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Investigating the Role of Long Distance Dispersal in the Response of Stream Fishes to Urbanization

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Investigating the Role of Long Distance Dispersal in the Response of Stream Fishes to Urbanization

Andrea Davis

A Thesis Presented in Partial Fulfillment of Requirements for the Master of Science in Integrative Biology for the Department of Evolution, Ecology, And Organismal Biology

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Abstract

I conducted a 7-month mark-recapture study in two watersheds differing in urban impact in order to assess the role that long distance dispersal plays in the response of tolerant stream fishes to urbanization. Our two stream sites included a heavily impacted urban stream (watershed impervious surface cover ~30%) and a mildly impacted rural stream (watershed impervious surface cover ~6%). Species of interest were marked with 12mm HPT PIT tags and included a specialist, *Campostoma oligolepis* (n=189 urban site, 200 rural site) and a generalist, *Lepomis auritus* (n=136 urban site, 182 rural site). Three resampling instances for each site were conducted between September of 2016 and February 2017 with a portable PIT antenna. Movement data was analyzed with R package Mclust to estimate the proportion of mobile and stationary components for each population as well as parameter estimates (mean, variance) for each component. *L. auritus* in the urban site exhibited a greater proportion of long distance dispersal than in the rural site, and the rural population did not fit a heterogeneous model better than a homogeneous model. *C. oligolepis* had a greater proportion of long distance dispersal in the rural site, but both the mobile and stationary components had a significantly greater mean movement distance in the urban site (mobile component mean=217m urban, 131m rural; stationary component mean=51m urban, 21m rural). Tolerant generalists may increase long distance dispersal but not home range movement in urban streams while specialists may increase movement distances for both home range and long distance dispersal.

Chapter 1- Literature Review

Urbanization

Stream habitats are a complex assemblage of physical, chemical, and biotic components- all of which can be influenced by their surrounding watershed. Natural variations in morphology, water chemistry, and biota of streams are to be expected based on their geographical location. However, human land use can alter streams in ways that consistently result in degraded habitats supporting lower overall diversity (Walsh et al. 2005, Wenger et al. 2009). Streams are facing increasing urban impact- a product of both an increase in populations residing in urban areas and the breadth of impervious surface cover resulting from urban and suburban development (Meyer et al. 2005). Research focusing on urban streams has shown that urbanization can set into motion a cascade of alterations to stream ecosystems ultimately altering their physical, chemical, and biological elements (Walsh et al. 2005, Wenger et al. 2009).

Urban land use alters the morphology and hydrology of streams. The most consistent shift seen in urban streams is a “flashier hydrograph” defined by Walsh et al. (2005) as the increased frequency of erosive flows large enough to cause hydraulic disturbance to biota and likely to cause channel incision and bank erosion. Studies consistently support the correlation between urban land use and a flashier hydrograph, where the frequency of peak flows are increased (Wheeler et al. 2005, Roy et. al 2005, Walsh et al. 2005, Steuer et al. 2009). Stream sedimentation also increases in unforested, urban areas (Wheeler et al. 2005, Walters et al. 2003) resulting in the transformation of clear streams with coarse beds to turbid streams with finer beds (Walters et al. 2003). Channel complexity, a measure of the spatial variance in habitat types and sediment sizes, is decreased while channel width is increased (Walters et al. 2003).

Water quality of urban streams is often altered with specific contaminants varying based on sources of point and non-point pollution located in the watershed (Wenger et al. 2009). Inorganic nutrient enrichment of streams is correlated with urban land use (Kaushal 2006) which may be a product of both increased nutrient input (Wheeler et al. 2005, Wenger 2009) and decreased nutrient uptake velocity (Meyer et al. 2005). Reduced benthic organic matter in urban streams may result from high discharges flushing organic matter from stream bed sediment (Meyer et al. 2005) or from a reduced input of organic matter caused by a reduction of riparian vegetation. While underlying geology produces naturally variable conductivity, solute concentrations associated with human land use of watersheds causes a consistent increase in conductivity along urbanization gradients (Wenger et al. 2009).

Physical and chemical alterations to streams in urbanized watersheds ultimately result in biotic compositions that are lower in diversity of sensitive, endemic species and dominated by tolerant, cosmopolitan species (Scott and Helfman 2001, Walters et al. 2003, Walsh et al. 2005, Wenger et al. 2009). This pattern is consistent among fishes and macroinvertebrates (Davies et al. 2010, Wheeler et al. 2005) and is termed biotic homogenization (Scott and Helfman 2001, Walters 2003, Scott 2006). Urban stream studies often aim to identify causes leading to biotic homogenization as the extirpation of endemic species leads to irreversible losses in species diversity. With the threat of urbanization growing, identifying agents of homogenization and what causes the sensitivity or tolerance of certain taxa can help anticipate land use effects on stream fish species before they are impacted and guide management in a more effective way.

Hydrology (Roy et al. 2005, Walters et al. 2003), sedimentation, and stream bed composition (Walters et al. 2003, Scott 2006) have all been identified as correlates of biotic homogenization. However, weighing the relative importance of any one variable engendered by urban land use on biotic composition may be only equally or less important than considering the proportion of watershed area that is impacted by urban land use. Studies have shown that land use variables were superior to topographic factors as predictors of fish homogenization (Scott 2006), that impervious surface cover proved a better predictor of stream condition than any combination of hydrological effects (Burns et al. 2014), and that global, historical, and current land use models were better predictors of fish species occurrence than hydrogeomorphic models (Wenger 2008). This suggests that the interaction of multiple effects following watershed urbanization is more impactful to fish assemblages than any one variable alone. For this reason, using landscape level metrics such as total imperviousness [TI] (Walters et al. 2003), effective imperviousness [EI] (Wang et al 2001), or attenuated imperviousness [AI] (Burns 2015) will be the best way to quantify the threat of biotic homogenization within streams.

Species Tolerance to degraded environments

While fish assemblages in urban streams are dominated by tolerant taxa, predicting how a species may react to urbanization based on its ecology may be difficult, especially if the life history traits of every species in a stream have not been studied (Kemp 2014). Studies have observed that regional endemics or fluvial specialists tend to show sensitivity to urban streams while lentic-tolerant species are less so Walters et al. 2003, Roy et al. 2005, Scott and Helfman 2001). However, intraspecies tolerance can vary among regions (Meador et al. 2005, Utz et al. 2010) and some species with life histories and morphologies similar to sensitive taxa exhibit robustness (Wenger 2009). Utz et al. (2010) observed genera which included species that exhibited both sensitivity and tolerance to land use gradients. Currently the only way to truly identify a species' tolerance to urbanized streams is by witnessing the impact urbanization has had on a population. Identifying more generalized metrics of tolerance that are quantifiable across all species and regions may be more useful than relying on life history traits and may aid in the vital step of preventive management by reliably assessing the sensitivities of stream fishes.

Geographic range serves as a good relative measure of a species’ risk of imperilment both in terrestrial and aquatic ecosystems (Angermeier 1995). Endemics’ small ranges suggest specialist ecologies that depend upon a narrow range of conditions for persistence. Cosmopolitan species, with larger ranges, may be more flexible in habitat requirements. Generalist habitat requirements cannot solely be responsible for the trends seen in some tolerant species following alterations to watershed land use, such as increases in both presence and abundance (Utz et al. 2010). Increases in presence and abundance indicate that some tolerant species are able to not only persist in disturbed habitat but colonize new areas and expand range. Albanese et al. (2009) found that mobility was an important determinant of colonization and population recovery. Theoretical models have demonstrated that movement may counterbalance the effects of habitat loss and fragmentation in altered landscapes (Niebuhr 2015) and that characteristics of a population’s dispersal can serve as an indicator of their ability to increase range (Goldwasser et al. 1994, Kot et al. 1996). Dispersal may be a fundamental aspect of fish behavior involved in their ability to tolerate stream systems affected by human land use.

Fish Dispersal

Increased long distance dispersal by non-generalist species in disturbed habitats may allow them to maintain presence by locating more suitable habitat in disturbed streams. A species exhibiting both a generalist habitat preference and high mobility would likely be able to increase rapidly in both presence and abundance in a degraded stream system. This process may be even further expedited by a sudden increase in unoccupied habitat due to the extirpation of more sensitive taxa. Dispersal may then be a factor in the increase of tolerant species observed in streams impacted by land use, the “native invasion” step of biotic homogenization referred to in Scott and Helfman (2001). Individuals of the generalist, urban-tolerant species *Semotilus atromaculatus* exhibited very high proportions of movement within streams with over 76% of individuals observed in the study moving >100m away from the nearest neighboring habitat they were captured in. Correspondingly, two darter species facing risk of imperilment were observed with very low rates of movement between habitats (Labbe and Fausch 2000, Holt et al. 2003). Further investigation into the connection between dispersal and species range will reveal the role that mobility plays in allowing for the persistence and expansion of stream species within disturbed habitats. Investigating dispersal’s role in the adaptability of stream fishes to degraded habitats first involves understanding the benefits or mobility, what can spur or alter fish movement, and commonalities among dispersal patterns of fish populations.

Movement is a fundamental process linking stream fish to their environment. understanding where fish move and why can reveal how fishes are utilizing habitat and what causes changes in their habitat usage. The benefits of dispersal for stream fishes depend on the scale at which it occurs. Small-scale movement behaviors are essential to occupying and exploiting the resources of the most suitable habitats for survival and growth (Gowan and Fausch 2002). Small scale movements can be associated with home range behavior and typically occur at scales between 20-50m (Rodriguez 2002). Large scale movements, also termed long-distance dispersal (LDD), occur at scales much greater than a habitat unit (typically in the 100’s of meters) and provide fishes with an opportunity to respond to habitat change (Walker and Adams 2016) and colonize new areas (Albanese et al. 2009). Environmental factors that can are correlated with movement include physical aspects of habitat, interspecific interactions, and modifications of habitat at the landscape scale. Correlates of movement, however, are often species specific and temporally variant. The study of fish movement can reveal patterns of habitat use and how they change with time, across regions, and in the face of environmental disturbance. Identifying the factors that alter fish movement also gives insight into what variables are most impactful on stream fishes.

The complexity of stream habitats can impact the dispersal of fishes within them. Habitat complexity was negatively correlated with movement for chub species in two studies. Albanese et al. (2004) observed that *Semotilus corporalis* were less likely to disperse out of more complex habitats. Similarly, pool complexity and area were negatively related to movement for creek chub *Semotilus atromaculatus* (Walker and Adams 2016). Three darter species had varying movement responses to environmental variability (Roberts and Angermeir 2006), demonstrating a common trend of movement correlates in stream fishes: species-specificity. Since urbanization often results in an overall reduction of stream diversity due to channelization and sedimentation, it is likely to impact the movement behaviors of resident fishes and may drive higher rates of long distance dispersal for species that have to locate more suitable habitat.

Stream velocity has also been shown to affect dispersal both positively and negatively. Albanese et al. (2004) observed increased movement rates for several species during flow events. However, increasing velocity at road crossings negatively correlated with the probability of fishes dispersing across them (Warren and Pardew 1998). Velocity of riffles was also negatively correlated with between-pool movement of three cyprinid species (Schaefer 2001). The effect that velocity has on stream fish movement likely depends on that species' ability to swim against stronger currents.

Modifications to stream habitats can influence connectivity. Storm water drains have been shown to be used by stream fishes as intermittent habitat and connection points (Bliss et al. 2015). Some modifications reduce connectivity by acting as barriers to dispersal. Road crossings can reduce the overall movement of fishes with the effects varying by species. Culverts result in the greatest reduction of movement (Benton et al. 2008) and also reduce the diversity of species exhibiting movement (Warren and Pardew 1998). Efforts to quantify the permeability of structural barriers (Pépino et al. 2012) may aid stream management by maximizing connectivity when making modifications to stream habitats.

However important habitat variables may be to dispersal for a specific species, they can also be influenced by interspecific biotic factors as well. Gilliam and Fraser (2001) found that water level was only influential to movement of *Rivulus* in the presence of a predator fish species. Conversely, spatial complexity of pool areas increased the probability that *Rivulus* would move into pools occupied by predators. This interplay of abiotic and biotic variables reveals that the already complex, species-specific correlates of movement can be further modified by community assemblages. The presence of predators in particular seems to directly modify dispersal behaviors. Predator introduction led to a nearly three-fold increase in movement rates of three cyprinid species in a simulated habitat experiment (Schaefer 2001). Movement distances and overall movement distribution of *Rivulus* drastically increased in the presence of predators (Gilliam and Fraser 2001). In a follow-up experiment, Fraser et al. (2006) found that the increased movement of *Rivulus* was not linear with increased predation threat, presenting peak movement at an intermediate threat (one predator) and a decrease of movement at a higher threat level (two predators). Increasing movement may not always be beneficial to stream fishes in avoiding predation, particularly by terrestrial predators. Brook trout observed in a first-order stream has much higher rates of mortality caused by direct predation when exhibiting higher-than-average movements associated with spawning behavior (Pépino et al. 2015).

Dispersal is closely linked to habitat, interspecies influences, and alterations of stream structure. All of these correlates, however, are likely to vary by species and temporally. Identification of common factors influencing fish movement may be more useful on a management scale (Albanese et al. 2009) than those that are inconsistent across regions, species, and seasons. To do so requires identifying aspects of dispersal that are common to all fish species by analyzing their dispersal patterns. Dispersal patterns concern rates and distances of movement demonstrated by fish populations. Studies have revealed that fish dispersal patterns share common distributions that may help in comparing movement across species and regions (Skalski and Gilliam 2000, Rodriguez 2002). Comparing dispersal patters on the same species in different environments can reveal if the environment is altering the way in which they disperse. Dispersal patterns can also be compared across species in the same environment to determine if they are responding differently to the same habitat variables. Fish dispersal patterns can be analyzed in order to determine dispersal's role in a species' ability to persist within streams and colonize new habitats. Evaluating dispersal distributions may also be able to reveal the invasive potential of fish species and their ability to respond to disturbances.

Patterns of stream fish dispersal

Dispersal distributions, a plot of the probability (y-axis) that an individual in a population will move a certain distance (x-axis), are leptokurtic in nature. Leptokurtic distributions vary from normal distributions by displaying more observations in the center and tails than would be expected of a normal distribution (higher peak/more outliers). This distribution is thought to result from heterogeneity in dispersal within fish populations. While the majority of fish in a population exhibit only small-scale movements associated with home range behavior, some fishes disperse over distances much greater than the mean movement for the population. Studies that have evaluated the distribution patterns of fish dispersal have supported this heterogeneity and typically refer to the two resulting groups as the "stationary" and "mobile" components of a population (Skalski and Gilliam 2000, Rodriguez 2002, Radinger and Wolter 2014) (see Figure 2). These two groups do not likely result from individuals in a population having a static tendency to either "move" or "stay". Individual fish have been observed exhibiting periods of long term dispersal followed by periods of stationary behavior in long-term studies (Alldredge et al. 2011, Booth et al. 2014) which is consistent with the home range shift model of fish behavior (Crook 2004). Home range sizes can also be variable across seasons and with habitat morphology (i.e. stream slope, depth, substrate) (Slavik et al. 2005). Deviations from the heterogeneous model could be expected if a population exhibits periods of population-wide dispersal such as mating migrations. Empirical studies of stream fishes, however, consistently result in distributions that best fit the heterogeneous model (Rodriguez 2002, Coombs and Rodriguez 2007, Radinger and Wolter 2014) which provides a consistency across species and populations that can be used to evaluate dispersal in any environment. Since the mobile component of a population's movement distribution is representative of that population's long-distance dispersers, evaluating this component of a fish population may reveal their ability to respond to habitat disturbance.

Models have shown that variability among individuals in their rates of movement can markedly increase the rate of spread of a population and that even a small proportion of the population exhibiting long distance dispersal can influence spread of the population (Goldwasser et al. 1994). This indicates that the proportion of the mobile component of fish populations, which is represented in the tails of the leptokurtic distribution, may indicate their ability to increase in range and colonize new habitats. This may become especially important in habitats affected by urbanization where the reduction of overall habitat complexity may reduce preferred habitat. An empirical study supporting this idea observed higher dispersal probability and distances in two invasive *Gambusia* species compared to similar, endemic *Gambusia* (Rehage and Sih 2004). Calculating the proportion of a population represented in the mobile component of a fish population will provide the number of fish exhibiting the long distance dispersal behaviors that may be important in the tolerance to degraded habitat such as urban stream systems. If long distance dispersal is important in tolerating urbanization, the mobile component of populations of the same species in stream habitats of varying urban impact would be expected to represent differing proportions of the population.

Study Design

If long distance dispersal is important in the tolerance of stream species to urbanization, the mobile component of a population would be expected to differ in environments based on their level of impact from urban land use. To investigate this, populations of the same species should be studied within stream habitats that have differing levels of urbanization in their watershed area. Mark recapture studies provide the best way to observe fish movement and recent advances in Passive Integrated Transponder (PIT) tagging technology allow for non-invasive resampling methods that can be conducted more often than traditional resampling methods. While comparing sensitive and tolerant species would be preferable, study species would need to be present in urban and rural streams in populations large enough to provide a good sample size for a mark-recapture study which is unlikely for sensitive species. Studying two species both known to be tolerant to urbanization presents a way to look at how mobility changes in habitat generalists and habitat specialists. Since generalist habitat requirements are a common attribute for tolerant species, comparing the importance of this trait with a population's level of long distance dispersal will help to reveal the relative importance of either aspect in the urban tolerance of stream fishes. In order to do so, one generalist species and one specialist species present in all selected sampling sites can be compared across varying levels of stream urbanization. In the Etowah River Basin, two species are present in suitable numbers and stream sites that fit this criteria: *Campostoma oligolepis* and *Lepomis auritus*.

Campostoma oligolepis, the largescale stoneroller, is a cyprinid species present in the Etowah River Basin. It has been documented to not only persist but thrive in urban systems (South and Ensign 2013). They are herbivorous and feed on algal communities that grow on stream rocks. Despite their lack of generalist attributes they have persisted within urban streams and urban populations have been observed with longer spawning seasons, higher GSI values, and shorter gut lengths than populations within rural streams (Mutchler et al. 2013, South and Ensign 2013). Since they depend on coarse substrate for algal feeding, this relationship is especially counterintuitive as urban streams are typically dominated by silt/sand bottoms with coarse substrate occurring more sparsely than in rural streams. I hypothesize that higher rates of long distance dispersal have allowed *C. oligolepis* to locate and exploit suitable habitat within degraded urban streams.

Lepomis auritus, the redbreast sunfish, is a centrarchid species present in the Etowah River Basin. It is a documented urban tolerant generalist that will feed on insects, snails, and other small invertebrates. It is lentic tolerant and prefers low-velocity, deeper habitats. I hypothesize that *L. auritus* will exhibit a lower proportion of long distance dispersal in both urban and rural streams since movement for *L. auritus* is not as important for locating foraging. However, I still hypothesize that they will have higher rates of long distance dispersal in urban environments than rural environments due to the low habitat complexity.

I also hypothesize that longer distances of dispersal will be observed in populations of both species residing in urban streams. Urban streams have increased sedimentation and channelization that results in long stretches of homogenous habitat. This results in isolation of complex habitat units that will require longer distances of dispersal to move between.

Predictions:

1. The mobile components of *C. oligolepis* and *L. auritus* will represent a larger proportion of the population in urban stream systems than rural stream systems.
2. Individuals grouped into the mobile component within urban populations of *C. oligolepis and L. auritus* will have an average distance of dispersal significantly higher than individuals in the mobile component of rural populations.

3. The mobile component for *C. oligolepis* will be greater in both urban in rural sites than *L. auritus* and both components will have a higher mean movement distance for *C. oligolepis* than *L. auritus* in both urban and rural streams.

Chapter 2- Validation of Experimental Methods

Section 2.1- Investigating the suitability of Campostoma oligolepis for marking with 12mm PIT tags

Introduction

Prior to a mark-recapture study implemented in field, I wanted to ensure the marking methodology did not negatively impact the species we intended to tag. Although other cyprinid fishes have been used for studies involving PIT telemetry (Bolland et al. 2009), *C. oligolepis* has a unique physiology due to their herbivorous ecology. Their intestines occupy a large portion of the coelom, which is the site most commonly used for PIT tag implantation in small fishes (intramuscular tagging is only suggested in fishes >250mm) (Figure 3.2). A study focused on survival and retention of PIT tags found significant growth and mortality effects compared to control groups in fishes less than 55mm TL (Richard et al. 2012). For this reason, I chose to use 55mm as the minimum size for PIT tagging. A preliminary tag retention study was undertaken in a laboratory setting to investigate the effects of PIT tagging on *C. oligolepis* specimens.

Methods

In order to ensure that *C. oligolepis'* unique physiology did not make it less suitable to retain tags and rule out any effects of mortality the tagging procedure might induce, wild caught *C. oligolepis* were kept in the laboratory and split into three groups which were housed in separate aquaria. Fish were collected via backpack electrofishing in Pumpkinvine Creek located in Paulding County, Georgia. All groups were allowed 14 days to acclimate to aquaria conditions before being either tagged (Group 1), injected with the surgical needle but not tagged (Group 2), or anesthetized but undergoing no surgical procedure (Group 3).

All fishes were anesthetized in a solution of 140mg/L of MS-222 until loss of motor function was evident. Fishes were then either allowed to recover (Group 3) or injected in the coelom with a surgical PIT-implantation gun (Biomark, Idaho) which was either removed (Group 2) or used to inject a 12mm PIT tag (Group 1). Tagged fishes had their total length (TL) and weight recorded and stored with their unique tag ID. Fish were then allowed to recover until motor function was fully restored. If fish were showing signs of extreme stress or injury that resulted in the inability to recover they were euthanized in a solution of 240mg/L MS-222.

Recovered fishes were kept in aquaria for an observation period of 4 weeks and any mortalities removed and noted. Fishes were fed daily and kept on a 12 hour light/dark cycle. After the observation period, fishes that had received tags were captured, anesthetized, and had their TL and weight recorded to compare to values collected at time of tagging.

Results

After this initial round of tagging, it became clear that any mortalities would be evident immediately following injection. Inexperience with tagging technique led to high mortality for the tagging procedure, with 5 out of 15 *C. oligolepis* in Group 1 having to be euthanized post-tagging due to surgical error. Similar mortalities were present in Group 2. All fish in Group 3 recovered fully. All fish that survived the initial procedure retained their tags for a 4-week observation period and had no further mortalities. TL/weight ratios were not affected in any consistent direction by the tagging procedure in Group 1 with the ratios increasing for about half of the individuals and decreasing for the other half.

Discussion

Perhaps the most valuable information garnered from this investigation into PIT-tagging *C. oligolepis* specimens was that the location of the tag injection site was important. Further analysis of tagging mortalities showed that, in most cases, the implantation needle had punctured the intestines. Since *C. oligolepis* has such a long digestive tract which is coiled around its swim bladder, incorrect placement of the tagging gun easily results in intestinal penetration. Refinement of tagging techniques via experience with other *C. oligolepis* specimens allowed us to reduce mortality significantly. Experience with tagging has been shown to influence mortality. Richard et al (2012) found that survival for juvenile brown trout could be affected by the individual tagging the fish, but only for fishes <55mm. Although all of our fish greatly exceeded this size class (the smallest was 79mm), it is possible that the unique morphology of *C. oligolepis* makes experience with tagging important.

Consistent with Acolas et al. (2007), no impact on growth was seen in tagged fish. 100% tag retention of those that survived the tagging procedure align with the 96%-100% retention observed in other cyprinid species (Bolland et al. 2009). Survival of tagged fishes was not different in tagged fish than the control group, which has also been observed in several studies (Richard et al. 2012, Bolland et al. 2009). We concluded from this investigation that as long as *C. oligolepis* specimens recovered from the initial tagging procedure, they would not experience any increased mortality or fitness reduction from being implanted with 12mm PIT tags.

Section 2.2- Estimating efficiency of the portable PIT detection antenna

Introduction

Since a portable PIT-detection antenna (Biomark, Idaho) was my primary method of tracking fish movements, I was interested in gauging the efficiency of this methodology and any potential influences on detection. Previous studies have indicated that detection using a portable antenna may be affected by size of tagged fishes (Breen et al. 2009; Cucherousset et al. 2005, 2010; Sloat et al. 2011) and varies among species (Cucherousset 2010; Banish et al. 2016). Abiotic variables such as stream discharge (O'Donnell et al. 2010) and percent boulder substrate (Keeler et. al 2007) have also been observed having an effect on the detection efficiency of portable PIT antennas. Further investigation into influences on the probability of detection can provide information on suitability of sites or species for PIT studies as well as fish length ranges that may maximize detection. This could increase the ability to efficiently utilize materials and field hours when conducting PIT telemetry studies.

In typical mark-recapture studies, the probability of detecting a marked individual can be influenced by whether the marked individuals have survived, moved, and the efficiency of the detection methodology. In order to be able to quantify efficiency of detection alone, we designed a study where movement outside of the resampling area and mortality within the time frame of the study could be effectively eliminated by blocking off the area and resampling within a time frame small enough to assume zero mortality (48 hours). Knowing the actual number of tagged fishes in a sampling area to which are confined to gives the rare opportunity to directly calculate the percentage of tagged fishes detected when sampling with the portable antenna.

In addition to analyzing detection probabilities in a closed system, we were also interested in comparing detection probabilities attained from a 7-month mark-recapture study in two stream sites. Since these systems were open to movement outside of the reach and the time frame was large enough to assume the risk of mortality, only estimates of detection frequencies can be calculated.

Methods

Kennesaw Creek- 48 hour Study

Sampling of the closed system was conducted in “Kennesaw Creek”, a small stream located on Kennesaw State University’s campus. A 240m study reach bordered by an upstream movement barrier was selected and a block net was installed at the downstream reach limit. Eight separate sections were delineated at the intersection of a pool/riffle sequence and sampled via backpack electrofishing with collected fishes of suitable size (>55mm) anesthetized in a MS-222 solution (140mg/L) and implanted with a 12mm PIT tag. Tagged fishes were allowed to recover fully before being released in their section of capture. Species, TL (mm), weight (g), and tag ID were recorded at time of marking.

Over the next 36 hours following marking, four passes of the full 240m reach were made with the portable PIT antenna. The detection frequencies for each resampling occasion were calculated simply by dividing the number of detected fishes for each antenna pass by the number of tagged fishes present in the sampling reach. An overall detection frequency was calculated by dividing the total number of detections for each species by the total number of possible detections for four passes (number of tagged fishes*4). Two species were present in numbers suitable for this analysis, *Lepomis auritus* and *Semotilus atromaculatus*. Frequencies were calculated for each species and species detections were further divided into two size subclasses (<100mm TL and ≥100mm TL) to analyze the effect of size on detection probability. Estimates of the probability of detection were also calculated with program MARK. The probability of survival could be considered constant and fixed at 1, leaving two possible models (constant detection and detection which varies for each sample) which were compared by AIC scores. The model of best fit was used to generate estimates and 95% confidence estimates of detection probability.

Little Noonday and Picketts Mill Creek- 7 month Study

Data collected from the 7-month mark-recapture study using PIT tagging and portable antenna resampling provided three full antenna collections for each of two sites. The two sites, Little Noonday Creek and Picketts Mill Creek, both had initial tagging conducted in July of 2016. Little Noonday Creek had a 740m sampling reach divided into 20 sections and was resampled with the portable antenna in October 2016, January 2017, and February 2017. Picketts Mill Creek had a 649m sampling reach divided into 20 sections and was resampled with the portable antenna in September 2016, October 2016, and February 2017. Two species were included in this study, *L. auritus* and *C. oligolepis*.

Estimation of the probability of detection with the portable PIT antenna was carried out via program MARK. The program returns probabilities of detection and survival that are either fixed (constant across each sampling occasion) or vary by time (different for each sampling occasion). Models containing a mixture of these variables (i.e. fixed for both, fixed for one but varying across time for the other) are generated along with an AIC score. AIC scores for all models were compared via a likelihood ratio test in order to determine if any model fit the data significantly better than another. If no significance was observed in the likelihood ratio test, the model was chosen with the lowest AIC score. Data for each species at each site were analyzed separately. Detection probability for size subclasses within each species were analyzed, with <100mm TL or ≥100mm TL used as the size groups for *L. auritus* and <80mm TL the size groups for *C. oligolepis*.

Habitat Analysis- Picketts Mill and Little Noonday

Habitat data was collected in January 2017 across the entire reach of both study sites to evaluate differences in habitat complexity between urban and rural sites. A top-set wading rod and flow meter were used to measure depth to the nearest .1 ft and velocity (ft/s), and a modified Wentworth scale was used to quantify substrate (Table 3.2). Ten measurements of each variable were taken at random points in each sampling section, with a single measurement of the dominant substrate being recorded at each point of depth/velocity measurement. Since each of our sites consisted of 20 habitat units, 200 total measurements of each variable were recorded for each site.

Measurements from both sites were pooled with habitat scores assigned to each measurement depending on its value. Values in the upper 1/3rd of depths (>1ft) were assigned a 3, between upper 1/3rd-2/3rd were assigned a 2, and the lower 1/3rd (≤.69ft) a 1. For velocity, values in the upper 1/3rd (>.41ft/s) were assigned a 1, between upper 1/3rd-2/3rd assigned a 2, and the lower 1/3rd (≤.1ft/s) a 1 (Table 3.3). Substrate received a 1 for measurement of coarse substrate (anything besides sand, silt, or clay; Table 3.2) and a 2 for fine substrate (sand, silt, or clay). Pivot tables in Excel allowed us to quantify the number of habitat score combinations that occurred in each stream and compare the prevalence of habitat scores or combinations of habitat scores by stream. For example, comparing the prevalence of coarse substrate (all habitat score combinations that had a 1 value for substrate) or “riffle” type habitat (all habitat scores that had a 1 value for velocity and 1 for substrate) among streams.

Analyses of the proportions of habitat types from each stream were conducted that accounted for differences in habitat complexity and availability of preferred habitat types for the species in our study. For *L. auritus*, a lentic-tolerant generalist, deep habitat was considered ideal with all habitat score combinations that had a 3 for depth compared between the two sites. For *C. oligolepis,* a fluvial specialist, we compared the prevalence of “riffle” type habitat score combinations among streams (those that had the highest 1/3rd velocity and coarse substrate). All comparisons were done using a z-test for proportions.

Results

Kennesaw Creek- 48 hour Study

Over all 4 sampling occasions and including both *S. atromaculatus* and *L. auritus*, 32% of tagged fishes were detected in Kennesaw Creek. Individual passes ranged from 28%-40% pooled detection of both species (Figure 2.1). *S. atromaculatus* had an overall 37% detection rate across all 4 antenna passes with individual antenna pass detection varying from 28%-50%(Figure 2.1). *L. auritus'* overall detection was 26% with individual passes yielding percentages from 24%-26% (Figure 2.1). 33% of tagged *S. atromaculatus* and 45% *L. auritus* were never detected on any of the four passes.

Estimates of detection probability from program MARK were identical to calculated detection efficiencies. Both species fit a model that indicated that detection probability of a tagged individual was constant rather than different for each sampling occasion.

For both species, detection was greater for smaller (<100mmTL) than larger (≥100mm TL) fish (Figure 2.2). Smaller *L. auritus* had an overall detection percentage of 36% compared to 13% for larger *L. auritus*. However, 95% confidence intervals for both size classes overlapped. Smaller and larger *S. atromaculatus* had 56% and 31% overall detection frequencies, respectively, and 95% confidence intervals did not overlap. (Figure 2.2)

Little Noonday and Picketts Mill Creek- 7 month Study

For both species, *C. oligolepis* and *L. auritus*, in both streams, Little Noonday Creek and Picketts Mill Creek*,* MARK indicated that the model of best fit was a constant detection and survival probability across all sampling occasions. This means that each of the four populations included in the study will have one value for their estimated detection probability which corresponds to the probability of detection for each antenna sample (Figure 2.3). In Little Noonday Creek, detection percentage was estimated at 34% for *C. oligolepis* and 45% for *L. auritus*. In Picketts Mill Creek, *C. oligolepis* had 18% estimated detection and *L. auritus* 12% estimated detection. *L. auritus* had significantly less of a probability of detection in Picketts Mill, with no overlap in the 95% confidence intervals generated by MARK. *C. oligolepis*' 95% confidence intervals in Picketts Mill and Little Noonday, however, do overlap (Figure 2.3). Smaller fishes for both species had higher detection frequencies than larger fishes in Picketts Mill with *C. oligolepis* <80mm TL at 26% compared to 17% for fish ≥80mm and *L.* auritus <100mm TL at 15% compared to 7% for fish ≥100mm. However, there is significant overlap in the confidence intervals for both species. This relationship was not present in Little Noonday (Figure 2.4).

Discussion

Despite the inability of fishes to leave the sampling reach and a time period that allowed for the assumption of no mortality, we saw variation in the detection efficiency of the portable antenna in Kennesaw Creek. Even in a study using hidden tags in the stream rather than swimming fish, variation in detection efficiency was seen among passes (O'Donnell et al. 2010). However, since all species fit a model in MARK that indicated that the probability of detection is constant rather than variant among sampling occasions, it is not likely that this variation is significant. Differences in detection efficiency were seen in different species. Several studies using the portable PIT antenna have observed detection efficiencies that vary among species (Banish et al. 2015, Cucherousset et al. 2010), sometimes dramatically. Cucherousset et al. (2010) observed species detection efficiencies ranging from .7% to 43%. While *L. auritus* had an overall lower probability of being detected than *S. atromaculatus* in Kennesaw Creek, the 95% confidence intervals generated in MARK displayed overlap.

Size of PIT-tagged fishes seems to have an effect on the detection efficiency of the portable antenna but the direction of this relationship is not consistent across studies. A study done with mottled sculpin found that larger fish were less detectable (Breen et al. 2009) while fish total length (TL) was positively correlated with detection efficiency for salmonids (Banish et al. 2015). Our results indicate that larger fishes of all of *L. auritus* and *S. atromaculatus* have lower detection efficiencies, but not across all sites. This is likely due to differences in available habitat and the preference of habitat for larger fishes. Breen et al. (2009) speculated that the lower detection for larger mottled sculpin was due to their use of habitats containing large woody debris (LWD), which is more difficult to sample. Electrofishing and portable antenna detection efficiencies were similar in Sloat et al. (2011), except for large fishes in pools with high cover complexity. Since the detection distance of the portable antenna is 17-36 cm for 12mm PIT tags (Cucherousset 2005), sampling deeper areas with the portable antenna may reduce efficiency. A study of the life history of *S. atromaculatus* found that fish >age 2 preferred deeper habitat than younger, smaller fish (Moshenko 1973). *L. auritus* are typically pool-associated, and larger fish are also likely to prefer deeper habitats. Larger *S. atromaculatus* were significantly less detectable than smaller ones in Kennesaw Creek, with no overlap in 95% confidence intervals, likely due to their location in harder to sample areas. Larger *L. auritus* in Kennesaw Creek were also less detectable than smaller fish, but there was a slight overlap in the confidence intervals (Figure 2.2). *C. oligolepis* does not show a significant relationship with length and detection efficiency in either site, likely because both large and small fish prefer shallow habitat (Figure 2.4).

L. auritus was significantly less detectable in Picketts Mill Creek than in Little Noonday Creek, with no overlap in confidence intervals of detection estimates. Habitat analysis of the two sites showed that Picketts Mill had significantly more deep area (Z-test, p=.002) and coarse substrate (Z-test, p=<.00001) (Figure 3.6). Detection efficiency has been found to be negatively correlated with the presence of boulder substrate (Banish et al. 2015, Cucherousset et al. 2009, Linnansaari et al. 2007).

High complexity pools have also been observed to negatively impact the detection efficiencies of larger fish (Sloat et al. 2011). Comparison of all three sites for *L. auritus* (Figure 2.5) indicates that complexity of habitat alters detection efficiency for this species. Little Noonday is a highly impacted urban site with very little habitat complexity. Kennesaw Creek is moderately impacted, and Picketts Mill is the least impacted and most complex site. The detection efficiencies of *L. auritus* are negatively correlated with the relative complexity of these streams. Larger *L. auritus* are also significantly less detectable in Picketts Mill than Little Noonday. Large *L. auritus* in Picketts Mill can find their preferred habitat, deep pools, whereas those in Little Noonday are more likely to use refuge in shallower areas that are much easier to sample with the antenna. This also explains why size does not impact *L. auritus* detection probability in Little Noonday. Larger fishes that would typically take advantage of deeper, complex habitat areas do not have it available.

This study has shown that the efficiency of detection of PIT tags with a portable antenna can be influenced by the size and species of the fish, but that the degree and direction of these effects will vary based on stream habitat. The prevalence of deep habitat and coarse substrate is likely to negatively impact detection efficiencies for all species, but particularly those that prefer deep pools. Studies wishing to maximize the number of detections in a mark/recapture study using portable PIT antennas may want to focus on smaller fish if their species of interest prefers pool habitat. It may also be worthwhile to combine portable PIT sampling with backpack electrofishing in highly complex streams.

Chapter 3- The Role of Long Distance Dispersal in the Response of Stream Fishes to Urbanization

Introduction

Urbanization of a watershed has multiple impacts on the physical, chemical, and biological elements of streams. Urbanization consistently results in the increased frequency of erosive flows (Roy et al. 2005, Steuer et al. 2009, Walsh et al. 2005, Wheeler et al. 2005) and sedimentation (Walters et al. 2003, Wheeler et al. 2005). Overall, watershed urbanization reduces channel complexity and results in the transformation of clear streams with coarse beds to turbid streams with fine beds (Walters et al. 2003). Water quality is often altered, with an increase in contaminants (Wenger et al. 2009), inorganic nutrient enrichment (Kaushal 2006), and conductivity (Wenger et al. 2009). The physical and chemical alterations to streams in urbanized watersheds ultimately result in biotic compositions that are lower in diversity of sensitive, endemic species and dominated by tolerant, cosmopolitan species (Scott and Helfman 2001, Walters et al. 2003, Walsh et al. 2005, Wenger et al. 2009). This "biotic homogenization" (Scott and Helfman 2001, Walters 2003, Scott 2006) leads to irreversible losses of species diversity. Species tolerance to urbanization varies, generally with fluvial specialists exhibiting sensitivity and lentic tolerant species exhibiting tolerance (Walters et al. 2003, Roy et al. 2005, Scott and Helfman 200). However, this relationship is not consistent and species with life histories and morphologies similar to sensitive taxa can exhibit tolerance (Wenger 2008). Identifying aspects of tolerant fishes' ecology or behavior that allow them to respond to the habitat alterations resulting from land use changes may help anticipate land use effects on stream fish species before they are impacted and guide management in a more effective way.

Geographic range serves as a good relative measure of a species' risk of imperilment both in terrestrial and aquatic ecosystems (Angermeir 1995). Endemics' small ranges suggest specialist ecologies that depend upon a narrow range of conditions for persistence. Cosmopolitan species, with larger ranges, may be more flexible in habitat requirements. Being a habitat "generalist", however, could not alone result in the trends seen in some tolerant species following land use such as an increase in both presence and abundance (Utz et al. 2010) which entails an expansion of range and colonization of new areas. Albanese et al. (2009) found that mobility was an important determinant of colonization and population recovery. Theoretical models have demonstrated that movement may counterbalance the effects of habitat loss and fragmentation in altered landscapes (Niebuhr 2015) and that characteristics of a population's dispersal tendencies can serve as an indicator of their ability to increase range (Goldwasser et al. 1994, Kot et al. 1996). Dispersal may be a fundamental aspect of fish behavior involved in their ability to tolerate stream systems affected by human land use, particularly for fishes that exhibit tolerance despite specialist ecologies.

Studies have shown that stream habitat complexity, consistently decreased by urban land use in a watershed, can have an impact on movement of fishes. Albanese et al. (2004) observed that *Semotilus corporalis* were less likely to disperse out of more complex habitats. Similarly, pool complexity and area were negatively related to movement for creek chub *Semotilus atromaculatus* (Walker and Adams 2016). This negative correlation with movement and habitat complexity was also observed in Cutthroat trout (Harvey et al. 1999). However, darter species had varying movement response to environmental variability (Roberts and Angermeir 2006). Fishes may use increases in dispersal as a mechanism to respond to changes in habitat complexity.

Empirical dispersal studies of stream fishes consistently result in distributions that best fit a heterogeneous model (Radinger and Wolter 2014, Rodriguez 2002, Skalski and Gilliam 2000) with two components termed the "mobile" and "stationary" components (See "patterns of stream fish dispersal" (pg. 10); Figure 3.1). Individuals grouped into the stationary component primarily display movement associated with home range behaviors while those in the mobile component exhibit long distance dispersal. The same species have been observed to have differing proportions of their population represented in the mobile component in different studies (Radinger and Wolter 2014). This shows that this proportion is not fixed and may be able to change in response to environmental needs. Since urban streams lack the habitat complexity negatively correlated with movement and fishes may need to disperse further to locate suitable habitat, especially if they have specialist ecologies, I hypothesize that:

1. Urban tolerant species will have a greater proportion represented in the mobile component in urban streams than rural streams.
2. The mean movement distances for urban tolerant species will be greater for both the stationary and mobile component in streams that are heavily impacted by urbanization than in less impacted streams
3. Urban tolerant generalists will have a lower proportion of their population represented in the mobile component and mean movement distances for both the stationary and mobile component will be lower for tolerant generalists than urban tolerant specialists

Methods

Site selection

Two sites were selected in the Etowah river basin that contained the species of interest to our study, had similar watershed areas, and varied in the degree to which the watershed area was impacted by urban land use. Total imperviousness (TI) of the watershed area was used as a metric to estimate urban impact. Our less impacted/rural site, Picketts Mill Creek, has a TI of ~6% and is located in Paulding County, Georgia (Figure 3.2, Figure 3.3). It is a second-order tributary of Little Pumpkinvine Creek. Our more impacted/urban site, Little Noonday Creek, has a TI of ~30% and is located in Cobb County, Georgia (Figure 3.2, Figure 3.3). It is a second order tributary of Noonday Creek.

Species selection

Species were selected that would allow comparison of the importance of long distance dispersal for tolerant species in urban and rural streams. In addition, we were interested in comparing habitat generalist and specialist fishes to see the degree to which these ecological differences impact the need for long distance dispersal in streams impacted by urban land use. Two species were selected that fit these criteria: *Campostoma oligolepis* and *Lepomis auritus*.

Campostoma oligolepis, the largescale stoneroller, is a cyprinid present in the Etowah River Basin. It is a documented urban tolerant species (Meador et al. 2005, Roy et al. 2005, South and Ensign 2013) and a habitat specialist that relies upon coarse substrate for feeding on epilithic communities. *C. oligolepis* have been observed to not only persist but thrive in streams impacted by urban land use, with populations in urban streams exhibiting longer spawning seasons, higher GSI values, and shorter gut lengths than populations in less impacted streams (Mutchler et al. 2013, South and Ensign 2013).

Lepomis auritis, the redbreast sunfish, is a centrarchid species present in the Etowah River Basin. It is a well-known urban tolerant generalist that will feed on insects, snails, and other small invertebrates. It is lentic tolerant and prefers low-velocity, deeper habitats.

Given my hypotheses, my predictions are as follows:

1. The proportion of the population represented in the mobile component will be greater in Little Noonday Creek than Picketts Mill Creek for *L. auritus* and *C. oligolepis*.

2. The mean movement distances for the stationary and mobile components of *L. auritus* and *C. oligolepis* will be greater in Little Noonday Creek than in Picketts Mill Creek.
3. The proportion of the population represented in the mobile component of *L. auritus* will be lower than *C. oligolepis* and the mean movement distances for both the stationary and the mobile component of *L. auritus* will be lower than *C. oligolepis*.

Sites were divided into sections that were comprised of a single erosional-depositional sequence with block nets set at the upstream and downstream end of each section prior to sampling via backpack electrofishing. Collected fishes were identified to species with species of interest anesthetized with 140mg/L MS-222 and surgically injected with a 12mm passive integrated transponder (PIT) tag (Biomark, Idaho) in the abdomen posterior to the pelvic fins (Figure 3.4). Tagged fishes had section of capture, lengths, and weights recorded before being allowed to recover fully. Recovered fishes were returned to the midpoint of the section from which they were collected.

In total, 20 sections were sampled in each site. Picketts Mill Creek had an average sampling unit length of 32m and a total sampling reach of 649m. Little Noonday Creek had an average sampling unit length of 37m and a total sampling reach of 740m. 200 *C. oligolepis* and 182 *L. auritus* were tagged in Picketts Mill Creek and 189 *C. oligolepis* and 136 *L. auritus* were tagged in Little Noonday Creek. (Table 3.1)

Resampling

Resampling via a portable antenna connected to an HPR Plus tag reader unit (Biomark, Idaho) was conducted for the full length of each site three times. Little Noonday creek was resampled via antenna in October 2016, January 2017, and February 2017. Picketts Mill Creek was resampled via antenna in September 2016, October 2016, and January 2017. Ten sections at each site were also resampled in December of 2016 via backpack electrofishing. Resampling occurred from downstream to upstream. Block netting was placed at the upstream and downstream ends of a sampling unit to avoid the possibility of movement between sections during sampling, with the downstream block net removed and placed at the downstream boundary of the next section following sampling of each unit.

For antenna recaptures, a “sampling marker” (PIT tag in a centrifuge tube) was scanned with the antenna at the beginning of each unit prior to sweeping the entire section with the antenna. Following the completion of antenna sampling, the “sampling marker” was scanned again. This provided the ability to determine which sections the tag IDs were located in without relying solely upon GPS data or time of detection. Although both of these variables are stored along with detected tag IDS, their accuracy is proven to be suspect in areas of poor satellite reception. Upon detection of a tag, efforts were made to evaluate the possibility of the detection being a shed tag rather than a live, marked individual. Habitats that seemed unlikely for fish to be located were agitated following detection (in an attempt to cause a live fish to flee), then scanned with the antenna again after the unique tag code timeout (set to 30 seconds). Repeated detections despite agitation of the area resulted in the tag ID being deleted from our data set and not used in the calculation of movement distances.

Habitat Analysis

Habitat data were collected in January 2017 across the entire reach of both study sites to evaluate differences in habitat complexity between urban and rural sites. A top-set wading rod and flow meter were used to measure depth and velocity, and a Rosgen Stream Classification scale was used to quantify substrate (Table 3.2). Ten measurements of each variable were taken at random points in each sampling section, with a single measurement of the dominant substrate being recorded at each point of depth/velocity measurement. Since each of our sites consisted of 20 habitat units, 200 total measurements of each variable were recorded for each site.

Measurements from both sites were pooled with habitat scores assigned to each measurement depending on its value. Values in the upper 1/3rd of depths (>1ft) were assigned a 3, between upper 1/3rd-2/3rd were assigned a 2, and the lower 1/3rd (≤.69ft) a 1. For velocity, values in the upper 1/3rd (>.41ft/s) were assigned a 1, between upper 1/3rd-2/3rd assigned a 2, and the lower 1/3rd (≤.1ft/s) a 1 (Table 3.3). Substrate received a 1 for measurement of coarse substrate (anything besides sand, silt, or clay; Table 3.2) and a 2 for fine substrate (sand, silt, or clay). Pivot tables in Excel allowed us to quantify the number of habitat score combinations that occurred in each stream and compare the prevalence of habitat scores or combinations of habitat scores by stream. For example, comparing the prevalence of coarse substrate (all habitat score combinations that had a 1 value for substrate) or “riffle” type habitat (all habitat scores that had a 1 value for velocity and 1 for substrate) among streams.

Analyses of the proportions of habitat types from each stream were conducted that accounted for differences in habitat complexity and availability of preferred habitat types for the species in our study. For *L. auritus*, a lentic-tolerant generalist, deep habitat was considered ideal with all habitat score combinations that had a 3 for depth compared between the two sites. For *C. oligolepis,* a fluvial specialist, we compared the prevalence of “riffle” type habitat score combinations among streams (those that had the highest 1/3rd velocity and coarse substrate). All comparisons were done using a z-test for proportions.

Movement Analysis

Movement distances were calculated from the midpoint of the section of marking/last recapture to the midpoint of the section of recapture. Fishes encountered on multiple recapture events had movement distances for each encounter included as a repeated measure.

Observed movement distances (including those=0m) for each population were analyzed in R package Mclust (Scrucca et al 2016) which uses density estimation via Gaussian finite mixture modeling to estimate the model of best fit for sample data. Mclust returns the best fit model and the number of components in that model, along with log likelihood values and parameter estimates. A two-component model would return estimates of the proportion of both components (corresponding to the proportion of the mobile and stationary components), a mean for each component, and variance. Log-likelihood ratio tests were used to compare the fit of 2-component models with one-component models for each population. Parameter estimates were used from the 2-component model if it fit the data significantly better than the 1-component model (the null model for the likelihood ratio test because it estimates the fewest parameters) for comparison across species and stream sites. Otherwise, one-component model parameter estimates were used.

Results

Habitat Analysis

Picketts Mill had significantly more habitat with a coarse bed than Little Noonday (z-test, p=<.00001). Little Noonday had significantly more “run” habitat, low complexity habitat characterized by shallow water with a fine bed (z-test, p=<.00001). “Riffle” habitat, characterized by high velocity water and a coarse stream bed (the preferred habitat for *C.oligolepis*), was significantly more prevalent in Picketts Mill (z-test, p=.001). The deepest habitat was also found significantly more in Picketts Mill (preferred for *L. auritus*) than Little Noonday (z-test, p=.002). (Table 3.3)

Movement Analysis

In Picketts Mill *L. auritus*, a total of 46 recaptures were included in our movement data with 21 observations of 0 movement and 25 observations of movement >0m. Picketts Mill *C. oligolepis* had 77 total recaptures with 22 observations of 0 movement and 55 observations of movement >0m. Little

Noonday *L. auritus* had 45 total recaptures with 23 observations of 0 movement and 22 observations of movement >0m. Little Noonday *C. oligolepis* had 64 total recaptures with 11 observations of 0 movement and 53 observations of movement >0m. All movement distributions were leptokurtic and non-zero movement observations occurred in the downstream direction more often than the upstream direction for all populations (Table 3.4).

For three out of four of our study populations, the two-group model fit our observed distance data significantly better than a one-group model. *C. oligolepis'* movement distributions fit the two-group model best in both Little Noonday (LR test, p=.00001) and Picketts Mill Creek (LR test, p=.00002). The *L. auritus* population in Little Noonday Creek also fit the two-group model best (LR test, p=.000002), however the population in Picketts Mill did not fit the two-group model significantly better than the one group model (LR test, p=.43). (Table 3.5)

Proportions of the components generated from Mclust indicated that 80% of the *C. oligolepis* population in Little Noonday Creek would be represented in the stationary component and 20% in the mobile component. *L. auritus* in Little Noonday had 86% represented in the stationary component and 14% in the mobile component. 65% of the *C. oligolepis* population in Picketts Mill Creek were represented in the stationary component and 35% in the mobile component. Since the Picketts Mill *L. auritus* population did not fit the two-component model significantly better, the entire population's movement data is a single component (Table 3.6).

Although the proportion of the population represented in the mobile component was greater in Picketts Mill for *C. oligolepis*, mean movement distances for both the stationary components (t-test, p=<.00001) and mobile components (t-test, p=.005) were greater in Little Noonday Creek (Figure 3.5). The stationary component of *C. oligolepis* had an estimated mean movement distance of 38.3m in Little

Noonday and 21m in Picketts Mill. The mobile component of *C. oligolepis* had an estimated mean movement distance of 217.7m in Little Noonday and 131.3m in Picketts Mill (Table 3.6).

The stationary component of Little Noonday's *L. auritus* population does not have a significantly greater mean movement distance than the single component fit for Picketts Mill *L. auritus* (Figure 3.6). *L. auritus* in Little Noonday had a mean movement distance of 21.9m for the stationary component and 193.3 for the mobile component. The mean movement distance for Picketts Mill *L. auritus* was 24.5m (Table 3.6).

In Little Noonday Creek, *C. oligolepis'* stationary component (μ =50.8m) had a significantly higher mean movement distance than the stationary component of *L. auritus* (μ =21.9m) (t-test, p=.00004) although the mobile components were not significantly different (Table 3.6).

In Picketts Mill, *C. oligolepis'* mobile component (μ =131.3m) had a significantly greater mean movement distance than the single component of *L. auritus* (μ =24.5m) (t-test, p=.04) while the stationary component of *C. oligolepis* and the single component fit of *L. auritus* were not significantly different (Table 3.6).

Discussion

Similar to many other stream fish dispersal studies (Gilliam and Fraser 2001, Freeman 1995, Skalski and Gilliam 2000, Walker and Adams 2016) all of our populations' movement distributions were leptokurtic (Table 3.4). The consistent leptokurtosis of movement distributions in fish movement studies is one of the findings that led to the development of the heterogeneous model of fish movement (Radinger and Wolter 2014, Rodriguez 2002, Skalski and Gilliam 2000). However, one of our populations, *L. auritus* in Picketts Mill, deviated from the heterogeneous model and has a movement distribution that is better explained by a homogenous model. This population also has the lowest sample kurtosis value

(4.29) of all four of our populations. While two-component models have been supported in stream fish movement studies (Pépino et al. 2012, Rodriguez 2002, Skalski and Gilliam 2000, Wells et al. 2017), other studies have also found distributions that do not fit this model. Morrissey and Ferguson (2017) observed a population with homogeneous dispersal, likely due to population-wide spawning migration. In a study fitting 27 data sets to both one and two-component models, Rodriguez (2002) found that 10 of the data sets were not better explained by the two-group model.

It is important to consider whether our experimental design could have influenced the lack of a heterogeneous model fit for the movement distribution in Picketts Mill *L. auritus.* Our methodology for re-detections (portable PIT antenna) was less efficient in Picketts Mill than in Little Noonday, most likely due to the significantly greater habitat complexity in Picketts Mill and *L. auritus*' habitat preferences (see results, discussion in Chapter 2). Our recaptures for *L. auritus* in Picketts Mill provided us with only 46 movement observations despite tagging 182 fish (compared to 45 observations of 136 tagged *L. auritus* in Little Noonday). A study by Booth et al. (2014) evaluated the number of recaptures that would maximize estimates of the true movement distribution for a fish population and indicated that observations around our sample size may be able to effectively estimate overall mean distances in movement, but not maximum movement distances. Another issue in our methodology was that our marking and resampling occurred in the same reach area, which biases detections towards short distance movement observations (Albanese 2003, Rodriguez 2002). However, the same methodology and very similar sample size in Little Noonday's *L. auritus* population resulted in a distribution that fit the 2-component model significantly better ($p<.00001$, Table 3.5) than the one-component model.

One complication of the lack of a two-component fit for *L. auritus* is the inability to compare the proportion of the mobile component in Picketts Mill and Little Noonday. Proportions aside, the highest observed movement distance (115m) in Picketts Mill for *L. auritus* was lower than the smallest value in the observed individuals that were grouped into the mobile component for Little Noonday (129m).

Kurtosis, which was used in Lowe (2009) as a metric to compare yearly frequencies of long distance dispersal occurring in a stream-dwelling salamander, was much higher (almost triple) in Little Noonday *L. auritus* than in Picketts Mill *L. auritus* (table 3.4). The mean for the entire population of Picketts Mill *L. auritus* is very similar to the mean for the stationary component in Little Noonday (Table 3.6). This leads me to conclude that there is much more long distance dispersal occurring for *L. auritus* in Little Noonday Creek than in Picketts Mill Creek. This finding aligns with studies that have observed a negative correlation with movement and habitat complexity (Albanese et al. 2004, Harvey et al. 19999, Walker and Adams 2016). Long distances dispersal may be more important in low complexity habitats even for habitat generalists such as *L. auritus*.

The proportions of the population represented in the mobile component for *C. oligolepis* were significantly different, but not in the expected direction. A significantly higher proportion of *C. oligolepis*' population was represented in the mobile component in Picketts Mill, our less impacted site. However, the estimated mean movement distances were significantly lower in Picketts Mill for both the stationary (μ=21.4m Picketts Mill, μ=50.8m Little Noonday) and mobile (μ=131.3m Picketts Mill, μ=217.7m Little Noonday) components. It seems that although fewer fishes are grouped into the mobile component, the entire population of *C. oligolepis* in Little Noonday is moving significantly longer distances. It may be that increased movement is necessary simply for *C. oligolepis* to survive in Little Noonday Creek. Since they are foragers that feed on epilithic communities and Little Noonday has significantly less coarse substrate than Picketts Mill, locating suitable foraging may require much more movement for the stationary component. Individuals dispersing long distances have much more low complexity habitat to traverse in Little Noonday, which has significantly more run-type habitat than Picketts Mill. These long stretches of unsuitable habitat are likely causing the significant increase in long distance dispersal distances in the mobile component because fish have to travel much further to reach preferred habitat. This aligns with the findings of Harvey et al. (1999), who saw the most extensive movements in environments lacking structural complexity.

In Little Noonday, *C. oligolepis* had a greater proportion of their population represented in the mobile component (20%) than *L. auritus* (14%), but the movement distances of the mobile components were not significantly different. Both of these species are having to exhibit similar long distance movement distances to traverse the low complexity habitat of Little Noonday. However, the mean movement distances for the stationary components were significantly greater in Little Noonday *C. oligolepis* than *L. auritus*, likely due to the increased distance that *C. oligolepis* has to move to locate suitable areas to forage on epilithic communities. *L. auritus'* generalist ecology is likely what allows it to persist in Little Noonday without increased stationary component (home range) movement. For *C. oligolepis*, however, locating coarse substrate in an environment dominated by fine substrate, as in Little Noonday, requires increased home range movement.

In Picketts Mill, the stationary component for *C. oligolepis* did not have a greater mean movement distance than the single component of *L. auritus.* The mobile component for *C. oligolepis* did have a significantly greater mean movement distance than the single component of *L. auritus*, which was unsurprising considering that the mean movement distance for *L. auritus* in Picketts Mill was only 24.5m and is more comparable to a single stationary component. Dispersal may be more important overall for *C. oligolepis* since it has to move around to find suitable grazing wherever epilithic communities are available.

In conclusion, long distance dispersers for tolerant generalists and specialists are likely to move longer distances in low complexity urban streams. The increase in both home range and long distance dispersal distances may be important for specialists in order to persist in low complexity urban habitat. The proportion of the mobile component is not fixed for species but may not be a good metric alone for comparing dispersal among different sites and should be analyzed along with mean movement distances for those components. Additionally, while there is a lot of support for the heterogeneous model of stream fishes, the fit of this model should not be assumed. Had we fit all of our data to a 2-component model without first assessing model fit, the means for the stationary and mobile component of Picketts Mill *L. auritus* would have been less than 50m in difference, which does not fit with the idea of stationary and mobile groups in a population (Rodriguez 2002). Deviations from the heterogeneous model may be rare, but some species may be less likely to exhibit long distance dispersal based on their habitat preferences (e.g.- pool-dwelling species with access to high complexity habitat).

Integration of Thesis Research

This research project spans across multiple scales of biology by examining how landscape-level modifications affect population-scale movement distributions. The interconnectedness of natural environments promotes the integration of multiple-scale approaches when considering the effects that alterations of natural environments may have on species. Examining natural environments at only one scale may lead to overlooking crucial elements that may be influencing a study.

This research also relied heavily upon the integration of disciplines outside of biology, namely statistics and mathematical modeling. The findings of this research provide some evidence that increasing dispersal is important for tolerant species in degraded environments. Continuing to analyze dispersal of both tolerant and sensitive species using similar methods may provide a more rigorous, quantifiable means of analyzing the effects of urbanization on stream fishes. Integrating statistics and mathematical modeling into biological studies allows for studies conducted in nature to maintain rigor despite the inability to control many variables of your study environment.

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Figures

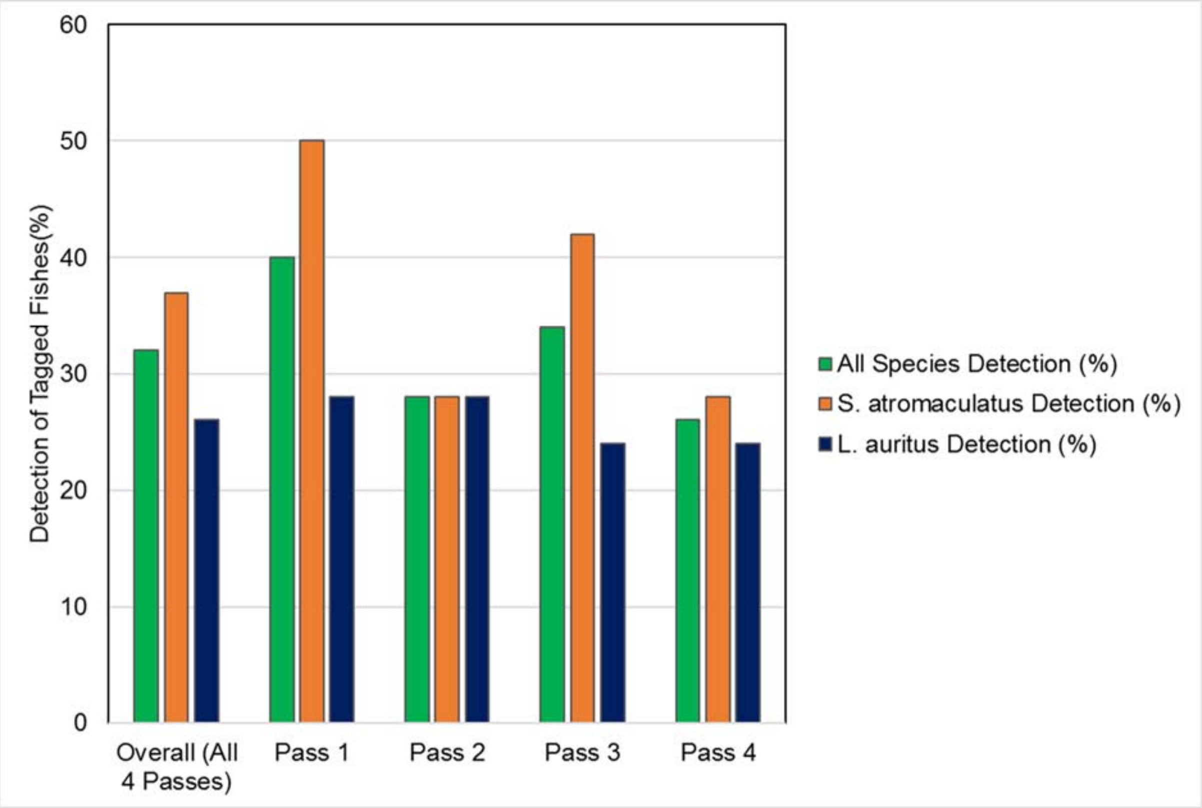


Figure 2.1- Frequency (%) of PIT-tagged fish in Kennesaw Creek detected by four passes of a portable antenna. These frequencies were calculated directly by dividing the number of detected PIT-tagged fishes by the number of known tagged fishes in the reach (movement outside the reach was prevented by block netting). Detection frequencies are given for *S. atromaculatus* (orange), *L. auritus* (blue), and both species combined (green).

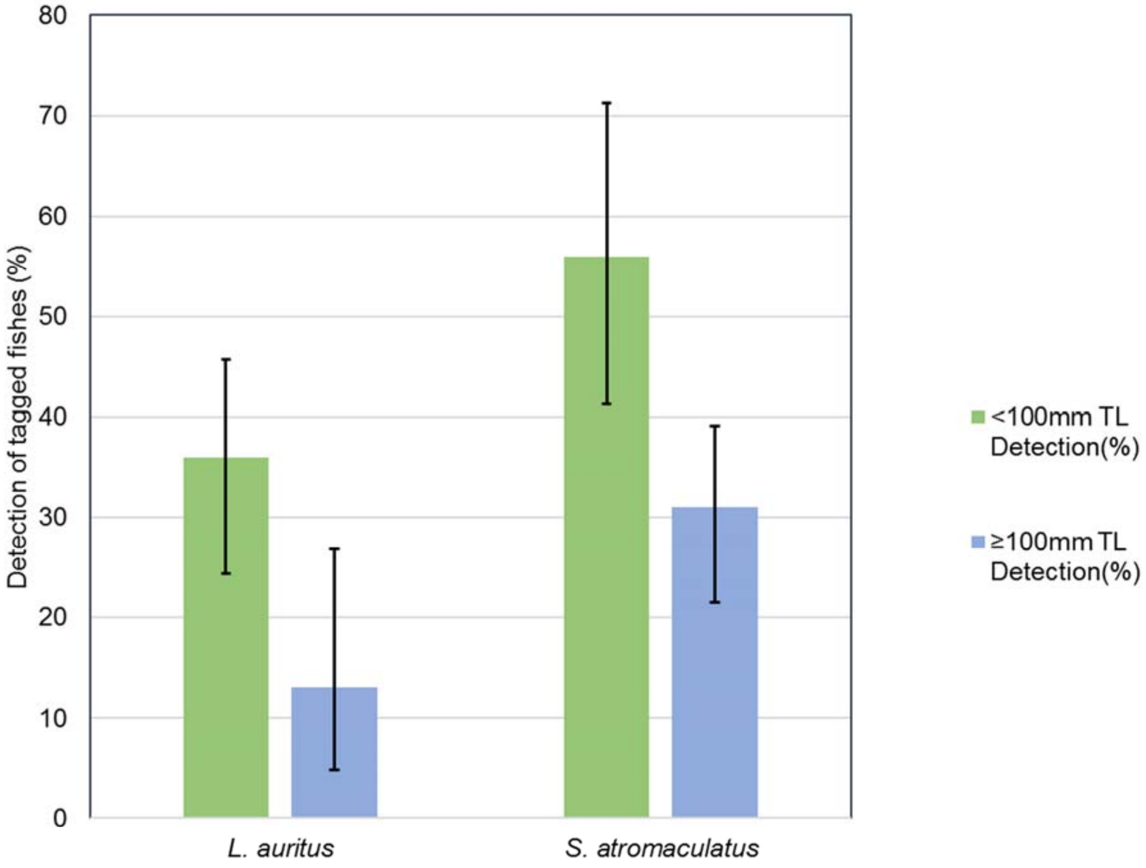


Figure 2.2- Portable PIT antenna detection(%) for two size classes of *Lepomis auritus* (left) and *Semotilus atromaculatus* (right) in Kennesaw Creek. Size classes were grouped by fishes with a measured total length of <100mm (green) and ≥100mm TL (blue). Mean detection estimation was conducted in program MARK and matched detection frequencies that were calculated by dividing the number of detected PIT tags by the number of known tagged fishes. Error bars= 95% confidence intervals derived from detection frequency estimation in program MARK.

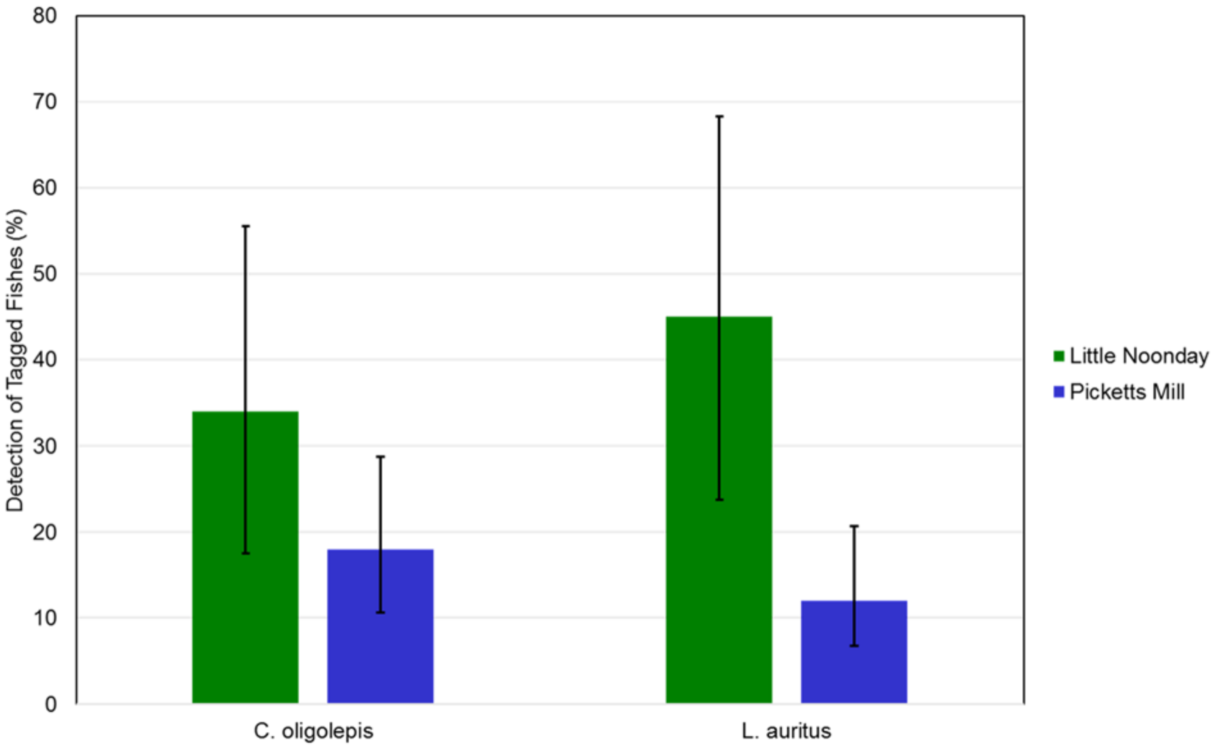


Figure 2.3- Portable PIT antenna detection probability estimates for *C. oligolepis* (left) and *L. auritus* (right) from a 7-month mark-recapture study conducted in Little Noonday Creek (green) and Picketts Mill Creek (blue). Estimates of detection were derived from program MARK which returned a constant detection probability between sampling occasions as the model of best fit. Error bars=95% confidence intervals.

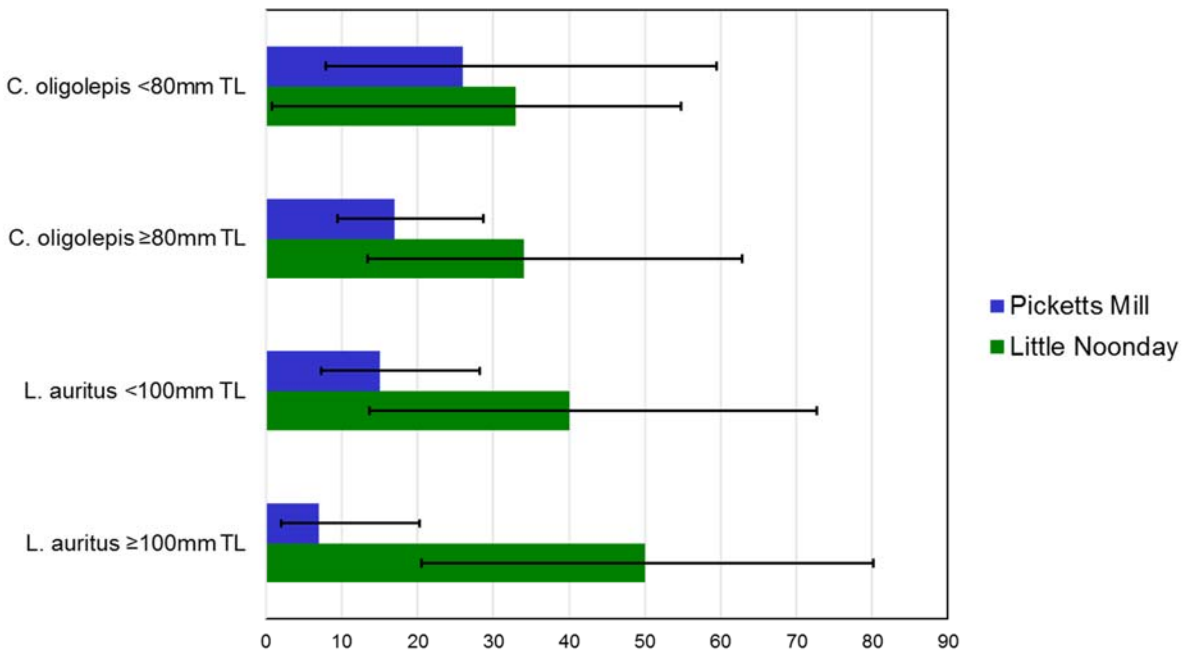


Figure 2.4- Portable PIT antenna detection probability (%) of tagged fishes for size classes of *C. oligoelpis* and *L. auritus* in Picketts Mill Creek (blue) and Little Noonday Creek (green). Data was derived from a 7-month mark-recapture study. Size classes for *C. oligolepis* were grouped by fish with a measured TL of <80mm (row 1) and ≥ 80mm (row 2). Size classes for *L. auritus* were grouped by fish with a measured TL of <100mm (row 3) and ≥100mm TL (row 4). Estimates of detection and confidence intervals were calculated in program MARK which returned a constant probability of detection across sampling occasions as the model of best fit. Error bars=95% confidence intervals.

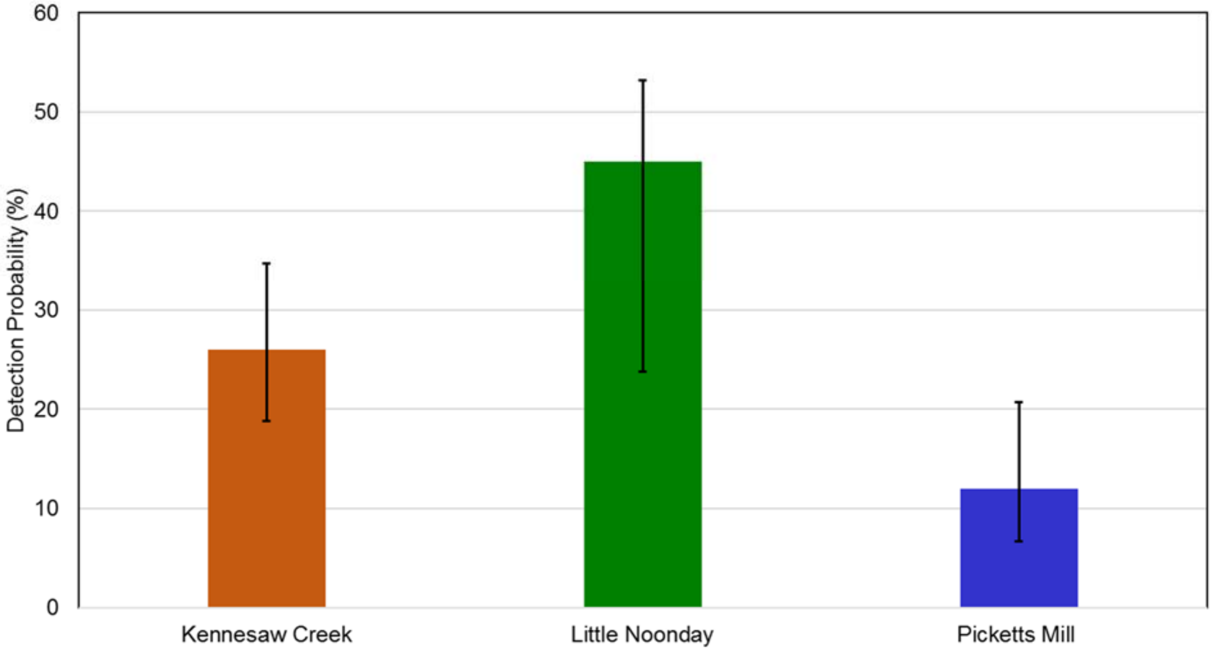


Figure 2.5- *L. auritus* detection probabilities (%) with a portable PIT antenna in three streams: Kennesaw Creek (left, orange), Little Noonday Creek (middle, green), and Picketts Mill Creek (right, blue). Detection probabilities and confidence intervals were estimated with program MARK which returned a constant detection between sampling occasions as the model of best fit. Kennesaw Creek was sampled with the portable antenna 4 times and Little Noonday and Picketts Mill were both sampled with the portable antenna three times (the full reach was sampled for each site). Error bars=95% confidence intervals.

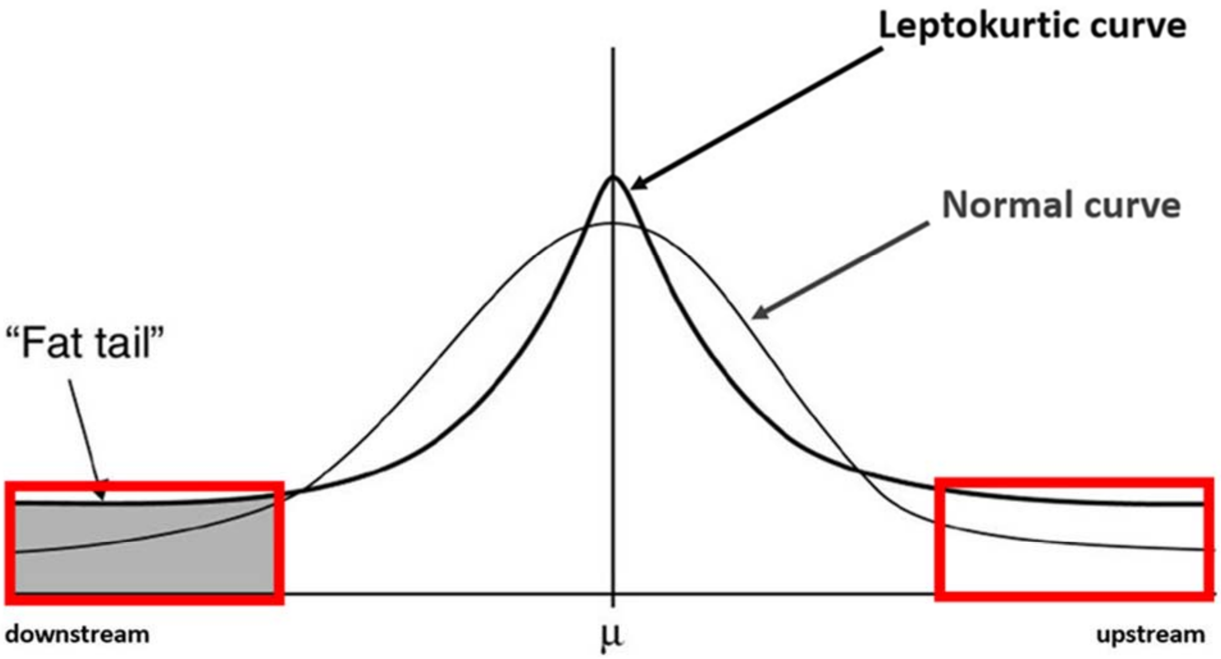


Figure 3.1- Visual representation of a typical leptokurtic curve (with a normal curve for reference) of the probability for an individual (y-axis) distributing a distance (x-axis, "+" indicates upstream, "-" indicates downstream) for a stream fish population. The mobile component is represented by the red boxes surrounding the "fat tails" of the leptokurtic curve. The stationary component presents as a tall peak in the middle and is related to home range behavior.

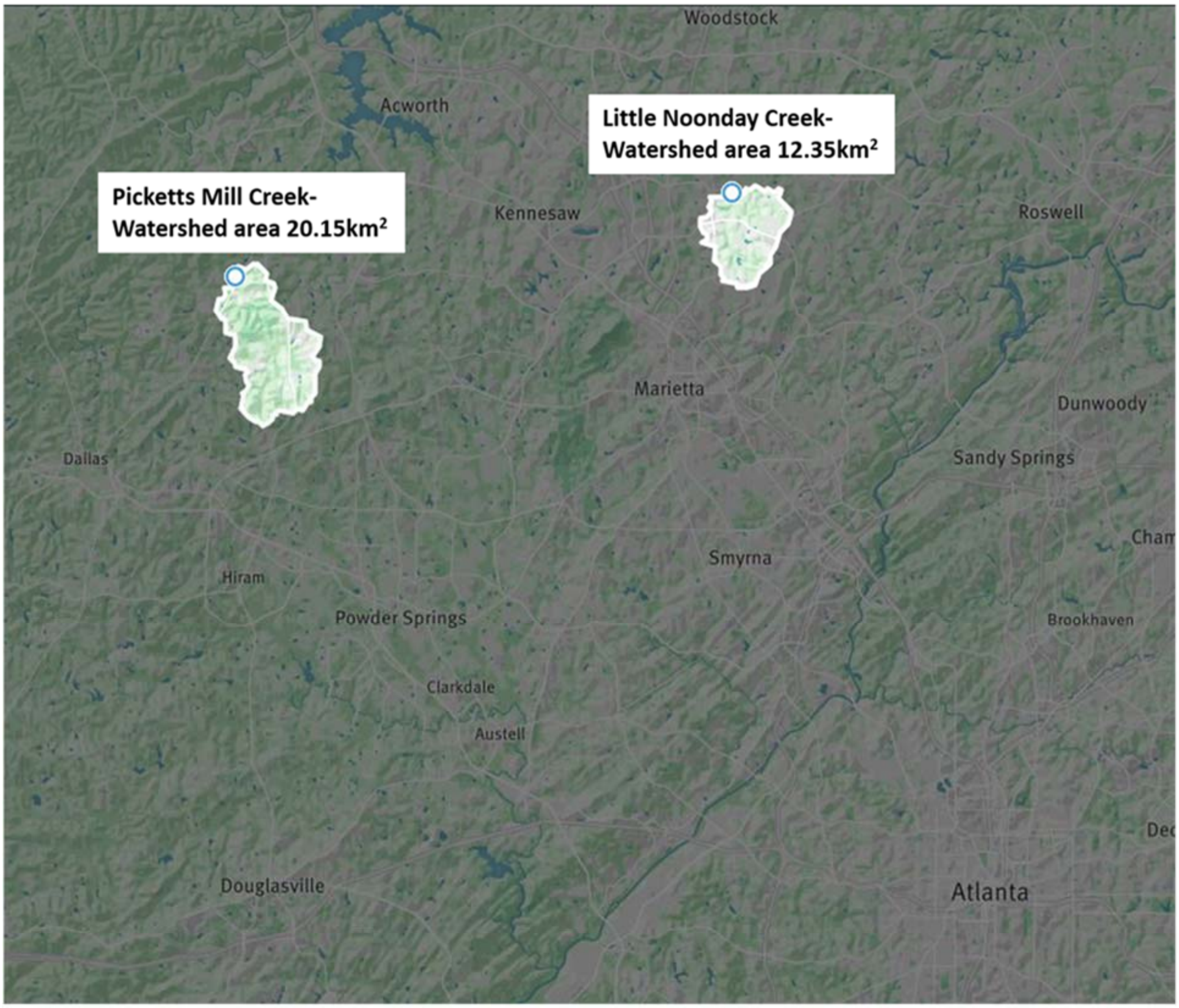


Figure 3.2- Map showing the watershed area and location of the two study sites containing the populations of *L. auritus* and *C. oligolepis* that were observed in our 7 month mark-recapture study. Picketts Mill Creek (left) is located in Paulding County, Georgia. Little Noonday Creek (right) is located in Cobb County, Georgia. Urban land use differs in these two watersheds with total imperviousness (TI) of Picketts Mill's watershed ~6% and the TI of Little Noonday's watershed ~30%.

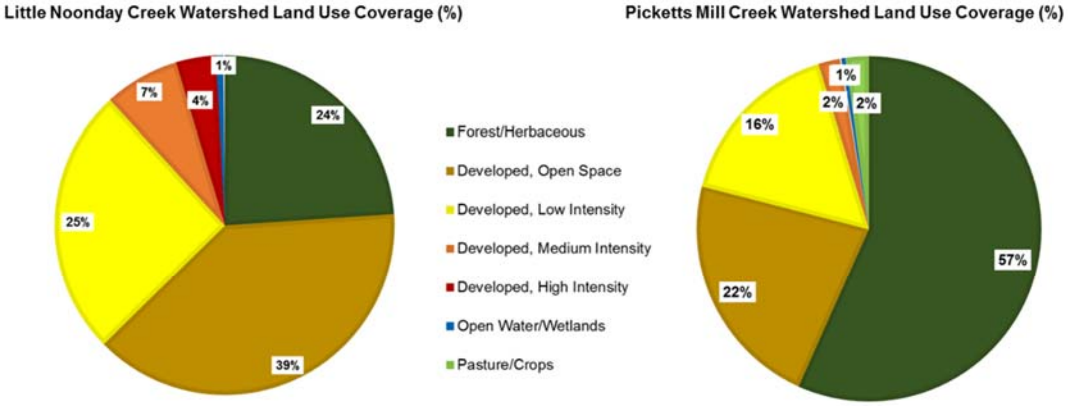


Figure 3.3- Pie charts showing the differences in land use coverage (%) for the watersheds of the two study sites where I observed movement of *L. auritus* and *C. oligolepis* in a 7-month mark-recapture study using passive integrated transponder (PIT) telemetry. Little Noonday Creek (left, watershed area= 12.35km^2) has much more total imperviousness (TI) than Picketts Mill (right, watershed area= 20.15km^2) resulting from increased urbanization in its watershed. TI of Little Noonday is ~30% while TI of Picketts Mill is ~6%. These differences in TI result from a greater percentage of developed area and a reduction of forested area in Little Noonday Creek's watershed when compared to the watershed of Picketts Mill Creek.

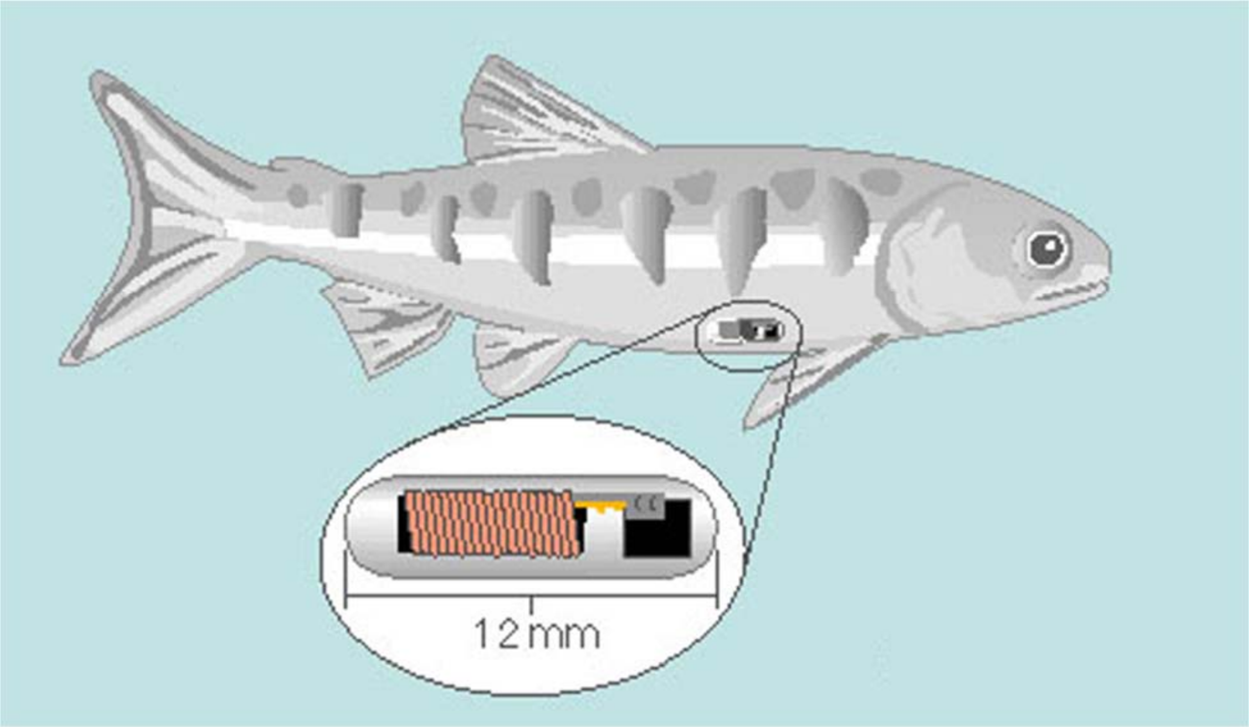


Figure 3.4- Picture showing the site where 12mm passive integrated transponder (PIT) tags (Biomark, Idaho) were inserted surgically via an injection gun in *C. oligolepis* and *L. auritus* for a 7-month mark-recapture study in Little Noonday Creek (Cobb County, Georgia) and Picketts Mill Creek (Paulding County, Georgia).

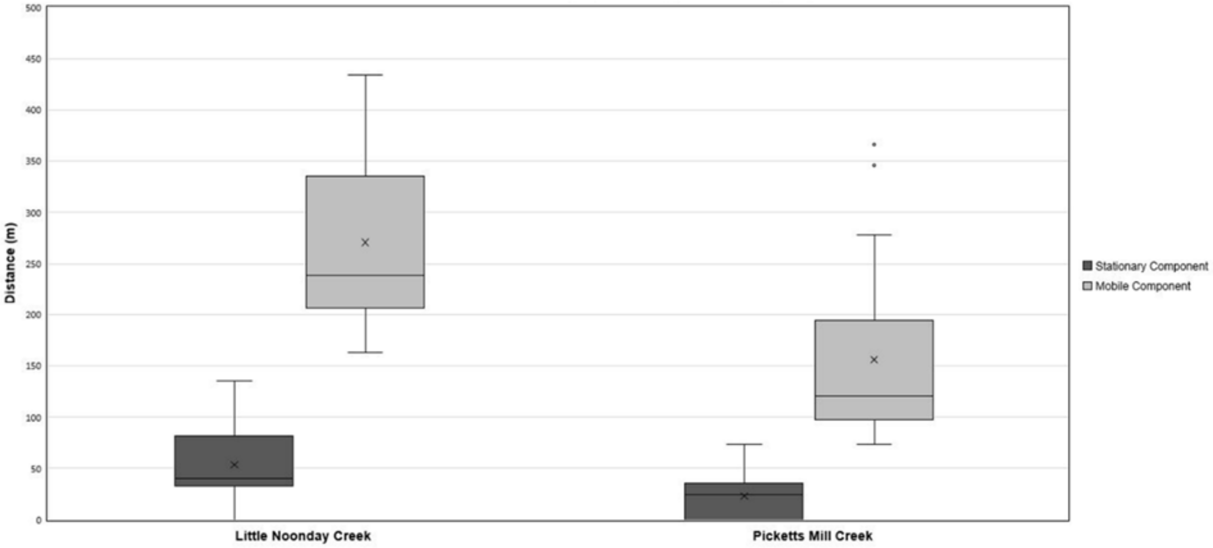


Figure 3.5- Box and Whisker Plot for movement distance estimates of the stationary (dark gray) and mobile (light gray) components of *C. oligolepis* in Little Noonday Creek (left) and Picketts Mill Creek (right). Proportions of the mobile component were .2 in Little Noonday and .35 in Picketts Mill. All proportions, mean movement estimates, and SD were calculated in R package Mclust.

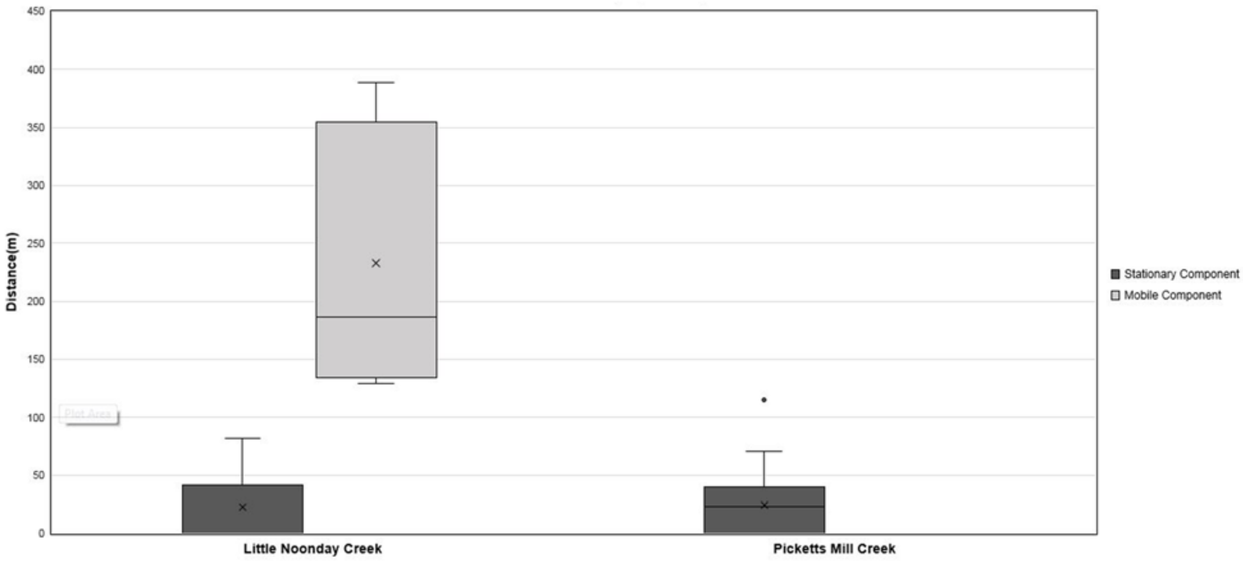
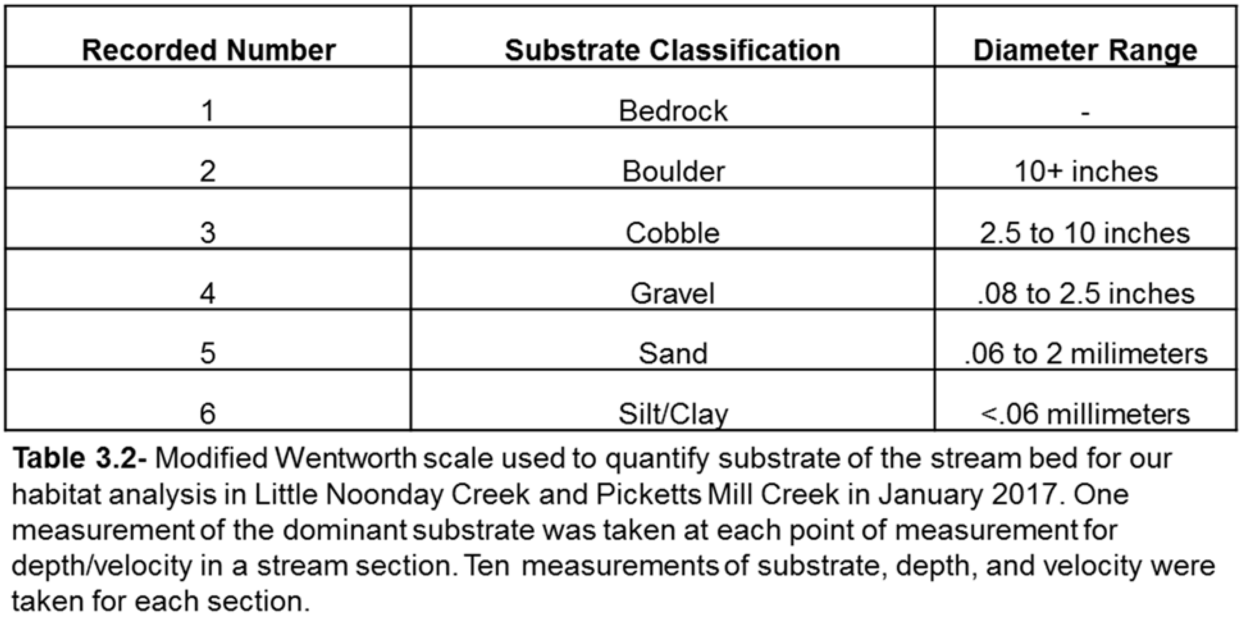


Figure 3.6- Box and Whisker plot for the movement distances of the stationary (dark gray) and mobile (light gray) component of *L. auritus* in Little Noonday Creek (left) and the movement distances of all *L. auritus* in Picketts Mill Creek (right). The proportion of the mobile component was .14 for Little Noonday Creek. Picketts Mill Creek's movement data did not fit a two-component model better than a one-component model based on a likelihood ratio test. All model fit, proportions, means, and SD were calculated used R package Mclust.

Tables

Site	Total Reach Length	Average length of sampling units	Number of *C. oligolepis* tagged	Number of *L. auritus* tagged
Picketts Mill Creek (Rural Site)	649m	32m (SD=11.7m)	200	182
Little Noonday Creek (Urban Site)	740m	37m (SD=6.5)	189	136

Table 3.1- Reach and section lengths for the two sites sampled in our 7-month mark-recapture study and the number of *C. oligolepis* and *L. auritus* tagged with 12mm passive integrated transponder (PIT) tags in each reach. Picketts Mill and Little Noonday differ in the urbanization of their watershed. Picketts Mill, the less impacted or “rural” site has a total imperviousness (TI) of ~6%. Little Noonday, the “urban” site, has a TI of ~30%. Sampling units were divided at habitat intersections (e.g. pool/riffle) and were block netted at the upstream and downstream ends when sampling via backpack electrofishing (for initial tagging) and when resampling units with a portable PIT antenna.



Recorded Number	Substrate Classification	Diameter Range
1	Bedrock	-
2	Boulder	10+ inches
3	Cobble	2.5 to 10 inches
4	Gravel	.08 to 2.5 inches
5	Sand	.06 to 2 milimeters
6	Silt/Clay	<.06 millimeters

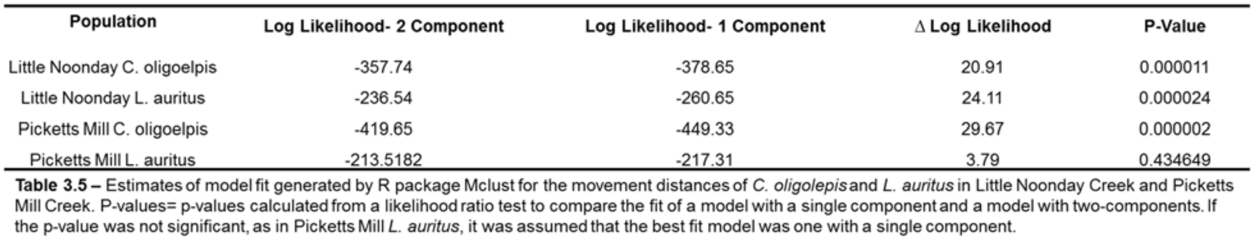
Table 3.2- Modified Wentworth scale used to quantify substrate of the stream bed for our habitat analysis in Little Noonday Creek and Picketts Mill Creek in January 2017. One measurement of the dominant substrate was taken at each point of measurement for depth/velocity in a stream section. Ten measurements of substrate, depth, and velocity were taken for each section.

Habitat Type	Proportion Little Noonday	Proportion Picketts Mill	Z-Score	P-Value
Coarse Stream Bed	0.44	0.74	-5.88344	<.00001
Deep area	0.26	0.36	-3.06337	0.002191
Sandy Runs	0.22	0.05	6.778753	<.00001
Coarse Riffles	0.22	0.32	-3.23746	0.001208

Table 3.3- Overview of the habitat analysis conducted to quantify differences in the occurrence of habitat types between our two sites, Little Noonday Creek and Picketts Mill Creek. Coarse stream bed was characterized as any substrate measurement that was not silt, sand, or clay. Deep area was calculated by comparing the prevalence of the deepest 1/3rd (>1ft) of depth measurements that occurred in either stream. Sandy runs were considered any habitat that had the shallowest 1/3rd (>.41 ft/s) of depth measurements along with a fine bed (silt, sand, or clay) substrate measurement. Coarse riffles were considered any habitat that had the highest 1/4rd of velocity measurement on a coarse substrate measurement.

Species	Site	Movement Observations (0m)	Kurtosis	Upstream	Downstream	Max Distance
Campostoma oligolepis	Little Noonday	53(11)	7.57	20%	63%	434.3m
Campostoma oligolepis	Picketts Mill	55(22)	7.96	23%	48%	366.2m
Lepomis auritus	Little Noonday	23(22)	12.59	16%	33%	388.5m
Lepomis auritus	Picketts Mill	25(21)	4.29	24%	33%	115.1m

Table 3.4- Overview of the movement observations for *C. oligolepis* and *L. auritus* in Little Noonday Creek and Picketts Mill Creek. Number of movement observations (fishes redetected in the same sampling unit with movement of 0 in parentheses) includes repeated measures for fishes recaptured on more than one sampling occasion. Kurtosis= sample kurtosis calculated from observed data. The movement observations not accounted for in the upstream and downstream movement percentages were 0m.



Population	Log Likelihood- 2 Component	Log Likelihood- 1 Component	∆ Log Likelihood	P-Value
Little Noonday C. oligoelpis	-357.74	-378.65	20.91	0.000011
Little Noonday L. auritus	-236.54	-260.65	24.11	0.000024
Picketts Mill C. oligoelpis	-419.65	-449.33	29.67	0.000002
Picketts Mill L. auritus	-213.5182	-217.31	3.79	0.434649

Table 3.5 – Estimates of model fit generated by R package Mclust for the movement distances of *C. oligolepis* and *L. auritus* in Little Noonday Creek and Picketts Mill Creek. P-values= p-values calculated from a likelihood ratio test to compare the fit of a model with a single component and a model with two-components. If the p-value was not significant, as in Picketts Mill *L. auritus*, it was assumed that the best fit model was one with a single component.

Population	Component	Proportion	Mean(m)	Standard Deviation
Little Noonday *C. oligoelpis*	Stationary	0.8	50.819	38.3
	Mobile	0.2	217.693	109.7
Little Noonday *L. auritus*	Stationary	0.86	21.912	28.4
	Mobile	0.14	193.245	119
Picketts Mill *C. oligoelpis*	Stationary	0.65	21.376	21
	Mobile	0.35	131.29	90.1
Picketts Mill *L. auritus*	Single	1	24.495	27.3

Table 3.6- Summary of the estimated mobile and stationary components for the four populations in our 7-month mark-recapture study. Proportions, mean, and SD were calculated in R package Mclust which determines the model of best fit for sample data. Likelihood ratio tests were used to determine if two-component models fit the data significantly better than one-component models. In all cases except for Picketts Mill *L. auritus*, two-component models provided a significantly better explanation of the observed movement data for our study.

References

References

Acolas, M. L., Roussel, J. M., Lebel, J. M., & Bagliniere, J. L. (2007). Laboratory experiment on survival, growth and tag retention following PIT injection into the body cavity of juvenile brown trout (salmo trutta). *Fisheries Research, 86*(2-3), 280-284.

Albanese, B., Angermeier, P. L., & Gowan, C. (2003). Designing mark–recapture studies to reduce effects of distance weighting on movement distance distributions of stream fishes. *Transactions of the American Fisheries Society, 132*(5), 925-939.

Albanese, B., Angermeier, P. L., & Dorai-Raj, S. (2004). Ecological correlates of fish movement in a network of virginia streams. *Canadian Journal of Fisheries and Aquatic Sciences, 61*(6), 857-869.

Albanese, B., Angermeier, P. L., & Peterson, J. T. (2009). Does mobility explain variation in colonisation and population recovery among stream fishes? *Freshwater Biology, 54*(7), 1444-1460.

Alldredge, P., Gutierrez, M., Duvernell, D., Schaefer, J., Brunkow, P., & Matamoros, W. (2011). Variability in movement dynamics of topminnow (fundulus notatus and F. olivaceus) populations. *Ecology of Freshwater Fish, 20*(4), 513-521.

Angermeir, P. (1995). Ecological attributes of extinction-prone species - loss of fresh-water fishes of virginia. *Conservation Biology, 9*(1), 143-158.

Banish, N. P., Burdick, S. M., & Moyer, K. R. (2016). Efficiency of portable antennas for detecting passive integrated transponder tags in stream-dwelling salmonids. *PloS One, 11*(2), e0149898.

Benton, P. D., Ensign, W. E., & Freeman, B. J. (2008). The effect of road crossings on fish movements in small etowah basin streams. *Southeastern Naturalist, 7*(2), 301-310.

Bliss, S., Midwood, J., Stamplecoskie, K., & Cooke, S. (2015). Seasonal movements and residency of small-bodied fish in a north temperate urban watershed demonstrate connectivity between a stream and stormwater drain. *Hydrobiologia, 742*(1), 327-338.

Bolland, J., Cowx, I., & Lucas, M. (2009). Evaluation of VIE and PIT tagging methods for juvenile cyprinid fishes. *Journal of Applied Ichthyology, 25*(4), 381-386.

Booth, M. T., Flecker, A. S., & Hairston, N. G., Jr. (2014). Is mobility a fixed trait? Summer movement patterns of catostomids using PIT telemetry. *Transactions of the American Fisheries Society, 143*(4), 1098-1111.

Breen, M. J., Ruetz, C. R., Thompson, K. J., & Kohler, S. L. (2009). Movements of mottled sculpins (cottus bairdii) in a michigan stream: How restricted are they? *Canadian Journal of Fisheries and Aquatic Sciences, 66*(1), 31-41.

Burns, M. J., Walsh, C. J., Fletcher, T. D., Ladson, A. R., & Hatt, B. E. (2015). A landscape measure of urban stormwater runoff effects is a better predictor of stream condition than a suite of hydrologic factors. *Ecohydrology, 8*(1), 160-171.

Coombs, M. F., & Rodriguez, M. A. (2007). A field test of simple dispersal models as predictors of movement in a cohort of lake-dwelling brook charr. *Journal of Animal Ecology, 76*(1), 45-57.

Crook, D. (2004). Is the home range concept compatible with the movements of two species of lowland river fish? *Journal of Animal Ecology, 73*(2), 353-366.

Cucherousset, J., Britton, J., Beaumont, W., Nyqvist, M., Sievers, K., & Gozlan, R. (2010). Determining the effects of species, environmental conditions and tracking method on the detection efficiency of portable PIT telemetry. *Journal of Fish Biology, 76*(4), 1039-1045.

Cucherousset, J., Marty, P., Pelozuelo, L., & Roussel, J. (2009). Portable PIT detector as a new tool for non-disruptively locating individually tagged amphibians in the field: A case study with pyrenean brook salamanders (calotriton asper). *Wildlife Research, 35*(8), 780-787.

Cucherousset, J., Roussel, J., Keeler, R., Cunjak, R. A., & Stump, R. (2005). The use of two new portable 12-mm PIT tag detectors to track small fish in shallow streams. *North American Journal of Fisheries Management, 25*(1), 270-274.

Davies, P. J., Wright, I. A., Findlay, S. J., Jonasson, O. J., & Burgin, S. (2010). Impact of urban development on aquatic macroinvertebrates in south eastern Australia: Degradation of in-stream habitats and comparison with non-urban streams. *Aquatic Ecology, 44*(4), 685-700.

Fraser, D. F., Gilliam, J. F., Albanese, B. W., & Snider, S. B. (2006). Effects of temporal patterning of predation threat on movement of a stream fish: Evaluating an intermediate threat hypothesis. *Environmental Biology of Fishes, 76*(1), 25-35.

Freeman, M. C. (1995). Movements by two small fishes in a large stream. *Copeia*, 361-367.

Gilliam, J. F., & Fraser, D. F. (2001). Movement in corridors: Enhancement by predation threat, disturbance, and habitat structure. *Ecology, 82*(1), 258-273.

Goldwasser, l., Cook, j., & Silverman, e. (1994). The effects of variability on metapopulation dynamics and rates of invasion. *Ecology, 75*(1), 40-47.

Gowan, C., & Fausch, K. D. (2002). Why do foraging stream salmonids move during summer? *Ecology, behaviour and conservation of the charrs, genus salvelinus* (pp. 139-153) Springer.

Harvey, B. C., Nakamoto, R. J., & White, J. L. (1999). Influence of large woody debris and a bankfull flood on movement of adult resident coastal cutthroat trout (oncorhynchus clarki) during fall and winter.*Canadian Journal of Fisheries and Aquatic Sciences, 56*(11), 2161-2166.

Holt, D. E., Jelks, H. L., & Jordan, F. (2013). Movement and longevity of imperiled okaloosa darters (etheostoma okaloosae). *Copeia,* (4), 653-659.

Kaushal, S., Lewis, W., & McCutchan, J. (2006). Land use change and nitrogen enrichment of a rocky mountain watershed. *Ecological Applications, 16*(1), 299-312.

Keeler, R. A., Breton, A., Peterson, D. P., & Cunjak, R. A. (2007). Apparent survival and detection estimates for PIT-tagged slimy sculpin in five small new brunswick streams. *Transactions of the American Fisheries Society, 136*(1), 281-292.

Kemp, S. (2014). The potential and limitations of linking biological monitoring data and restoration needs of urbanized waterways: A case study. *Environmental Monitoring and Assessment, 186*(6), 3859-3873.

Kot, M., Lewis, M., & vandenDriessche, P. (1996). Dispersal data and the spread of invading organisms. *Ecology, 77*(7), 2027-2042.

Labbe, T., & Fausch, K. (2000). Dynamics of intermittent stream habitat regulate persistence of a threatened fish at multiple scales. *Ecological Applications, 10*(6), 1774-1791.

Linnansaari, T., Roussel, J., Cunjak, R. A., & Halleraker, J. H. (2007). Efficacy and accuracy of portable PIT-antennae when locating fish in ice-covered streams. *Hydrobiologia, 582*(1), 281-287.

Lowe, W. H. (2009). What drives long-distance dispersal? A test of theoretical predictions. *Ecology, 90*(6), 1456-1462.

Meador, M., Coles, J., & Zappia, H. (2005). Fish assemblage responses to urban intensity gradients in contrasting metropolitan areas: Birmingham, alabama and boston, massachusetts. *Effects of Urbanization on Stream Ecosystems, 47*, 409-423.

Meyer, J., Paul, M., & Taulbee, W. (2005). Stream ecosystem function in urbanizing landscapes. *Journal of the North American Benthological Society, 24*(3), 602-612.

Morrissey, M. B., & Ferguson, M. M. (2011). Individual variation in movement throughout the life cycle of a stream-dwelling salmonid fish. *Molecular Ecology, 20*(2), 235-248.

Moshenko, R. W., & Gee, J. H. (1973). Diet, time and place of spawning, and environments occupied by creek chub (semotilus atromaculatus) in the mink river, manitoba. *Journal of the Fisheries Board of Canada, 30*(3), 357-362.

Mutchler, T., Ensign, W. E., & Yates, C. C. (2014). Differences in gut morphology of campostoma oligolepis populations from two watersheds in northwest georgia, USA. *Journal of Freshwater Ecology, 29*(2), 289-293

Niebuhr, B. B. S., Wosniack, M. E., Santos, M. C., Raposo, E. P., Viswanathan, G. M., da Luz, M. G. E., & Pie, M. R. (2015). Survival in patchy landscapes: The interplay between dispersal, habitat loss and fragmentation. *Scientific Reports, 5*, 11898.

O'Donnell, M. J., Horton, G. E., & Letcher, B. H. (2010). Use of portable antennas to estimate abundance of PIT-tagged fish in small streams: Factors affecting detection probability. *North American Journal of Fisheries Management, 30*(2), 323-336.

Pepino, M., Rodrigues, M. A., & Magnan, P. (2012). Fish dispersal in fragmented landscapes: A modeling framework for quantifying the permeability of structural barriers. *Ecological Applications, 22*(5), 1435-1445.

Pepino, M., Rodriguez, M. A., & Magnan, P. (2015). Shifts in movement behavior of spawning fish under risk of predation by land-based consumers. *Behavioral Ecology, 26*(4), 996-1004.

Radinger, J., & Wolter, C. (2014). Patterns and predictors of fish dispersal in rivers. *Fish and Fisheries, 15*(3), 456-473.

Rehage, J., & Sih, A. (2004). Dispersal behavior, boldness, and the link to invasiveness: A comparison of four gambusia species. *Biological Invasions, 6*(3), 379-391.

Richard, A., O'Rourke, J., Caudron, A., & Cattanéo, F. (2013). Effects of passive integrated transponder tagging methods on survival, tag retention and growth of age-0 brown trout. *Fisheries Research, 145*, 37-42.

Roberts, J. H., & Angermeier, P. L. (2007). Spatiotemporal variability of stream habitat and movement of three species of fish. *Oecologia, 151*(3), 417-430.

Rodríguez, M. A. (2002). Restricted movement in stream fish: The paradigm is incomplete, not lost. *Ecology, 83*(1), 1-13.

Roy, A. H., Freeman, M. C., Freeman, B. J., Wenger, S. J., Ensign, W. E., & Meyer, J. L. (2005). Investigating hydrologic alteration as a mechanism of fish assemblage shifts in urbanizing streams. *Journal of the North American Benthological Society, 24*(3), 656-678.

Schaefer, J. (2001). Riffles as barriers to interpool movement by three cyprinids (notropis boops, campostoma anomalum and cyprinella venusta). *Freshwater Biology, 46*(3), 379-388.

Scott, M. (2006). Winners and losers among stream fishes in relation to land use legacies and urban development in the southeastern US. *Biological Conservation, 127*(3), 301-309.

Scott, M., & Helfman, G. (2001). Native invasions, homogenization, and the mismeasure of integrity of fish assemblages. *Fisheries, 26*(11), 6-15.

Scrucca, L., Fop, M., Murphy, T. B., & Raftery, A. E. (2016). Mclust 5: Clustering, classification and density estimation using gaussian finite mixture models. *The R Journal, 8*(1), 289-317.

Skalski, G., & Gilliam, J. (2000). Modeling diffusive spread in a heterogeneous population: A movement study with stream fish. *Ecology, 81*(6), 1685-1700.

Slavik, O., Bartos, L., & Mattas, D. (2005). Does stream morphology predict the home range size in burbot? *Environmental Biology of Fishes, 74*(1), 89-98.

Sloat, M. R., Baker, P. F., & Ligon, F. K. (2011). Estimating habitat-specific abundances of PIT-tagged juvenile salmonids using mobile antennas: A comparison with standard electrofishing techniques in a small stream. *North American Journal of Fisheries Management, 31*(5), 986-993.

South, E. J., & Ensign, W. E. (2013). Life history of campostoma oligolepis (largescale stoneroller) in urban and rural streams. *Southeastern Naturalist, 12*(4), 781-789.

Steuer, J. J., Stensvold, K. A., & Gregory, M. B. (2010). Determination of biologically significant hydrologic condition metrics in urbanizing watersheds: An empirical analysis over a range of environmental settings. *Hydrobiologia, 654*(1), 27-55. doi:10.1007/s10750-010-0362-0

Utz, R. M., Hilderbrand, R. H., & Raesly, R. L. (2010). Regional differences in patterns of fish species loss with changing land use. *Biological Conservation, 143*(3), 688-699.

Walsh, C. J., Roy, A. H., Feminella, J. W., Cottingham, P. D., Groffman, P. M., & Morgan, R. P. (2005). The urban stream syndrome: Current knowledge and the search for a cure. *Journal of the North American Benthological Society, 24*(3), 706-723.

Walker, R. H., & Adams, G. L. (2016). Ecological factors influencing movement of creek chub in an intermittent stream of the Ozark Mountains, Arkansas. *Ecology of Freshwater Fish, 25*(2), 190-202.

Walters, D., Leigh, D., & Bearden, A. (2003). Urbanization, sedimentation, and the homogenization of fish assemblages in the etowah river basin, USA. *Hydrobiologia, 494*(1-3), 5-10.

Wang, L., Lyons, J., Kanehi, P., Bannerman, R., & Emmons, E. (2000). Watershed urbanization and changes in fish communities in southeastern Wisconsin streams1. *JAWRA Journal of the American Water Resources Association, 36*(5), 1173-1189.

Warren, M., & Pardew, M. (1998). Road crossings as barriers to small-stream fish movement. *Transactions of the American Fisheries Society, 127*(4), 637-644.

Wells, W. G., Johnson, T. C., Gebhard, A. E., Paine, R. T., Hix, L. A., Ferrell, H. N., . . . Perkin, J. S. (2017). March of the sculpin: Measuring and predicting short-term movement of banded sculpin cottus carolinae. *Ecology of Freshwater Fish, 26*(2), 280-291.

Wenger, S. J., Peterson, J. T., Freeman, M. C., Freeman, B. J., & Homans, D. D. (2008). Stream fish occurrence in response to impervious cover, historic land use, and hydrogeomorphic factors. *Canadian Journal of Fisheries and Aquatic Sciences, 65*(7), 1250-1264.

Wenger, S. J., Roy, A. H., Jackson, C. R., Bernhardt, E. S., Carter, T. L., Filoso, S., . . . Walsh, C. J. (2009). Twenty-six key research questions in urban stream ecology: An assessment of the state of the science. *Journal of the North American Benthological Society, 28*(4), 1080-1098.

Wheeler, A., Angermeier, P., & Rosenberger, A. (2005). Impacts of new highways and subsequent landscape urbanization on stream habitat and biota. *Reviews in Fisheries Science, 13*(3), 141-164.