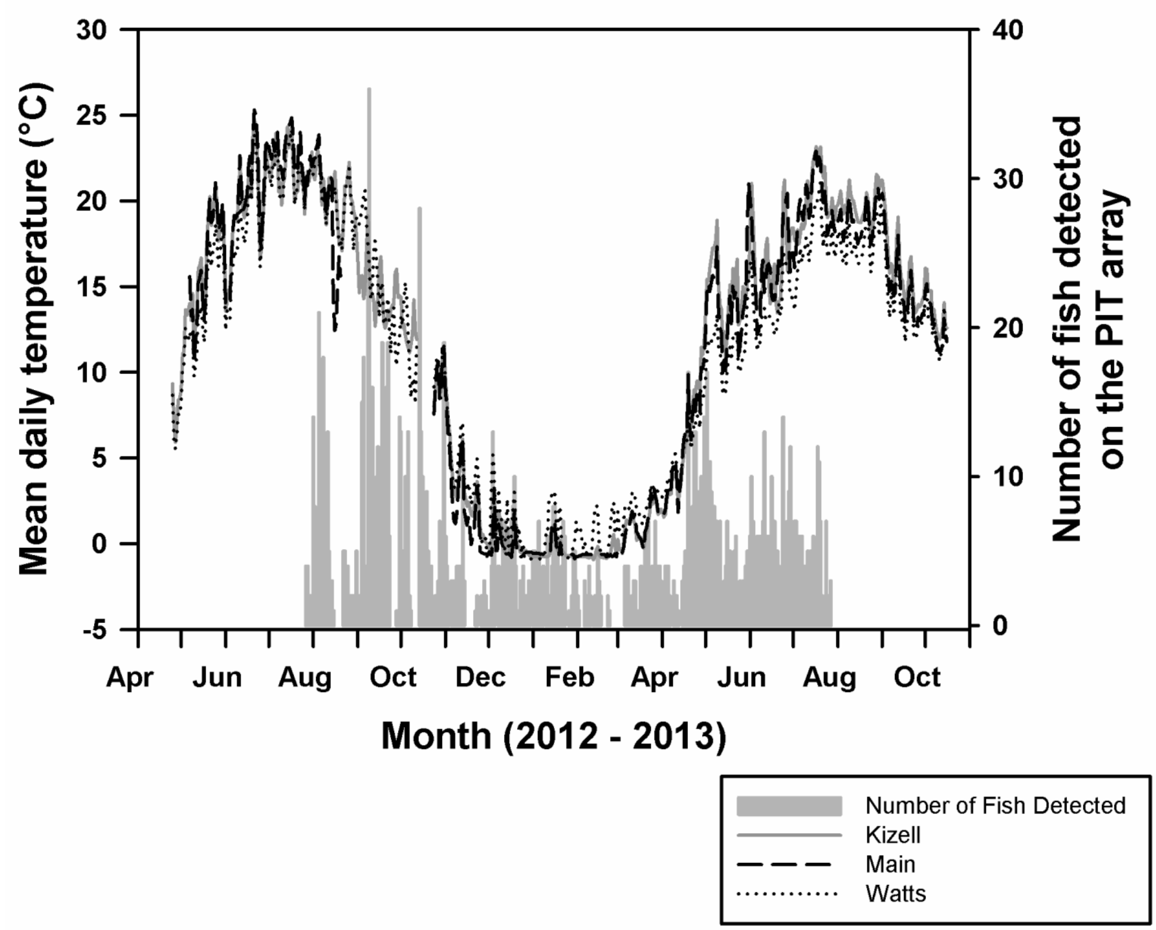

Figure 3.3. Mean daily temperatures for Kizell (dark grey line), Main (black dashed line), and Watts (black dotted line) from 25 April 2012 – 12 October 2012, and 24 October 2012 – 16 October 2013. Also included is the number of fish detected on the PIT array (grey bars) each day from 27 July 2012 – 27 July 2013.

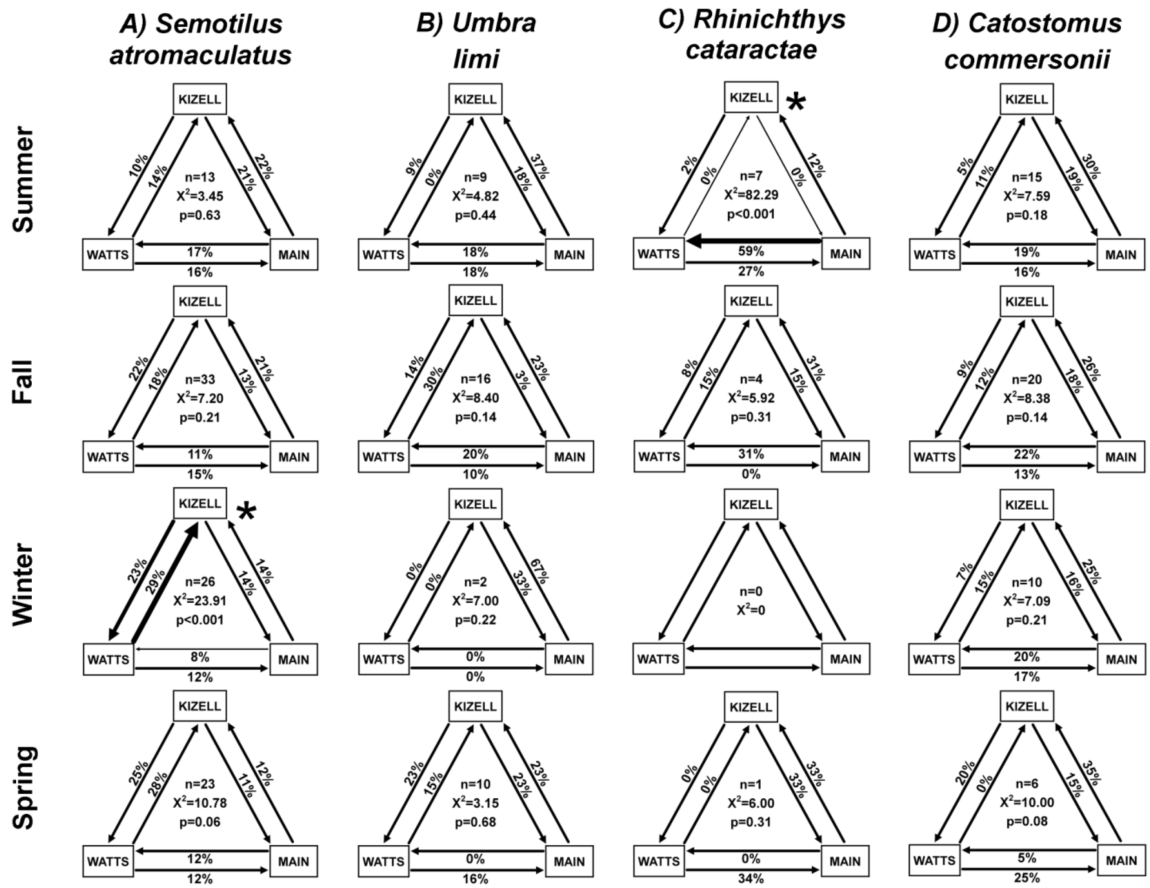

Figure 3.4. Proportion of movement among reaches for (A) creek chub, (B) central mudminnow, (C) longnose dace, and (D) white sucker and across the seasons. Each percentage is the proportion of movement in that given direction. In the middle of each diagram the sample size (n) and results of the chi-square goodness-of-fit test. Degrees of freedom = 2 for all tests, and statistical significance is $p<0.05$ and identified with (*). For test with significant results, the lines representing the directional movements which contributed more to the chi-square value are either thicker (representing a greater proportion) or thinner (representing a smaller proportion).

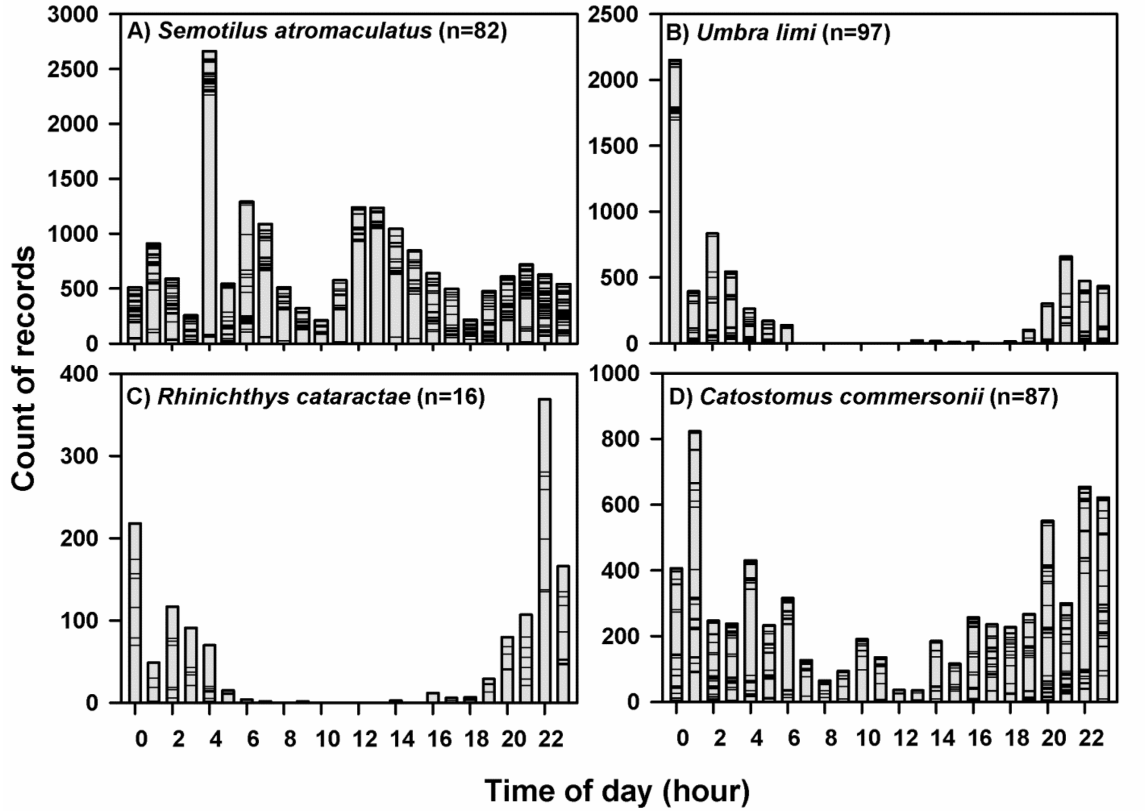

Figure 3.5. Diel records (detections) for a) creek chub, b) central mudminnow, c) longnose dace, and d) white sucker over the entire study period. Stacked bars show the relative contributions of individuals. Note that the scale differs on the x-axis.

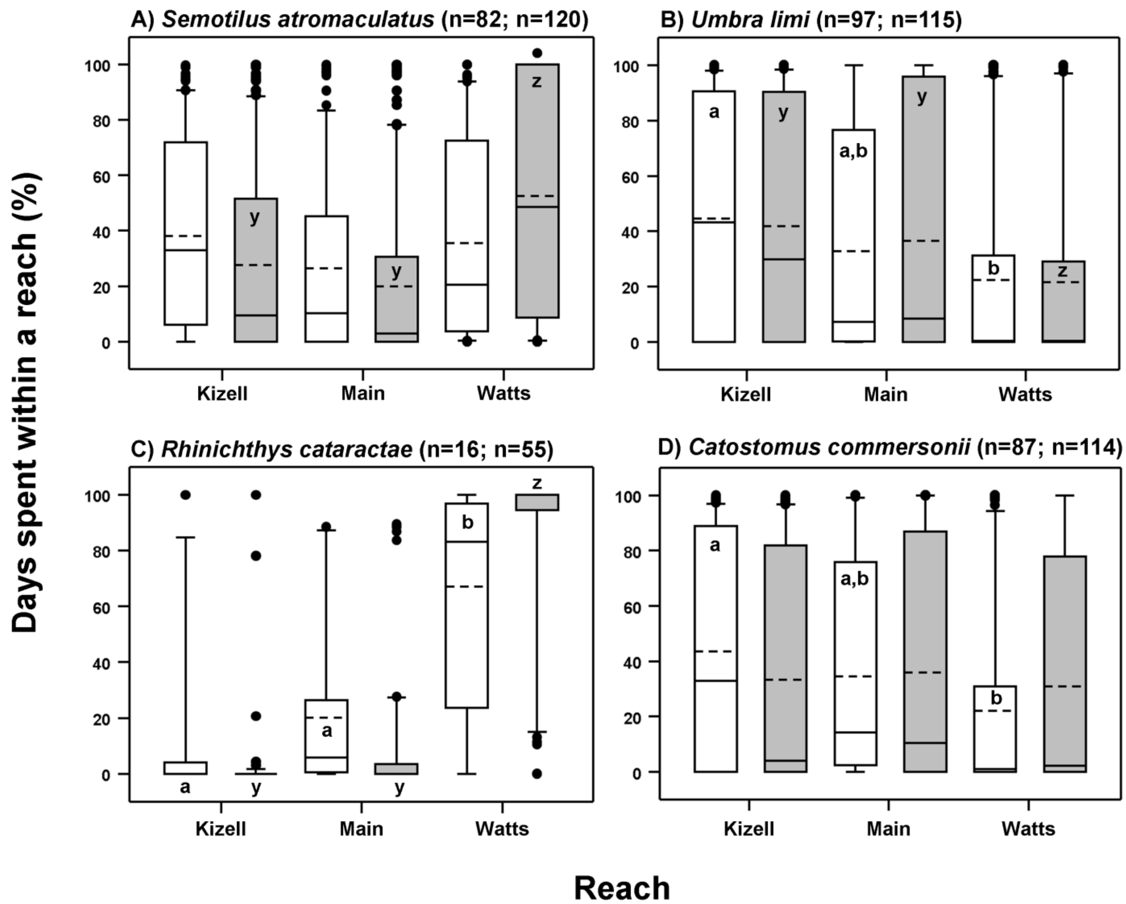

Figure 3.6. Box-plots outlining the residency (proportion of days spent) within each reach for a) creek chub, b) central mudminnow, c) longnose dace, and d) white sucker. The data source shown was either compiled from PIT array detections (white boxplots), or a combination of PIT array detections, recaptured tags, and tags detected using the mobile PIT reader (grey boxplots). Residency times that differed significantly ($p<0.05$) between reaches are indicated by a different letter. The mean and median proportion of days are represented by the dashed line and solid lines, respectively. The lower and upper boundaries of the boxes represent the 25th and 75th percentiles, respectively. The whickers (error bars) are the 10th and 90th percentiles, and outliers are shown as points.

Chapter 4. General Discussion

4.1 Findings and implications

Research on surface drains, both urban and agricultural, and their connection to stream and river ecosystems is only beginning to emerge in the literature. The biological value and potential that earthen drains (ditches or swales) offer as anthropogenic refuges has lacked recognition to date (Chester & Robson 2013). Yet, studies have determined that agricultural drains can support aquatic and terrestrial taxa including plants, invertebrates, fish, amphibians, birds, and mammals (Mazerolle 2004; Stammler 2005; Herzon & Helenius 2008); and urban surface drains can sustain equally diverse macroinvertebrate communities as that found in less urban systems (Vermonden et al. 2009). In addition, Stammler et al. (2008) demonstrated that fish assemblages, as well as the proportion of spawning, mature, and young-of-year fish, found in agricultural drains did not differ from that found in streams. To my knowledge, similar research within urban systems is rare and so through my research I have demonstrated that urban surface stormwater drains are also capable of supporting various stream fish species. In chapter 2, I showed that the fish assemblage in earthen surface drains could be completely distinct from contiguous streams. In chapter 3, I demonstrated that these drains and streams are highly interconnected, and fish are able to, and do, move freely between these systems.

The biological potential of earthen surface drains provides an opportunity to restore and enhance urban landscapes; however adjustments to the current infrastructure to accommodate various species and seasons would be advisable if sustainable water management is to be achieved. In my thesis, I demonstrated that the fish assemblage structure of Kizell Drain was very similar between summer and winter and that various fish species (e.g. creek chub, central mudminnow, and white sucker) continue to move between Kizell Drain and Watts Creek over the winter. This implies that surface drains, like Kizell, can support several fish species throughout the year and that the design of these types of drains should reflect the range of seasonal habitats used by fish. Considering that suitable overwintering habitat for stream fish is understood to include in-stream cover, increased volume (e.g. deep pools), and reduced velocity (Moshenko & Gee 1973; Cunjak 1996), the current design for channelized, shallow, and debris-free stormwater drains would need to be altered in order to enhance the biological integrity of the system. Winter, especially in Northern regions, presents unique challenges and stressors to stream fish, such as reduced temperatures, barriers from ice jams, and reduced primary productivity (Cunjak 1996; Brown et al 2011). In addition, the behaviours (e.g. movements) and habitats used in winter differ among fish species, which makes it difficult to determine exactly how habitat in drains could be adapted to incorporate a whole fish community, and suggests that habitat complexity will be the key to successfully enhancing surface stormwater drains (Cunjak 1996). Therefore, redesigning these drains to include habitat complexity and the connectivity between drains and streams will be vital to the conservation of stream fish within anthropogenic refuges and urban streams.

The emerging research on both agricultural and urban drains has called to attention the management of water within human developed landscapes. A common recommendation made by studies is that the perception and management of drains needs to be adjusted to consider the biological element of anthropogenic waterways (Chester & Robson 2013). Although researchers, conservationists, and resource managers may recognize the biological potential of stormwater systems, it is often municipalities that manage and maintain drains (Roy et al 2008). Around the globe, urban stormwater drains are viewed as infrastructure and as such the management and policy does not reflect their biotic potential. In addition, multiple entities tend to govern different components of a watershed resulting in fragmented responsibilities, and leads to significant impediments to managing stormwater systems (Roy et al 2008). Therefore, in order to move toward sustainable water management that incorporates biological features, cooperative and adaptive management among various governing entities will be essential.

4.2 Future directions

Although emerging research has demonstrated the biological potential of some types of drains, huge gaps in our understanding and knowledge of these systems still exist. In order to successfully enhance and maintain healthy urban aquatic ecosystems, further research is imperative to bridge these gaps in our knowledge. Through my research, several questions have emerged with regard to Kizell Drain, such as how far are fish traveling and residing within the drainage system, how the assemblage structure would respond to cleaning or channel relocation, and what specific habitats are being used over the winter. In general, the physiological condition and stress of fish residing within drains would need to be evaluated in order to determine specific thresholds and management goals to sustain the biological integrity of these systems. For example, Vermonden and colleagues (2009) suggested that lowering nutrient levels and turbidity, and increasing aquatic vegetation would result in higher macroinvertebrate diversity in urban surface drains. Similar habitat and water quality requirements would need to be determined for fish populations. In terms of restoration, research on the mechanisms and processes that drive the biological patterns observed is highly lacking (Kaushal & Belt 2012), especially considering the uniqueness of urban ecosystems. A better understanding of the mechanisms (e.g. impervious surfaces) driving species dispersal and habitat usage could help conservationists and resource managers create more specific targets for aquatic restoration. Also comparative research across many types of drains (i.e. surface vs subsurface and earthen vs man-made), drain cleaning histories, or geomorphology and hydrology will help to better understand the variability among these systems and define more specific management goals. Lastly, as urban environments evolve and expand, long-term studies will be an asset in monitoring how aquatic organisms respond to aging infrastructure.

4.3 Overall conclusions

Through the two studies presented in this thesis, I have shown that fish move into and exploit some urban drainage habitats across multiple seasons, including winter. Overall I conclude that earthen surface stormwater drains can be a functional component of urban watersheds throughout the year, and recommend that these types of drains be regarded and managed as interconnected to the larger stream and river network.

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