

THESIS

STONECAT ECOLOGY IN ST. VRAIN CREEK, CO

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WE HEREBY RECOMMEND THAT THE THESIS PREPARED UNDER OUR
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ABSTRACT

STONECAT ECOLOGY IN ST. VRAIN CREEK, CO

Stonecat *Noturus flavus* are a small-bodied native catfish found from southern Canada to the southern United States, and from the Appalachian Mountains to the Rocky Mountains. In Colorado, there are two remaining populations of Stonecat, including one geographically isolated population in St. Vrain Creek, which runs through the Front Range in Longmont, CO. There are five major drainages running through Colorado's Front Range, which is where most of the state's population is concentrated. As such, these streams are highly urbanized. When compared to the other four major Front Range streams, St. Vrain Creek contains a disproportionately high number of native fish species, including Colorado Species of Special Concern such as Stonecats. There has not yet been a quantitative analysis of population demographic parameters or individual habitat selection preferences of Stonecats. I sought to estimate both of these through a mark-recapture study using passive integrated transponder (PIT) tags. There are a number of assumptions associated with mark-recapture studies which I addressed through individual experiments, including tag loss, physical closure and detection probability of known tags.

I evaluated tag loss under laboratory conditions. PIT tags were surgically implanted into the peritoneal cavity of Stonecats ($n = 157$) ranging from 71 mm to 213 mm through an incision closed with a single Braunamid suture and the fish were monitored for 120 weeks. After 120 weeks, there were fifteen lost tags (9.6%) and eight mortalities (5.0%). I evaluated our dataset of individual encounter histories and covariates including time since tagging, fish length and tag type in a multistate model framework using Program MARK. Time since tagging has an inverse

effect on tag loss; if fish are going to lose tags, it will be relatively soon post-tagging.

Additionally, fish length has a negative effect, with tag loss decreasing with fish length. These results support our assumption that using PIT tags to individually mark Stonecats is an appropriate method, and we now have a better understanding of tag loss rates over a long-term study period.

I evaluated population demographic parameters and individual habitat selection preferences of Stonecats in a field experiment. PIT tags were surgically implanted in Stonecats ($n = 679$) ranging from 70 mm to 230 mm. I monitored tagged Stonecats with both static and mobile PIT antennae. Our results from the static antennae show that the proportion of Stonecat encounters are higher at night and during the summertime. From the mobile PIT antenna results, I determined Stonecats prefer coarse substrate at an intermediate velocity (0.29 m/s) and intermediate depth (0.3 m). Conclusions from this study will be used to inform future urban stream management in conjunction with managing for sensitive fishes such as Stonecats.

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CHAPTER 1: PIT TAG LOSS IN A SMALL-BODIED CATFISH

Introduction

Quantifying animal vital rates, movement and habitat selection is essential to understanding population dynamics (McClintock et al. 2014) and making sound management decisions. Typically, population parameters are estimated using marking studies in which animals are marked and actively recaptured (Williams et al. 2002) or passively monitored using techniques such as radio telemetry (Millspaugh and Marzluff 2001). Historically, fisheries biologists have used radio telemetry, batch marking such as coded wire tags, or individual marking such as numbered floy tags to monitor fish populations (McKenzine et al. 2012, Nielsen 1992). Each of these has limitations; radio telemetry is expensive, batch marks do not provide individual data, and individual marks typically require that fish are physically recaptured. More recently, fisheries biologists have used radio frequency identification (RFID) passive integrated transponder (PIT) tags to study fish movement (Zydlewski et al. 2003; Cucherousset et al. 2005), habitat selection (Roussel et al. 2000, Roussel et al. 2004), demography, population structure, and vital rates (Al-Chokhachy and Budy 2008). The use of PIT tag technology has revolutionized our understanding of fish movement and habitat selection. The advantages of PIT tags are that they allow individual identification, have long lifespans, are easily applied, and, when properly sized, have minimal effects on growth, survival, and physiological performance (O'Donnell et al. 2010; Ficke et al. 2012). They are also relatively inexpensive compared to other tagging methods such as radio telemetry. Hence, PIT tags have allowed fisheries biologists to analyze movement and habitat preferences at much smaller spatial and temporal scales than previously possible. However, like other tagging methods, it is important to explicitly address

potential limitations of the technology that could influence parameter estimates. One critical factor that could affect PIT tag reliability is the potential for tag loss.

Tag loss is a problem for most tag types and varies across species and implantation location and procedure (Clugston 1996; Buzby and Deegan 1999). As such, it is preferable to conduct tagging experiments to address tag loss (Daugherty and Buckmeier 2009). Such studies should evaluate variables such as fish size, the time a tag is at large, and the type of tag used if more than one is being considered and other factors that could influence tag loss. The location of the tag and the procedure of implanting the tag are particularly important (Jepsen et al. 2002). For example, PIT tags may be injected into muscular tissue or the body cavity using a hypodermic injector. Alternatively, intra-peritoneal surgical implantation may be used, and is regarded as a more appropriate method for long-term studies (Bridger and Booth 2003). However, intra-peritoneal implantation by any method may not be suitable for all species because of physiological processes that allow them to remove foreign material from their body cavities (Moore et al. 1990).

Catfish species (Order *Siluriformes*) from several taxonomic families are known to be capable of expelling internal tags that are implanted in their peritoneal cavity and as such, there has been varied success in tagging catfishes. Some examples of tag types that catfish have been known to expel include PIT tags (Baras and Westerloppe 1999, Daugherty and Buckmeier 2009, Moore 1992), ultrasonic tags (Sakaris et al. 2005) and telemetry tags (Holbrook et al. 2012). They expel tags from the peritoneal cavity by first surrounding them in a fibrous capsule, migrating the fibrous capsule to the intestinal tract, creating a false intestinal wall around the fibrous capsule, absorbing the fibrous capsule into the intestine, and finally passing the capsule through the intestine. The process of transintestinal expulsion has been documented across

Siluriformes, including Channel Catfish *Ictalurus punctatus* (Summerfelt and Mosier 1984, Marty and Summerfelt 1986), Brown Bullhead *Ameiurus melas* (Sakaris et al. 2005), African Catfish *Heterobranchus longifilis* (Baras and Westerloppe 1999), Blue Catfish *Ictalurus furcatus* (Holbrook et al. 2012), and Mekong Giant Catfish *Pangasianodon gigas* (Mitamura et al. 2006). Clearly, transintestinal expulsion could have negative consequences for PIT tagging studies that involve implanting tags into the peritoneal cavity (Arnason and Mills 1981, Conn et al. 2004, Seber and Felton 1981). Transintestinal expulsion has been documented in large and medium-sized catfishes but not in smaller catfishes, such as madtoms *Noturus spp.* I was particularly interested in using PIT tags in a small catfish known as Stonecats *Noturus flavus*.

The Stonecats is the largest species of madtom and may share the ability to transintestinally expel PIT tags with the larger species in Siluriformes. My interest in tag loss was motivated by a field experiment I was conducting to evaluate Stonecat habitat selection and vital rates in a wild population. Mark-recapture studies rely on a number of modeling assumptions, including no significant tag loss or mortality from tagging (Burnham et al. 1987). The effects of unidentified tag loss on apparent survival estimates can be important (Pollock et al. 1990), as tag loss imposes a negative bias on survival estimates due to a reduction in marked individuals available for recapture (Bateman et al. 2009). Therefore, my specific objective in this study was to conduct a laboratory experiment to assess the rates of PIT tag loss and mortality from tagging in Stonecats prior to conducting my field experiment.

Methods

Stonecats (n = 157; mean TL = 152.8 mm, mean wet weight = 35.2 g) collected via backpack and barge electrofishing in Horse Creek, WY were transported to the Colorado State University Foothills Fishery Lab, Ft. Collins, CO in a 113-liter cooler with oxygen on three

occasions (May, August and November 2015). The collection occasions corresponded to individual tagging occasions (group) that had the same tagging procedures and tagging personnel. During transport, fish condition, oxygen levels and temperature were monitored approximately every 30 minutes. Fish were held in four 300-liter tanks receiving continuous flows of five liters per minute of air-saturated 15 - 21°C water. Initially, Stonecats were held at densities of 17 - 20 fish per tank; however, after assessing fish condition, I collected more fish, and increased the density to 30 - 35 fish per tank. Fish were randomly distributed across the four tanks. Tanks included sections of PVC pipe as cover for the fish. Fish were fed to satiation on a mixed diet of frozen chironomid larvae and a commercially available feed (Bio Oregon 2-mm pellets).

After an initial laboratory acclimation period of approximately two weeks, fish underwent surgical tagging procedures. Prior to surgery, fish were weighed (WW, g), measured (TL, mm) and anesthetized using neutrally buffered 10 mg/kg tricaine methanesulfonate (MS-222). I defined complete immobilization as the point at which fish were unable to maintain equilibrium and began the surgical procedure. PIT tags were surgically implanted into the peritoneal cavity of Stonecats ranging from 71 mm to 213 mm through an incision closed with a single Braunamid suture. Fish between 70 mm and 180 mm TL were tagged with Oregon RFID HDX 12-mm tags and fish greater than 180 mm were tagged with either Oregon RFID HDX 12-mm (n = 2; mean WW = 50.8 g) or Oregon RFID HDX 23-mm tags (n = 22; mean WW = 62.7 g) which were randomly assigned. I based the tag size on findings from Winter (1996) that determined tag weight should be 1 - 2% of fish body weight.

Tanks were monitored 6 days per week for mortalities and shed tags. Daily checks consisted of removing all habitat features and air-stones and visually inspecting tank-bottoms for

lost tags. Tags were easily seen on the tank bottom, but to ensure thorough monitoring, a magnet was used to scan for expelled tags. Full counts and tag scans using an Oregon RFID HDX Proximity Reader of all the fish in the tanks were conducted every three months to validate the daily census and determine whether fish had shed tags that were not detected during the checks. Mortality directly post-tagging was attributed to the surgical procedures. Mortality within the first week was attributed to complications from the surgical procedures. Mortality after the first week post-tagging was considered mortality not directly related to the surgical procedures (Caputo et al. 2009).

Throughout my tag retention study, fish were used in a concurrent study evaluating the effects of slope on Stonecat passage success in rock ramp fishways. A subset of fish ($n = 100$) were transferred from their tanks to the laboratory fish passage structure and placed in the downstream velocity refuge area. Fish were given 20 hours to attempt to ascend the structure. After each trial, the fish were captured, scanned and identified with a handheld PIT tag reader, and returned to their holding tanks. Although their use in the fish passage structure was not treated as a factor in the analysis of tag retention, I feel the passage trials exposed the fish to conditions simulating those that fish may encounter in the field, including varied velocity, depth and substrate compared to the conditions in the tanks.

Due to space constraints, on 20 July 2016 one tank was removed from the study and the 47 Stonecats within were euthanized (250 mg/L MS-222). The fish in the remaining three tanks ($n = 87$) were monitored for tag retention until the experiment was terminated on 9 Sept 2017.

Analyses

I analyzed the laboratory PIT tag dataset using a multistate mark-recapture framework (Brownie et al. 1993, Fetherman et al. 2015, Lebreton and Cefe 2002, Nichols and Kendall 1995)

in Program MARK Version 6.1 (White et al. 2006) with two states; “TAG” and “NO_TAG” (Figure 1.1). Fish could either stay tagged (Ψ_{TT}) or transition to NO_TAG (Ψ_{TN}). Fish could not transition from NO_TAG to TAG ($\Psi_{NT} = 0$). If fish transitioned from “TAG” to “NO_TAG”, I censored them from the dataset for the remainder of the study, as fish were not re-tagged ($\Psi_{NN} = 0$). I modeled the loss of the tag, conditional on surviving. I censored any fish that died, and thus did not model mortality. However, because fish were housed in enclosed tanks, I assumed all mortalities and lost tags were observed. As such, in Program MARK, the probability of survival for both states (tag and no tag) equal to each other and fixed at one ($S_{TAG} = S_{NO_TAG} = 1$). Similarly, I assumed that probability of detection for both states was equal and fixed at 1 ($p_{TAG} = p_{NO_TAG} = 1$).

I had three groups that corresponded to my three tagging events in May, August, and November 2015. This resulted in staggered entry times for each group and was modelled separately in the analysis. Specifically, individuals tagged in May 2015 were at large for 120 weeks, individuals tagged in August 2015 were at large for 108 weeks, and individuals tagged in November 2015 were at large for 96 weeks. I restricted these three groups to the appropriate age in weeks. I hypothesized that tag loss was a function of time since tagging. I also examined if individual covariates (fish length and tag type) affected tag loss. I used fish length (TL, mm) rather than wet weight because length is more easily measured in field situations and I wanted this laboratory study to inform my field observations. Length was highly correlated with length at the time of tagging ($r^2 = 0.95$; Figure 1.2). To better visualize the differences in estimated tag loss across fish length and time since tagging, I arcsin-transformed the estimated weekly probability of tag loss and the confidence bounds post-analysis (Figure 1.4).

Results

After 120 weeks, there were fifteen lost tags (9.6%) and eight mortalities (5.0%). Eighty-seven individuals were still alive and tagged at the end of the experiment (Table 1.1). Three of the mortalities occurred within a week of the fish being tagged, and I attributed these mortalities to surgical procedures. Subsequent necropsy confirmed that the surgical incision was too deep and punctured vital organs. The cause of the other five mortalities is uncertain.

My best model in predicting tag loss (the transition from tag to no tag; Ψ_{TN}) included group, time since tagging, fish length, and tag type, and the fish length \times tag type interaction (Table 1.2). My second-best model included group, time since tagging, and fish length (Table 1.2), while the third-best model included group, time since tagging, fish length and tag type (Table 1.2). The fourth-best model included group and time since tagging (Table 1.2). My top four models showed a decreasing trend in tag loss across the three tagging groups; and indicates that increased tagging experience may decrease tag loss. However, the 95% confidence intervals for the beta estimates (logit scale) from the top model for group overlapped zero and each other, suggesting that group does not have a statistically significant effect (Figure 1.3A). Additionally, the fifth-best model, which only had tagging group as a factor, had little support (Table 1.2). In the top model, the 95% confidence interval for the beta estimate of tag type also overlapped zero, and I conclude that tag type does not significantly affect tag loss at the 95% confidence level (Figure 1.3A).

Three parameters in my top model (time since tagging, fish length, and the interaction of length and tag type) have 95% confidence intervals for their respective beta estimates that do not overlap zero (Figure 1.3B). Time since tagging has a negative effect on the beta estimate for estimated probability of weekly tag loss, indicating that tag loss occurs early and that the longer

a tag is in a fish the less likely it is to be lost. The estimated weekly probability of tag loss decreased with increasing fish size during tagging periods (one, 52 and 120 weeks) and with increasing length of time since the fish were tagged (Figure 1.4). Fish length also has a negative effect on estimated probability of weekly tag loss, indicating that tag loss is higher in smaller fish (Figure 1.3B). The interaction between length and tag type was probably an artifact of my tagging protocols because large fish (> 180 mm TL) received either a 23-mm or a 12-mm tag, whereas small fish (70 – 180 mm TL) only received 12-mm tags.

Discussion

The overall goal of my study was to estimate the probability of PIT tag loss in Stonecat to inform my concurrent field study and I demonstrated that Stonecats can successfully be tagged with 12 and 23-mm PIT tags with low levels of mortality and tag loss. Tag loss was influenced by the time since tagging, fish length, tag type, and interaction of fish length by tag type.

One of my concerns was mortality associated with surgically implanting PIT tags. However, overall mortality in my study was low and furthermore, mortality associated with tagging usually occurred within one week of tagging and could be attributed to surgery, such as puncturing of internal organs. Therefore, I feel confident that Stonecats can be successfully PIT-tagged without biasing subsequent survival estimates. When tagging European bullheads *Cottus gobio*, another small-bodied benthic fish, with 12-mm PIT tags and surgical implantation, Bruyndoncx et al. (2002) found no tag loss or mortality during a four-week laboratory experiment (n = 6). After a seven-week field experiment in which fish were double-marked, only one fish had lost their PIT tag. They concluded that tagging small-bodied benthic fish is possible without high levels of tag loss or mortality.

Time since tagging in my study was inversely related to the estimated probability of weekly tag loss, and typically fish lost tags in the first few weeks post-tagging. Other studies have shown similar trends in tag loss (Welch et al. 2007; Feldheim et al. 2002). Welch et al. (2007) concluded that loss of surgically implanted acoustic tags in Steelhead Trout *Oncorhynchus mykiss* was a function of time since tagging, with most of their tag loss occurring between six and eight weeks post-tagging. In a five-year study, Feldheim et al. (2002) used both dorsally injected PIT tags and genetic marking to identify individual Lemon Sharks *Negaprion brevirostris* and found 47 of the 388 tagged sharks had lost their tags (12.1%). However, 11 of the 41 tags lost occurred within the first week post-tagging (23.4%). Studies on Shovelnose Sturgeon *Scaphirhynchus platyrhynchus* showed a different trend than my study, with an increasing trend in injected PIT tag loss over time in both the operculum and dorsal musculature (Hamel et al. 2012). Whether tag loss is high initially or increases over time, it is necessary to account for the relationship between time since tagging and tag loss to avoid biasing vital rate estimates. As such, time since tagging is an important covariate to consider in any mark-recapture study where tag loss is a concern.

Stonecat size was an important predictor of tag loss in my study and tag loss decreased with fish total length. PIT tag loss as a function of fish size has been well-studied in a number of fish families, including Salmonidae (Prentice et al. 1999, Dare 2003), Cyprinidae (Ward et al. 2008), Cottidae (Bruyndoncx et al. 2002) Fundulidae (Clark 2016) and Sparidae (Navarro et al. 2006). However, in many of these studies, tag loss rates increased with increasing fish size. However, most of my Stonecats were smaller than individuals used in other studies, and there was high variability surrounding the tag loss parameter in small (> 100 mm) Stonecats.

Group (May, August, and November) was a surrogate for tagging experience and presumably increased experience could lead to better tag retention. However, the beta estimates and associated 95% confidence intervals overlapped zero and each other. Although the beta estimates and variation indicate caution in interpreting the tagging group variable, it was a factor in the top five of the six models. The decreasing trend in beta estimates across the three groups indicate that tagging experience may be important. My third tagging group (November 2015, n = 79) only had one mortality, which occurred during surgical procedures, and only one fish lost their tag, approximately one-year post-tagging. Dare (2003) concluded there is an effect of tagging experience on tag loss in Chinook Salmon *Oncorhynchus tshawytscha*. Furthermore, Navarro et al. (2007) found that experienced tagging personnel had higher survival and tag retention when tagging smaller fish than inexperienced personnel. My results indicate that this also holds true for Stonecats, and likely could be considered true for any tagging studies. Therefore, to minimize tag loss and maximize survival in a tagging study on small-bodied fish such as a Stonecat, I suggest that tagging personnel should be highly trained on the specific organism and methods.

Although I did not assess the influence of PIT tags on physiological processes, mortality was low, and my fish grew during the experiment (initial mean TL = 153.5 mm, initial mean WW = 35.2 g, n = 157; mean final TL = 204.5 mm, mean final WW = 92.0 g, n = 87). Additionally, at the end of my experiment I found evidence of pre-spawning behavior (secondary sexual characteristics and egg development), suggesting PIT tag implantation did not affect normal physiological processes. However, a better understanding of the relationship between PIT tags and the health of Stonecat would be informative.

My results indicate that PIT tags are a viable method for individually identifying and tracking Stonecats and long-term tag loss is low. Time since tagging and fish length appear to be the best predictors of tag loss and I can now use each of these factors to analyze PIT tagging data collected from individuals in the field and adjust my demographic estimates accordingly. The effects of unidentified tag loss on apparent survival estimates can be important (Pollock et al. 1990), as tag loss imposes a negative bias on survival estimates due to a reduction in marked individuals available for recapture (Bateman et al. 2009). Initial loss could be adjusted for directly by correcting overall tag loss with apparent survival (Bateman et al. 2009) or by allowing initial apparent survival to be modeled differently from subsequent apparent survival (Pradel et al. 1997). By understanding the probability of tag loss from my laboratory study, I can correct for potential bias in the field study estimates and better estimate demographic parameters including survival and movement of the Stonecat population in St. Vrain Creek.

Tables and Figures

Table 1.1: Group and associated sample size by tag type for the laboratory PIT tag loss study.

Group	12-mm Glass	23-mm Glass	12-mm Polymer	Tags Lost (Type)	Mortalities (Type)
May 2015	35	11	0	7 (12-mm Glass = 6) (23-mm Glass = 1)	6 (12-mm Glass = 4) (23-mm Glass = 2)
August 2015	0	0	26	7 (12-mm Polymer = 7)	1 (12-mm Polymer = 1)
November 2015	0	34	45	1 (12-mm Polymer = 1)	1 (23-mm Glass = 1)

Table 1.2: Model selection results from the laboratory PIT tag loss study.

Model	AICc	Δ AICc	AICc Weights	Model Likelihood	# of Parameter	Deviance
ψ TN(Group) + Time since tagging + Length + Tag Type + Length*Tag Type	244.3	0	0.488	1	7	230.2
ψ TN(Group) + Time since tagging + Length	245.2	0.934	0.306	0.627	5	235.2
ψ TN(Group) + Time since tagging + Length + Tag Type	246.9	2.663	0.129	0.264	6	234.9
ψ TN(Group) + Time since tagging	247.9	3.669	0.078	0.16	4	239.9
ψ TN(Group)	304.7	60.43	0	0	3	298.7
ψ TN(.)	327.4	83.14	0	0	1	325.4

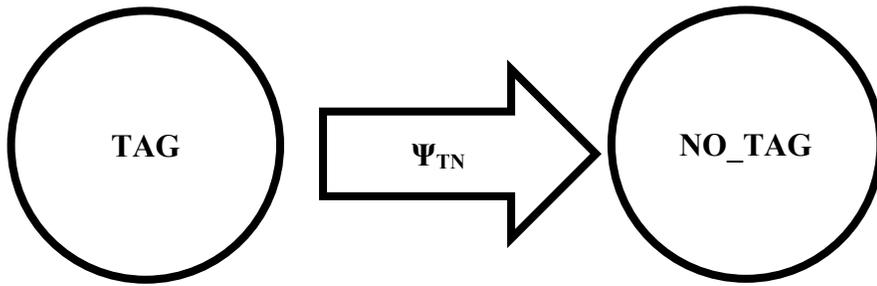


Figure 1.1: Multistate modeling framework with two states, “TAG” and “NO_TAG”. Probability of survival for both states was set equal to each other and fixed at one ($S_{TAG} = S_{NO_TAG} = 1$), as was probability of detection ($p_{TAG} = p_{NO_TAG} = 1$). I restricted the transition from NO_TAG to TAG to zero ($\Psi_{NT} = 0$). Fish could either stay tagged (Ψ_{TT}) or transition to NO_TAG (Ψ_{TN}). If fish transitioned from “TAG” to “NO_TAG”, I censored them from the dataset for the remainder of the study, as fish were not re-tagged ($\Psi_{NN} = 0$).

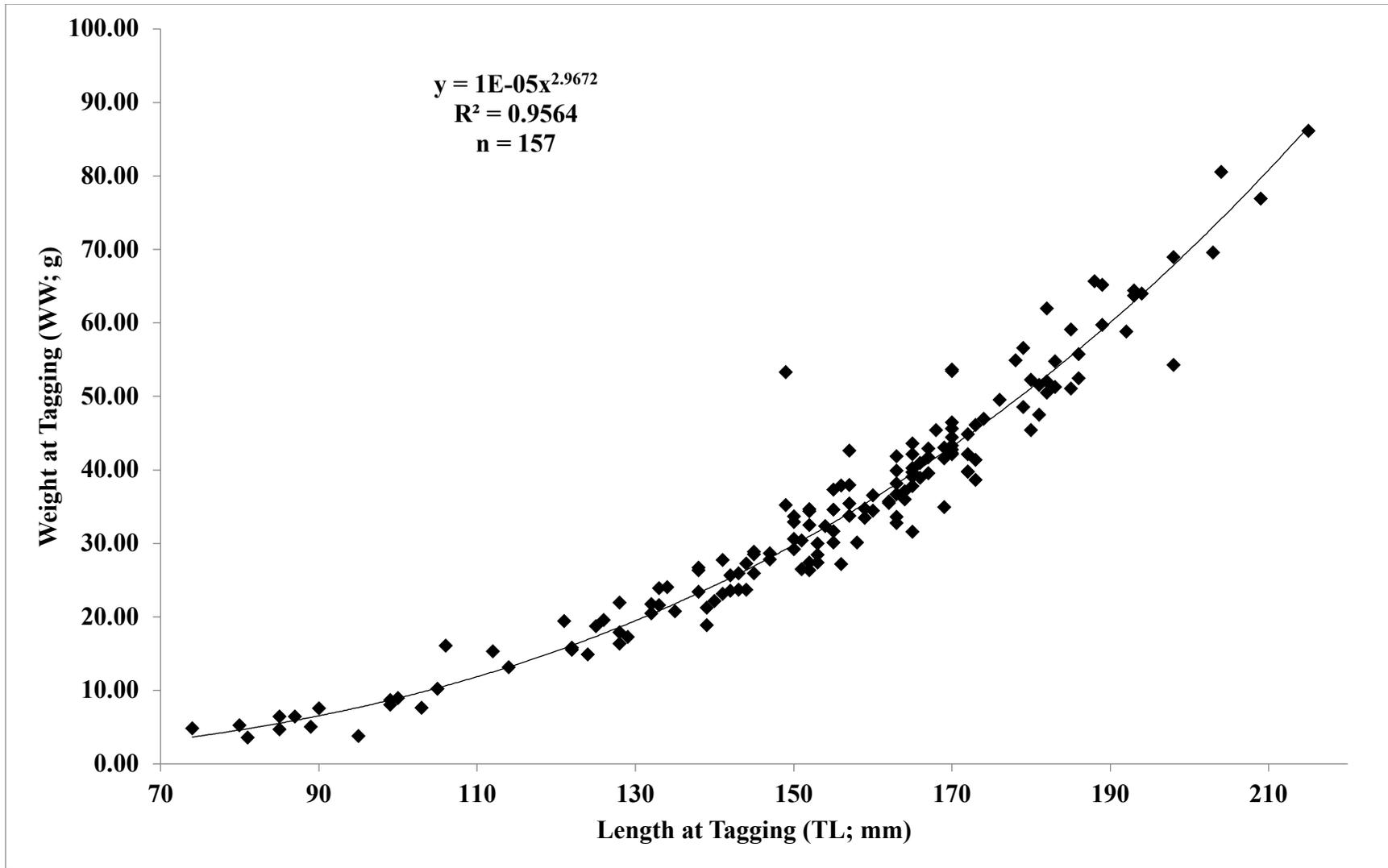


Figure 1.2: Weight at tagging (WW; g) as a function of length at tagging (TL; mm) for Stonecats.

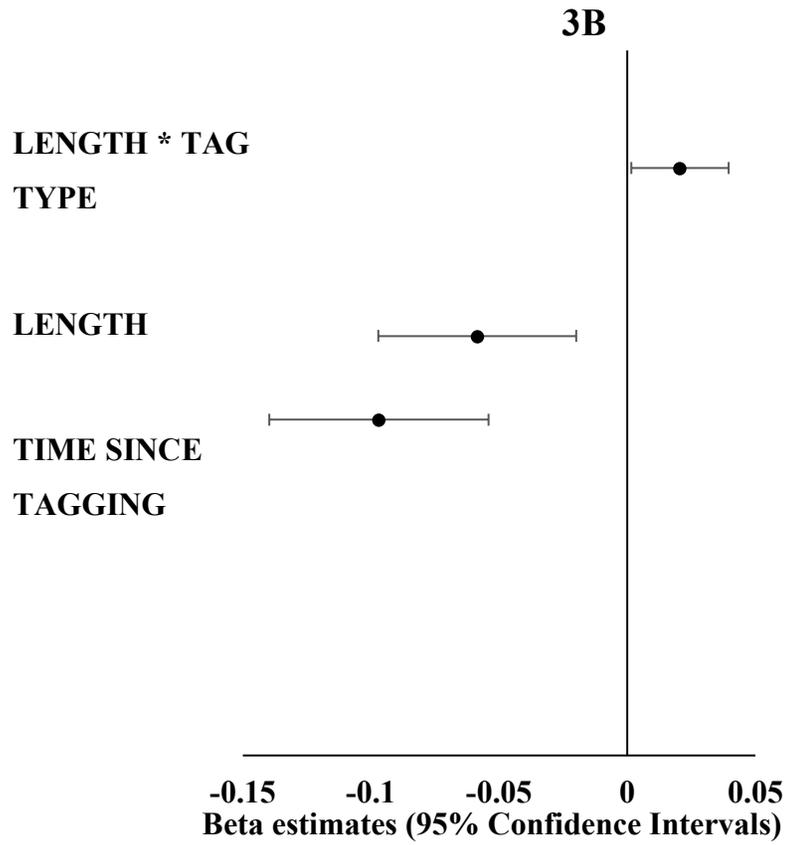
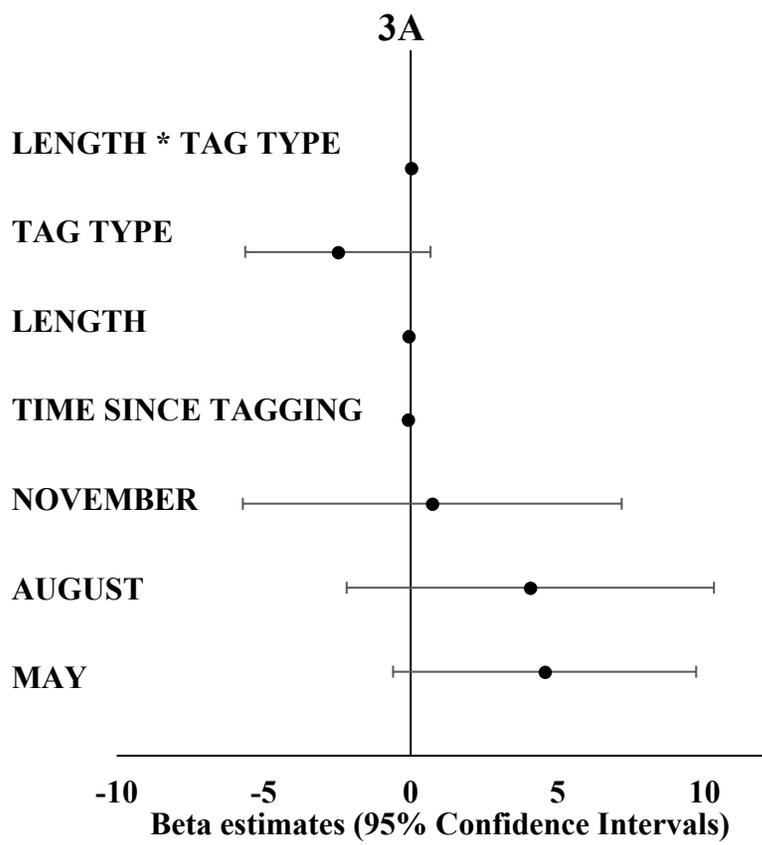


Figure 1.3 A) Beta estimates (logit scale) and associated 95% confidence intervals for all parameters in the top model. B) Parameters in the top model with beta estimates and 95% confidence intervals that do not overlap zero.

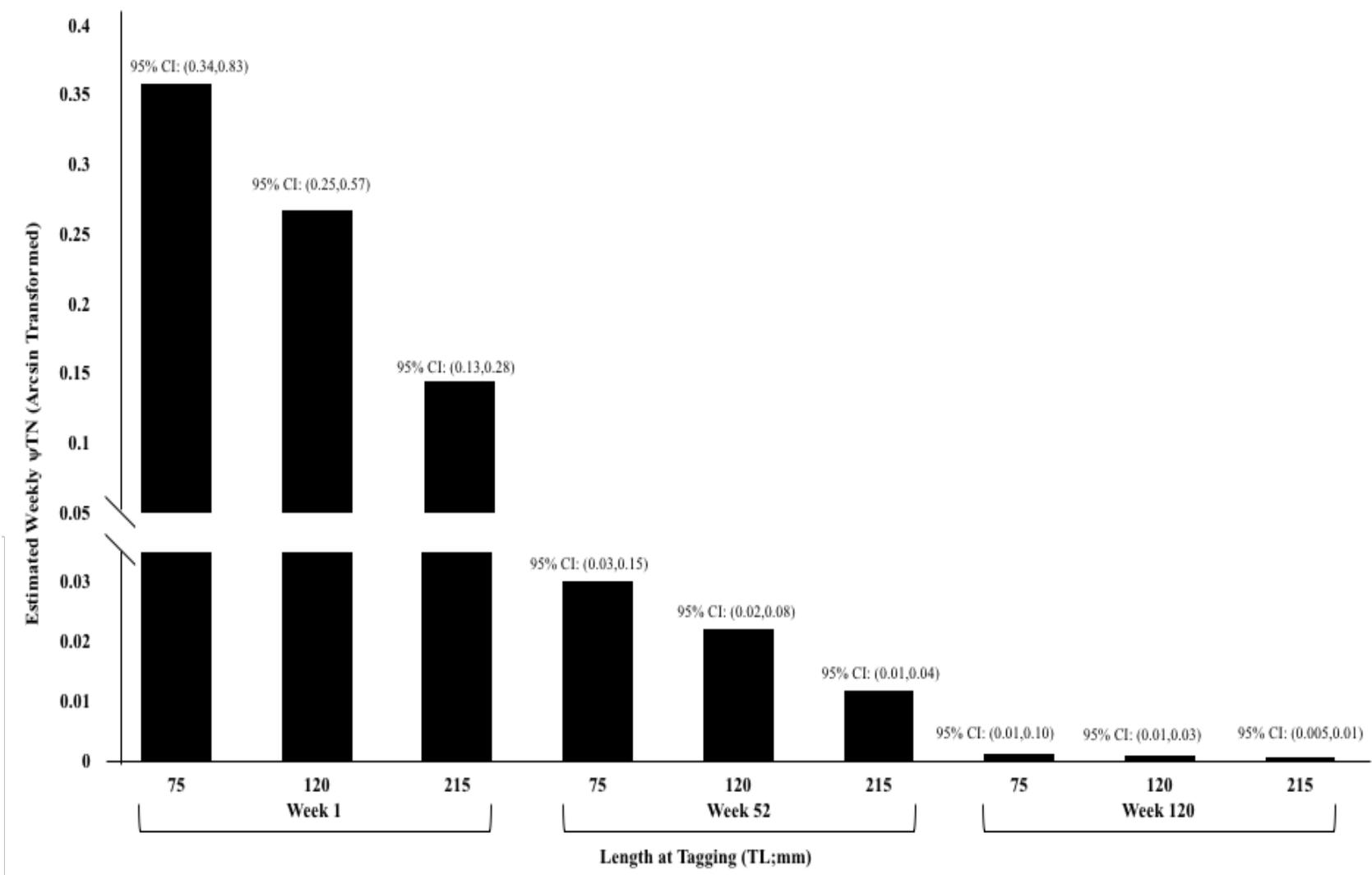


Figure 1.4: Arcsin transformed estimated weekly probability of tag loss as a function of fish length for week one, week 52 and week 120 post-tagging.

CHAPTER 2: HABITAT SELECTION AND MOVEMENT OF STONECATS IN ST. VRAIN CREEK

Introduction

Human populations are increasing worldwide and are typically concentrated near freshwater ecosystems in urban environments (United Nations 2008, Kummu et al. 2011). Therefore, freshwater resources will be increasingly developed and urbanized (Violin et al. 2011). Urbanization of streams leads to a number of impacts that include biotic and abiotic factors such as a flashier hydrograph, elevated concentrations of nutrients and contaminants, altered channel morphology, and reduced biotic richness, and increased dominance of tolerant species (Walsh et al. 2005). Furthermore, there is an inverse relationship between native species diversity and urbanization across a number of taxa, including reptiles (Germaine and Wakeling 2001), insects (Clark et al. 2007) and fishes (Weaver and Garman 1994).

Colorado's Front Range population is growing rapidly and comprises about 87% of the state's 5.6 million individuals (United States Census 2017). The growing human population means that most rivers and streams in the Front Range are at least partially urbanized and many are significantly impacted by urbanization. Five major drainages traverse the Colorado Front Range transition zone between smaller mountainous headwater streams and lower elevation Great Plains streams and rivers. The rivers and streams in each of the five drainages runs through one or more major urban landscapes before converging with the mainstem of the South Platte River. The Cache la Poudre River runs through Ft. Collins and Greeley, the Big Thompson River runs through Loveland, Saint Vrain Creek runs through Longmont, and Boulder Creek runs through Boulder, before converging with the South Platte River. The mainstem of the South

Platte River runs through several municipalities that comprise the greater Denver metropolitan area, such as Littleton, Englewood, and Denver (Figure 2.1).

One consequence of urbanization of stream ecosystems is a decreased resiliency to flooding due to local changes in hydrological conditions, increased urban runoff and flooding due to increased imperviousness (Huong and Pathirana 2013). In September 2013, a slow-moving cold-front stalled over Colorado and interacted with warm monsoonal air, resulting in a sustained rain event. Some affected areas received the annual average rainfall in only four days and many rivers across Colorado's Front Range experienced severe flooding. As a consequence of the 2013 floods, many Front Range municipalities began flood mitigation construction to alleviate the impact of future floods. Flood mitigation is typically concerned with efficiently moving water during high flow events and minimizing property damage and human risk (Schanze et al. 2007). Less emphasis is given to stream ecosystem function. Furthermore, many stream construction projects seek to increase recreational opportunities in urban streams, such as whitewater parks for recreational boating. Whitewater structures have been shown to have compounding detrimental effects on fishes (Fox et al. 2016).

It is critical to begin understanding the interaction of stream ecosystem function and diversity with urbanization processes associated with instream construction for flood mitigation and recreation. St. Vrain Creek is a highly urbanized stream; however, it contains a disproportionately high number of native fishes compared to the other Front Range streams (Nesler et al. 1997). The mechanisms underlying the native species diversity in St. Vrain Creek are unknown and I conducted observations to begin to understand how fishes were utilizing urbanized stream habitats. To accomplish this, I choose to focus on Stonecat *Noturus flavus*, a relatively common but spatially restricted species in St. Vrain Creek. There is currently no

information on Stonecat movement (Pollard 2004) but they are a benthic fish that may respond negatively to habitat alteration and increased water flow associated with flood mitigation and recreational construction. Therefore, I thought that they would provide a conservative estimate of native stream fish movement in the stream.

I conducted a field-based mark-recapture study of Stonecats in St. Vrain Creek, the westernmost population in Colorado, and potentially the most affected by anthropogenic actions and urbanization. The major objective of this study was to better understand Stonecat ecology throughout a highly-urbanized reach of St. Vrain Creek and to estimate population demographic rates and individual habitat selection preferences. Investigating habitat selection and movement patterns of Stonecats will help us understand how native fishes are utilizing urbanized stream reaches and allow for more effective management and conservation of existing populations.

Methods

Site Description

St. Vrain Creek is approximately 52 km long and drains approximately 2570 km². The headwaters are in Rocky Mountain National Park and flow eastward to the confluence with the South Platte River on Colorado's Eastern Plains. My study site was in Longmont, Colorado and is characterized as a transition-zone stream (Del Wayne et al. 1989). Transition zones are intermediate between true cold-water and warm-water sections of streams. Cold-water habitats are typically dominated by rocks and large boulders and range between 4-15°C. Warm-water habitats are typically silt and sand dominated, and range between 15-27°C. As such, transition zones are dominated by intermediate substrate sizes and intermediate temperatures between those that define the cold-water and warm-water habitat types (Tyus 2011). In this narrow transition zone from the mountain to the plains, streams are intermediate between high and low-

gradient habitats and results in high habitat heterogeneity (Haworth et al. 2016). As such, transition zone streams provide habitat for both cold-water and warm-water fishes (Rahel and Hubert 1991). Colorado transition zone streams historically featured high fish species diversity (Fausch and Bestgen 1997). St. Vrain Creek is a transition zone stream that contains a disproportionately high number of native fishes when compared to the other transition zone streams across Colorado's Front Range (Figure 2.2, Nesler et al. 1997). Not only is St. Vrain Creek a transition zone stream with disproportionately high native fish diversity, but it is also highly urbanized.

The upstream extent of my study site is defined by a 15' tall diversion structure (Beckwith Diversion) that is completely impassible to upstream movement of fishes. The downstream terminus was 4 km downstream and was defined by the upstream extent of post-flood reconstruction. The 4 km study section was primarily surrounded by industrial and commercial sites (about 2.4 km) and city park sites (about 1.6 km).

Species Description

The Stonecat is a small-bodied native catfish within the Ictalurid family of freshwater catfishes. They were first described in the Ohio River in 1818 (Rafinesque 1818). Stonecats can reach 300 mm (Trautman 1957) but typically do not exceed lengths of 203 mm (Pollard 2004). Stonecats are the latest maturing and longest lived in the genus *Noturus* (madtoms) and live 8-10 years maximum, reach sexual maturity at approximately 3-4 years old, and are slow growing (Scott and Crossman 1973; Walsh and Burr 1985). They are benthic and associated with cover making them cryptic, and difficult to sample (Puchala et al. 2016). Like most other catfishes, they are likely nocturnal (Bowman 1936). Stonecat spawning is linked to water temperatures

above 20 degrees Celsius (Steward and McColloch 1990, Scott and Crossman 1973, Walsh and Burr 1985).

Stonecat are distributed from southern Canada to the southern United States, and from the Appalachian Mountains to the Rocky Mountains (Figure 2.3). Along the western edge of their distribution, they follow the Missouri River, and are found in southeastern Montana, northeastern Wyoming, as well as southeastern Wyoming and northeastern Colorado (Figure 2.3, Page and Burr 1991). Stonecat populations are generally stable across the majority of their native range; however, populations at the fringes of their native range (including Colorado) may be declining. Stonecats are known as keystone species, indicators of water quality and ecosystem health and negatively impacted by industrial pollution (Miltner et al. 2004), habitat fragmentation, degradation and increased pollution (Kline and Morgan 2000) and dewatering (Brewer and Rabeni 2008). In Colorado, Stonecats are designated a Species of Special Concern and there are two known remaining populations of Stonecat, one in the Republican River basin near Wray, CO and one in St. Vrain Creek, in Longmont, CO (Platania et al. 1986, Woodling 1985) (Figure 2.4).

Study Description

The major objective of this study was to better understand Stonecat ecology throughout a highly-urbanized reach of St. Vrain Creek and to estimate population demographic rates and individual habitat selection preferences. I used both static and mobile passive integrated transponder (PIT) tag antennae to accomplish this objective (described below).

Radio frequency identification (RFID) PIT tags are widely used (Cucherousset et al. 2005) to study fish movement (Zydlewski et al. 2003), habitat selection (Roussel et al. 2000, Roussel et al. 2004), demography, population structure, and vital rates (Al-Chokhachy and Budy 2008) at both the population and individual level. PIT tags are passive in nature, they are

interrogated by an antenna rather than powered by a battery, which allows for long lifespan (>10 years) (Acolas et al. 2007). PIT tags provide unique identification of individuals that is crucial for estimating population demographic rates in a mark-recapture study (Dare 2003).

I installed static antennas at three sites along the study reach to monitor the movement of PIT tagged fish (Figure 2.5). The goal of the static antenna arrays was to quantitatively assess detection (p) and transition (ψ) probabilities. I evaluated Stonecat movement throughout the study reach based on electrofishing release location, static PIT antenna locations and the associated detection records from the static PIT tag antennas. On an individual scale, fish could either stay in the area (A, B, C) where they were released post-tagging, or transition between areas (A to B, B to A, B to C, C to B) thus passing over a static PIT antennae site and creating a detection event.

At each site there were paired PIT tag antennae that allowed us to establish directionality of movement (Archdeacon and Remshardt 2009) (Figure 2.6). Antenna were designed as flatbed antenna (lying on the stream bed) and the entire antenna was anchored to the substrate (Figure 2.6) (Lucas et al. 1999). I based the antennae placement and design decision on two criteria: 1) Stonecats are a benthic fish and are likely to swim close to antenna anchored on the stream bed and 2) during annual spring runoff, antenna anchored to the substrate have a lower likelihood of damage from debris. Static PIT tag antennae were constructed from either a single loop of 8 AWG THHN wire or a double loop of 10AWG THHN in order to approximate adequate inductance (25-50 μ H). To achieve proper resonance (134.2 kHz), each antenna was connected to an Oregon RFID Standard Remote Tuning Board, which was connected to a Multi-Antennae Half-Duplex (HDX) reader from Oregon RFID and tuned using an Oregon RFID HDX tuning indicator. The HDX tuning indicator is used to show when the inductance and capacitance are

tuned to 134.2 kHz by adjusting capacitor positions on the Oregon RFID Standard Remote Tuning Board. The reader box logged all detection events. All reader boxes were powered by two 12v deep-cycle marine batteries that lasted between 7-14 days. Readers were checked at least weekly to ensure proper function and charge and data were downloaded at least monthly. Static PIT tag antennae were constantly monitoring, and migration events could be evaluated on a daily, or seasonal basis. I placed all antennas in shallow riffle habitat to minimize fish loitering near the antenna and to assure that the antennae were covering the entire water column, thereby maximizing detection probability. I measured detection distance with both a 12-mm and a 23-mm tag to ensure that the entire water column was covered at the static antenna locations (Table 2.1) and feel that I could assume both geographic and demographic closure during my short and periodic mobile PIT tag sampling (see below). I also feel that I can assume geographic closure within most seasons each year, but fish may migrate without detection during the seasonal peak runoff.

I also used a mobile PIT tag antenna outlined in Richer et al. (2017) to estimate detection probability (p) and apparent survival (ϕ). Over the three years, I conducted four scans of the entire study reach, two during the summer (Summer 2016, 2017) and two during the winter (Winter 2016, Winter 2017) to examine seasonal variation in detection probability (p) and apparent survival (ϕ). A scan consisted of three passes through each reach, resulting in twelve total scans. The mobile PIT tag antenna records the GPS location at each tag detection. After each scan I returned to the GPS coordinates of each Stonecat and measured the habitat characteristics (velocity, depth, and substrate) associated with the site at each encounter.

I evaluated the effect of tag depth on detection probability (p) of the mobile PIT tag antenna using a known detection experiment. In Spring 2018, I epoxied PIT tags ($n = 50$; 12-mm

n = 25, 23-mm n = 25) to small concrete-filled cups, tied small foam floats to the cups to insure I knew their location, and placed them throughout the study reach in the three different meso-habitats: pools (n = 21) riffles (n = 16) and runs (n = 15). At each location, I measured depth (cm), which was my covariate of interest. I used the same scanning protocols (three-passes) as regular mobile antenna scans. This allowed us to explicitly test detection probability as a function of meso-habitat and depth.

Fish Collection and Tagging Procedure

Stonecats in St. Vrain Creek were captured using electrofishing and marked using PIT tags through surgical implantation into the peritoneal cavity. Electrofishing consisted of multiple backpack units or a single raft-mounted unit with multiple anodes depending on stream-reach configuration. To maximize the number of Stonecat for tagging, sampling sites corresponded to historical Colorado Parks and Wildlife sampling locations on St. Vrain Creek and known areas of high Stonecat density. The proper settings (volts, frequency, and duty cycle) for the electrofishing units were determined by the conductivity of the stream. Fish behavior, recovery, and damage were monitored during electrofishing operations and settings were adjusted accordingly to prevent any adverse effects.

Upon capture, but prior to handling, fish were held in buckets and transferred to a stationary in-stream net-pen to avoid hypoxia and other adverse effects of crowding in the bucket. Net-pens were placed in slow current to provide fresh oxygenated water without the possibility of impingement. After capture, fish were measured (TL, mm). Stonecats greater than 70 mm were anesthetized using submersion in approximately 1mL AQUI-S20E per gallon of fresh water (INAD Permit # 11-741). Induction times were noted for fish anesthetized via AQUI-S20E and reported to Colorado Parks and Wildlife for their AQUI-S20E dataset. I defined

complete immobilization as the point at which fish were unable to maintain equilibrium and began the surgical procedure. PIT tags (12-mm or 23-mm) were surgically implanted into the peritoneal cavity of Stonecats greater than 70 mm. Surgical implantation in my study included the following procedures: disinfect all surgical instruments, PIT tag and suture with 100% ethanol, create small (>5 mm) incision with scalpel in the peritoneal cavity of the fish, insert PIT tag, close surgical wound with single Braunamid suture, interrogate PIT tag with portable PIT tag scanner, place fish in recovery tank, monitor until fish regains equilibrium and release. Each tag did not constitute more than 2% of the wet body weight of the fish, as suggested by Winter (1996).

Habitat Quantification

I quantified available habitat within the study reach at the meso-habitat (Beisel et al. 1998) and micro-habitat (Frissell et al. 1986) scale. The meso-habitat delineation consisted of visually identifying individual channel geomorphologic units (pool, riffles and runs) based on previously established criteria (Beisel et al. 1998). Pools were visually identified as slow-moving deep water. Runs were visually identified as smooth, unbroken water connecting pools and riffles. Riffles were visually identified as swift shallow water with surface disturbance. I quantified the proportion of each meso-habitat by recording the perimeter using a GPS unit (Garmin eTrex 20e, Garmin USA). I created polygons for every meso-habitat and calculated the proportion of surface area by meso-habitat type.

The micro-habitat scale consisted of detailed measurements along transects within the study area. I located transects every 100 m along the entire study reach (n = 83). At each transect location, the stream width was divided into ten equidistant points, at which water velocity (m/s) and depth (m) were measured with a flow-meter and wading rod (Marsch-McBerney Flowmate

2000). Substrate was classified using a modified Wentworth Grain Size method (Blott and Pye 2001) (Table 2.4). Micro habitat measurements allowed us to quantitatively assess available micro-habitat within each meso-habitat.

St. Vrain Creek is a highly-urbanized, transition-zone stream on Colorado's Front Range and the available habitat may be different than that found in more natural systems. Therefore, I was interested in comparing Stonecat populations in my urbanized study site to a study site without urban influence. I sampled Stonecats in the Laramie River with the Wyoming Cooperative Fish and Wildlife Research Unit and evaluated capture data from electrofishing surveys for presence or absence of Stonecats in which micro-habitat metrics (velocity, depth, and substrate) were also recorded. This allowed us to compare the micro-habitat usage between an urbanized study and un-urbanized study site.

Analysis

Fish Movement

I evaluated movement at a meta-population scale in terms of proportion of tagged individuals by distance traveled (km) by comparing the distance between release location and subsequent detection location on the static antennae. Furthermore, I evaluated trends in proportion of encounters at two temporal scales; hourly and monthly to determine if there were any patterns in Stonecat encounters at the seasonal or diurnal scale.

Habitat Quantification

To quantify both the meso and micro-habitats, I used principal components analysis (PCA) in PRIMER (PRIMER 6) to determine if there were any trends in micro-habitat (velocity, depth and substrate) on meso-habitat type. I analyzed the mobile PIT antenna Stonecat data by including fish in my PCA analysis and compared meso and micro-habitat metrics taken from the

known Stonecat detections from the GPS coordinates in the mobile PIT tag antenna data. I further analyzed the PCA data by breaking the known Stonecat detections into their component meso-habitats and compared the micro-habitat characteristics of the known detections to the available habitat within each meso-habitat type. Additionally, I used the micro-habitat data from the Wyoming Cooperative Fish and Wildlife Research Unit fisheries surveys in which Stonecats were detected and compared those micro-habitat metrics against the micro-habitat metrics of Stonecats in St. Vrain Creek through a PCA framework.

Fish Distribution

I analyzed the meso-habitat results in ArcMap (Esri ArcMap 10.5). I imported all meso-habitat polygons and calculated the proportion of available habitat by meso-habitat type. After I developed the meso-habitat polygon map, I overlaid the micro-habitat transect data. By intersecting the meso and micro-habitat data, I was able to create a habitat map and link the individual velocity, depth and substrate to meso-habitat types. By determining the habitat available across the study reach, I was able to create a habitat base-layer in ArcMap and intersect spatial data collected from mobile PIT tag antennae to determine habitat selection of Stonecats. By comparing the PIT tag detection events from the mobile PIT tag antennae to the available habitat, I evaluated how individuals selected different habitat characteristics.

Detection Probability

I analyzed the known-detections dataset using a Cormack-Jolly-Seber modeling framework in Program MARK. Since the PIT-tagged concrete cups could neither move nor die, I held apparent survival fixed at 1 ($\phi = 1$). I allowed detection probability (p) to vary by the two groups (12-mm and 23-mm tag) as well as time (scan number 1-3). I also incorporated depth (cm) as a covariate and modeled estimated detection probability across a range of depths.

Results

Over three years, I captured and tagged 679 Stonecats ranging from 70-230 mm TL. Based on their size, fish were implanted with either a 12-mm glass tag ($n = 552$) or a 23-mm glass tag ($n = 127$). I had limited individual Stonecat encounters ($n = 31$ individuals) on the static PIT antennae, the majority of detections were at a single site 30 m upstream from a known location of high Stonecat density. Due to the limited Stonecat encounters on the static PIT tag antennae, I was not able to quantitatively assess detection (p) and transition (ψ) probabilities. However, the detections on the static antenna allowed for qualitative description of movement. Most of the individuals (approximately 65%) moved less than one kilometer, and all but one individual moved less than two kilometers (Figure 2.7). I was also able to evaluate Stonecat activity on two temporal scales; daily and monthly. Stonecats are documented as nocturnal (Bowman 1936) and I saw a higher proportion of stonecat encounters during the nighttime compared to the daytime (Figure 2.8). It appears that Stonecat are not highly active during the winter months, however during the late summer, there was a large spike in Stonecat detections on the static antennae (Figure 2.9).

I visually delineated 31 pools, 43 riffles and 27 runs in my study site. Micro-habitat measurements indicated that pools were characterized by greater depths (mean = 0.583 m, SD = 0.381 m), slower velocities (mean = 0.312 m/s, SD = 0.276 m/s) and smaller substrate (mean = 2.92, SD = 1.37). Riffles were shallower (mean = 0.282 m, SD = 0.146 m), had higher velocities (mean = 0.529 m/s, SD = 0.343 m/s) and larger substrate (mean = 3.10, SD = 1.03). Runs were characterized by intermediate depths (mean = 0.484 m, SD = 0.252 m), intermediate velocities (mean = 0.321 m/s, SD = 0.200 m/s) and intermediate substrate (mean = 2.78, SD = 1.34). The PCA1 axis explained 40.2% of the variation and the primary drivers were velocity and depth,

whereas the PCA2 axis explained 30.6% of the variation and the primary driver was substrate (Figure 2.15a, 2.15b ,Tables 2.3a, 2.3b).

During mobile PIT tag scans, I detected 195 unique individuals. Based on the mobile PIT antenna GPS data, Stonecats were overwhelmingly encountered in riffle meso-habitats (n = 229 encounters) and at shallow depths (mean = 0.307 m, SD = 0.106 m), intermediate velocities (mean = 0.29 m/s, SD = 0.18 m/s) and large substrate (mean = 4.68, SD = 0.58) (Figure 2.16). When I compared the Stonecat encounters within each meso-habitat type, I found that within pools, fish were associated with shallower depths, higher velocities and coarser substrate than that available (Figure 2.17a). Stonecats within riffles were associated with greater depths, lower velocities and coarser substrate than that available (Figure 2.17b). Within runs, Stonecat were associated with shallower depths, higher velocities and coarser substrate than that available (Figure 2.17c).

I was able to compare the micro-habitat selection of my urbanized Stonecats with those in a more natural setting in Wyoming. Both populations overlap in terms of depth and velocity habitats (Figure 2.18). However, the St. Vrain population shows a higher association with coarse substrate (Figure 2.18). My micro-habitat measurements show that the coarse substrate in the St. Vrain is riprap that was placed in the stream for bank stability.

My analysis of the effect of depth on detection probability indicates that detection probability (p) does not vary by group or time (Table 2.4). Thus, I had an equal probability of detection based on depth for both 12 and 23-mm tags (group), Additionally, and all three scans have an equal probability of detecting a tag (time). Depth has a dramatic effect on detection probability (Table 2.4; Figure 2.19). From the top model, estimated detection probability of the mobile PIT antenna system is 0.249 (95% CI 0.159 – 0.366) at a mean depth of 47.79 cm.

However, across the range of depths (5 – 120 cm), detection probability (p) varies from 0.813 to 0.003 (Figure 2.19). Depth influences my ability to detect Stonecats (Table 2.5) and affects each meso-habitat differently. For example, pools have a mean depth of 0.583 m. Based on my known detections experiment, at a depth of 0.58m, I estimated detection probability of 0.143. Conversely, riffles have a mean depth of 0.282 m, which equates to an estimated detection probability of 0.531 (Figure 2.19).

Discussion

Stonecats on the western extent of their North American distribution in both Colorado and Wyoming appear to be selecting riffle habitat suggesting that Stonecats are habitat specialists; although, they were found in all meso-habitats. Additionally, they appear to prefer moderate stream depths and flow and are associated with coarse substrates. When comparing my urban population in St. Vrain Creek to non-urban populations in Wyoming, they occupy similar intermediate depths and velocities. However, urban Stonecats were found in human-introduced riprap.

Stonecat in St. Vrain Creek appear to be selecting riffle habitat and this is consistent with other studies on Stonecat habitat preferences. My results also indicate that they are selecting large substrate, particularly anthropogenically derived riprap. Comparisons with stonecats in a Wyoming stream also suggest stonecat prefer large substrate. Several studies indicate that Stonecat prefer riffle or run meso-habitats between 15 and 30 cm deep (Wilhite and Hubert 2011; Johnson 1965; Bunt et al. 1997; Kline and Morgan 2000; Banks and DiStefano 2002). Stonecats also seem to prefer large rocky or woody substrate (Kline and Morgan 2000; Braaten and Berry 1997).

Although Stonecat appear to prefer riffle and run habitat across their range, they are difficult to sample and detection differences among habitats may influence inferences regarding habitat selection. I assessed detection probability of the mobile antenna and conclude that my estimated detection probability was highly influenced by depth. Therefore, my conclusion about meso-habitat selection and micro-habitat features need to be considered with caution. Either Stonecats in St. Vrain Creek are not found in pools or I may have failed to detect them in pool habitats. Detection probability decreases with depth and therefore is lower in pool habitats potentially biasing my conclusions regarding Stonecat habitat selection. My detection probability estimates suggest that pool habitat use may be higher than I report. I did not weight my habitat use estimates by detection probability because of uncertainty in how to apply detection probability across the range of habitat depths. However, I made a rough calculation by dividing the number of Stonecats in each mesohabitat by the detection probability for the mean depth in that mesohabitat. This analysis indicates that Stonecats may be as abundant in pools as riffles. Clearly, further studies are warranted to understand habitat use in deeper areas, such as pools. Detection as a function of stream depth is likely an issue that is important for many species.

I attempted to sample deep pools using a mobile PIT antenna system that had a sinking “dredge” style antenna, outlined in Fetherman et al. (2015) but no Stonecats were detected. I did not explicitly evaluate detection probability on the sinking antenna in the same manner as the floating mobile PIT antenna and I am still uncertain about Stonecat presence in pool habitats. Cucherousset et al. (2005) evaluated detection probability of a similar mobile PIT antenna system with a benthic fish, Slimy Sculpin *Cottus cognatus* and had a 0.82 probability of detection at relatively shallow depths. My detection probability of known depth PIT tags at

similar depths was comparable to those of Slimy Sculpin. However, I do not know of any studies that assessed detection in deeper pool habitats and feel that published studies may have the same potential bias as mine. Pollard (2004) showed that Stonecat in Alberta's Milk River had a patchy distribution and suggested that this may be due to sampling rather than the true distribution of Stonecats in the river. Brewer et al. (2006) suggested that Stonecats conducted pre-spawn activities in nearshore micro-habitats containing large substrates but probably moved to deeper habitats during summer to spawn. Their conclusion was based on a decrease in CPUE in shallow bedrock habitats, but they did not directly sample deeper habitats. Brewer and Rabeni (2008) found little change in seasonal habitat use in Stonecats but did not sample deeper habitat. Additionally, my survival data indicate that summer apparent survival is higher than that in the winter (Appendix 1). I feel this is probably due to Stonecats moving into deeper pool habitat in the winter because shallow riffle habitat is less available because of low winter flows and ice formation. Temporary emigration to deeper habitats may affect detection probability, apparent survival (ϕ) and other demographic parameters during mobile PIT antenna surveys (Kendall et al. 1997).

I had limited Stonecat encounters on the static PIT antennae. I measured detection distance on all antennae with both PIT tag sizes used and feel that the detection distances I had covered the water column at all flow levels except peak runoff. This allowed us to reasonably assume physical closure in my study site; fish could not pass the upstream barrier and most fish emigrating out of the study site would have been detected on the static PIT antennae. Despite the limited encounters on the static PIT antennae, the data I did gather indicated that Stonecats are not highly mobile at the reach (km) scale. Additionally, I encountered a higher proportion of Stonecats in the summer months and during nighttime. Stonecat are known to be nocturnal

(Bowman 1936) and the increased Stonecat encounters in summertime may correspond to ideal spawning temperatures above 20 degrees Celsius (Steward and McCulloch 1992, Scott and Crossman 1973, Walsh and Burr 1985).

It appears that Stonecat have relatively narrow habitat preferences, limited mobility, and highly clustered populations. Therefore, localized extirpations pose a significant threat. Fish passage designed for small-bodied Great Plains fishes is critical to mitigate these threats and avoid complete localized extirpation. Based on my findings from the short fish passage evaluation (Appendix 2), Stonecats are able to successfully navigate fish passage structures that are constructed at 2% slope with adequate roughness elements to provide velocity barriers (Swarr unpublished). If fish passage is achieved, Stonecats can not only persist in an increasingly urban environment, but, assuming suitable available habitat, are able to colonize new habitats (Pollard 2004) and expand their geographic distribution (McCulloch and Stewart 1998)

Based on the information gained from this study, I now understand some of the meso and micro-habitat requirements of Stonecats in an urban setting. However, I also feel that difficulties in sampling deeper habitats may influence my conclusions and it is important to consider habitat specific detection probability in future studies. In the face of increased population growth and the associated increase in urbanization, streams across Colorado's Front Range face challenges in managing for both human concerns (i.e. health, safety, flood mitigation and recreation) and for aquatic ecosystems, but understanding habitat requirements and increasing connectivity may ensure the longevity of some imperiled fishes.

Tables and Figures

Table 2.1: Antenna size, THHN wire gauge, configuration, dimensions and mean read range (cm) (3 PIT tag sizes) for static antenna.

Antenna Name	Wire size	Configuration	Dimensions (ft)	Mean Detection Distance (cm)		
				12-mm	23-mm	32-mm
BW_1	10 AWG	2 LOOP	30' x 2'	56.75	219.05	152.55
BW_2	10 AWG	2 LOOP	30' x 2'	48.75	145.85	81
BOS_1	8 AWG	1 LOOP	57' x 3'	87.87	270	273.35
BOS_2	8 AWG	1 LOOP	57' x 3'	79.25	292.5	258.4
MAIN_1	8 AWG	1 LOOP	57' x 3'	48.85	207.7	201.2
MAIN_2	8 AWG	1 LOOP	57' x 3'	49.4	191.2	192.05

Table 2.2: Modified Wentworth grain size for St. Vrain Creek.

Substrate Type	Score
Silt (3.9-62.5 μ m)	0
Sand (62.5 μ m-2mm)	1
Gravel (2-64mm)	2
Cobble (64-256mm)	3
Rocks (256-500mm)	4
Riprap (>500mm)	5
Bedrock	6

Table 2.3a: Eigenvalues from PCA analysis. As shown, the PCA1 axis explains 40.2% of the variation, while the PCA2 axis explains 30.6% of the variation.

PCA	Eigenvalues	%Variation	Cum.%Variation
1	1.21	40.2	40.2
2	0.918	30.6	70.8

Table 2.3b: Eigenvectors from PCA analysis. As shown, the primary drivers on the PCA1 axis are velocity and depth, while substrate is the primary driver on the PCA2 axis.

Variable	PC1	PC2
VELOCITY	0.603	0.306
DEPTH	-0.592	-0.447
SUB	0.534	-0.841

Table 2.4: CJS model selection results for PIT tagged rocks.

Model	AICc	Delta AICc	AICc Weights	Model Likelihood	# of Parameters	Deviance
{PHI(.)P(.)+DEPTH}	133.57	0	1	1	2	129.42
{PHI(.)P(..)}	198.52	64.95	0	0	2	194.37
{PHI(.)P(.t)}	200.33	66.76	0	0	4	191.82
{PHI(.)P(gt)}	203.62	70.05	0	0	7	188.13

Table 2.5: Proportion of available habitat (transect location points) as a function of detection probability.

p[Detect]	95% CI	Depth (cm)	Proportion of Habitat Transect points < Depth
0.8	(0.9,0.7)	5	0.00481928
0.7	(0.8,0.6)	16	0.13253012
0.6	(0.7,0.5)	23	0.22650602
0.5	(0.6,0.4)	30	0.32891566
0.4	(0.5,0.3)	35	0.45301205
0.3	(0.4,0.2)	43	0.59156627
0.2	(0.3,0.1)	51	0.67228916
0.1	(0.2,0.0)	64	0.79277108

Colorado's Front Range

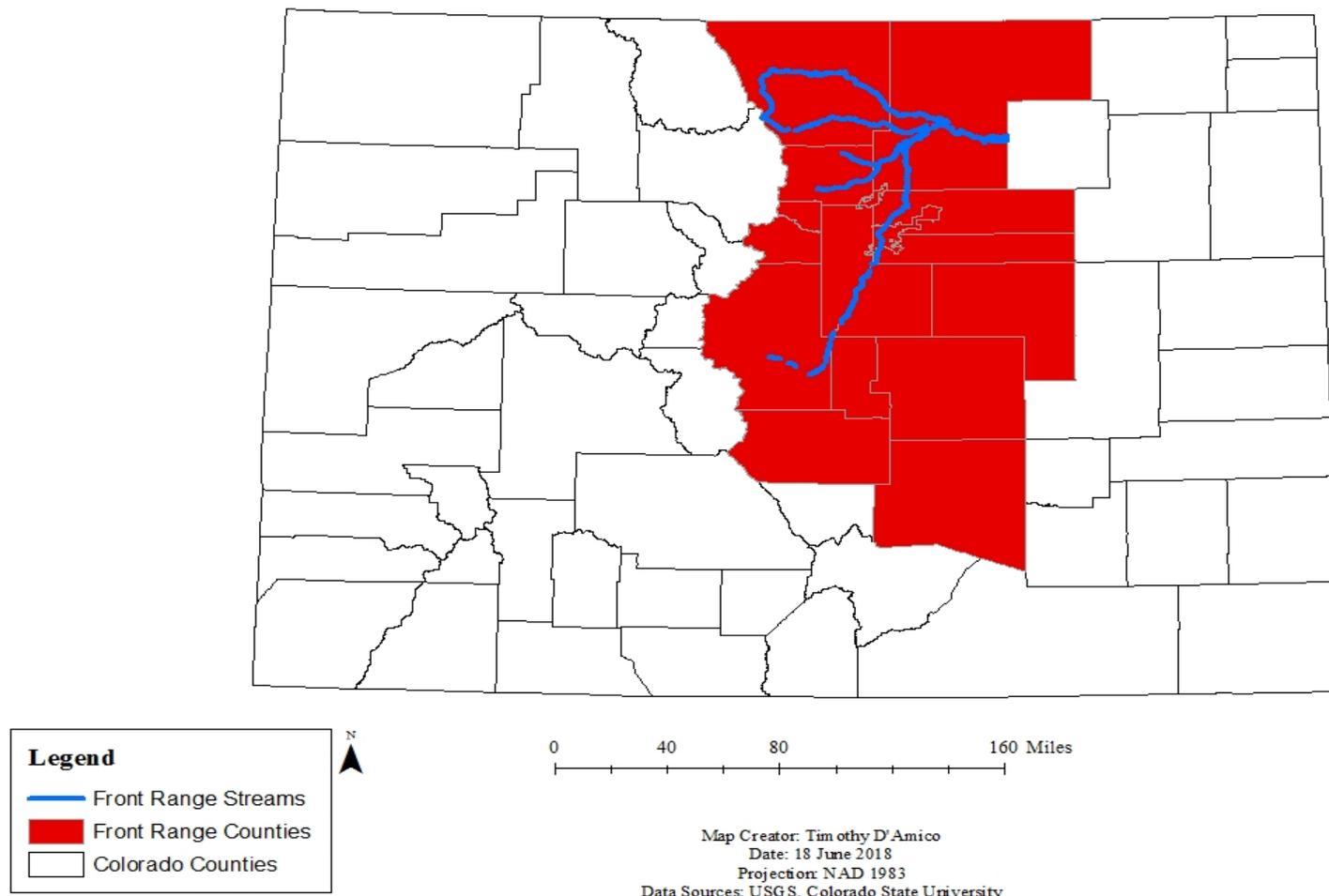


Figure 2.1: Colorado's Front Range where most of the state's population is concentrated. There are five major drainages through the Front Range, moving from North to South are the Cache le Poudre River in Ft. Collins, the Big Thompson River in Loveland, St. Vrain Creek in Longmont, Boulder Creek in Boulder and the South Platte River in Denver.

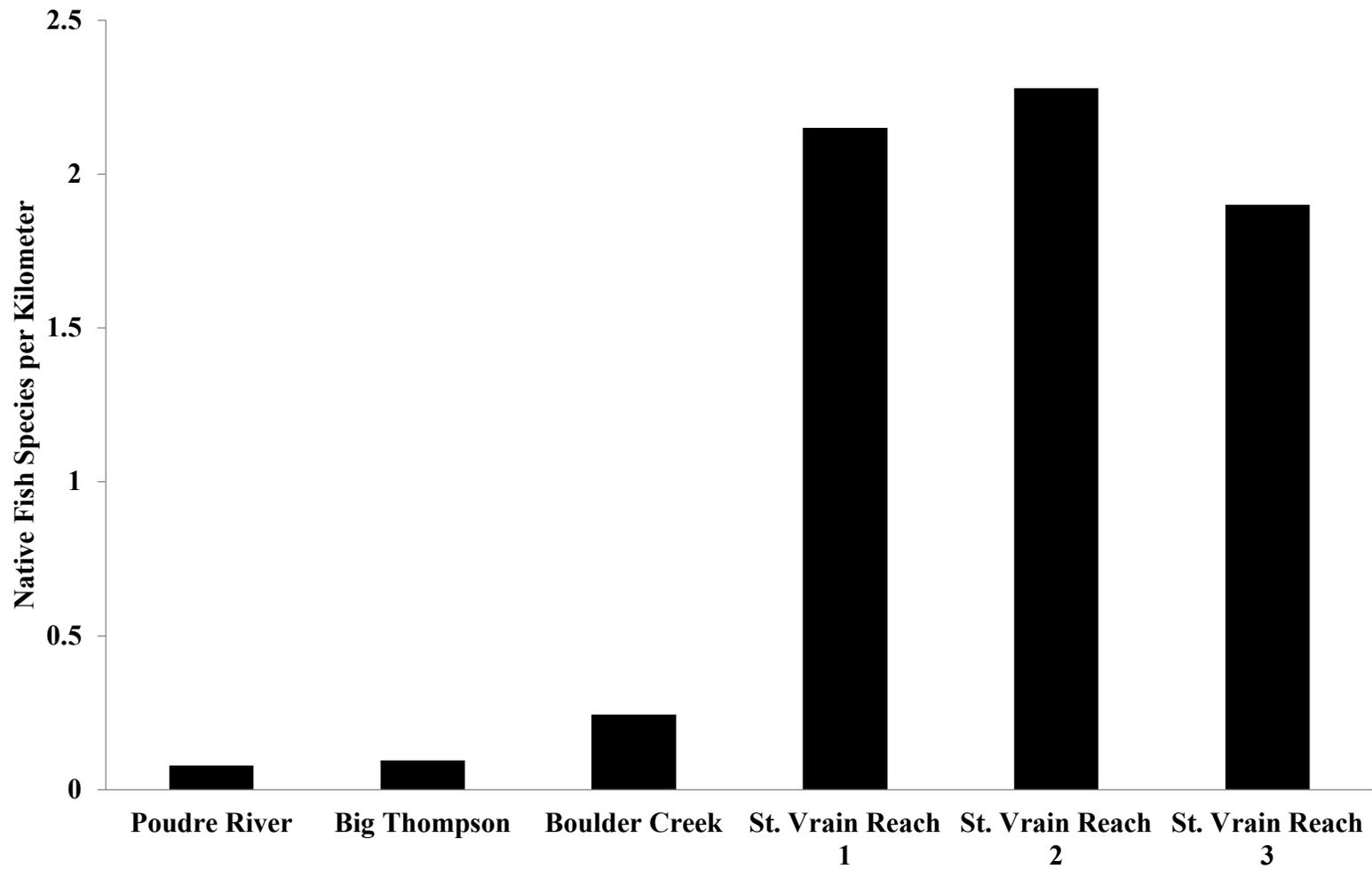


Figure 2.2: Native fish diversity in fish species per km across four major Front Range streams. St. Vrain Creek has been broken into three reaches that are sampled by Colorado Parks and Wildlife ranging from 5-7 km in length. Reach 2 corresponded with my study site.

Stonecat Native Range

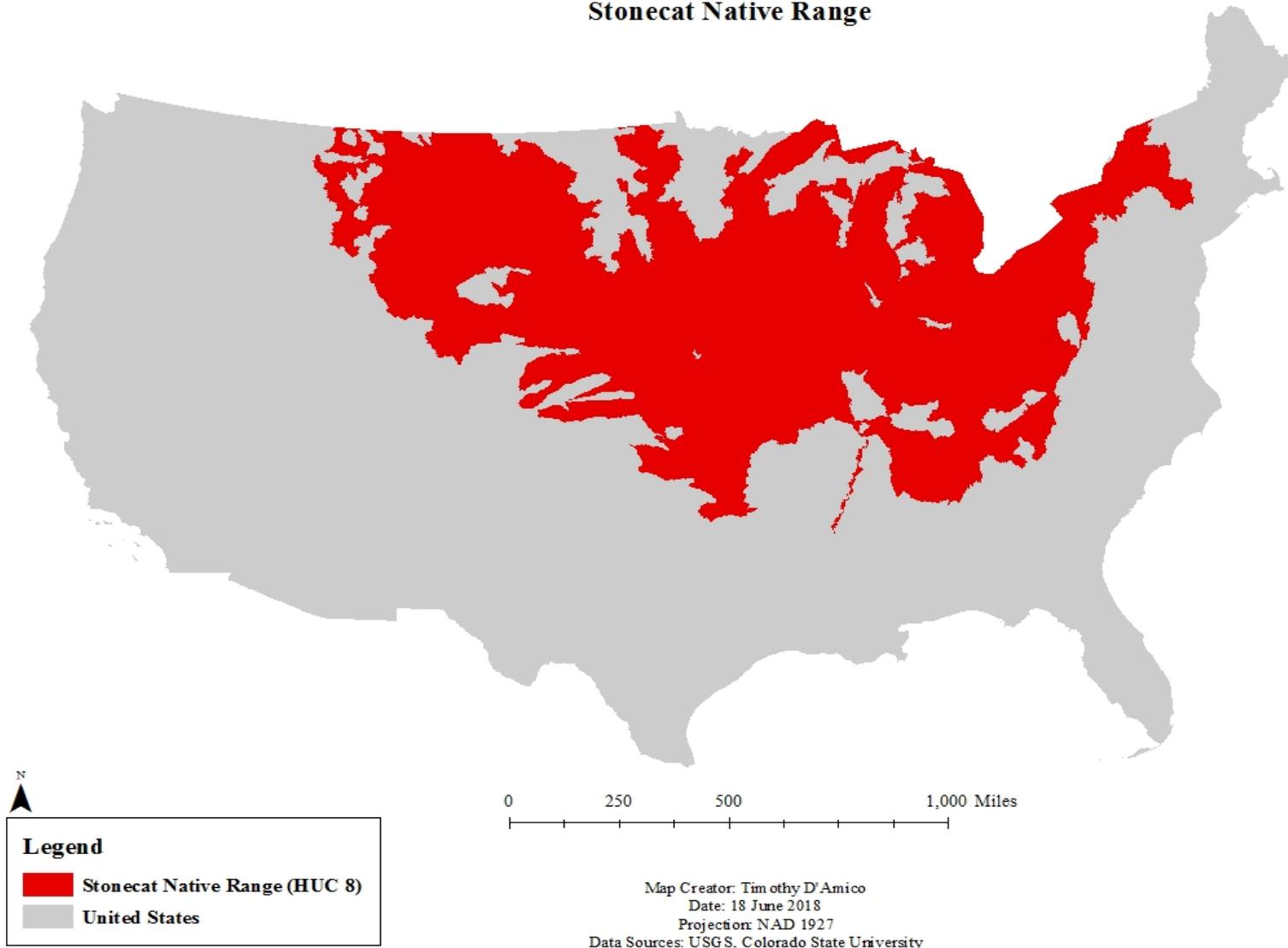
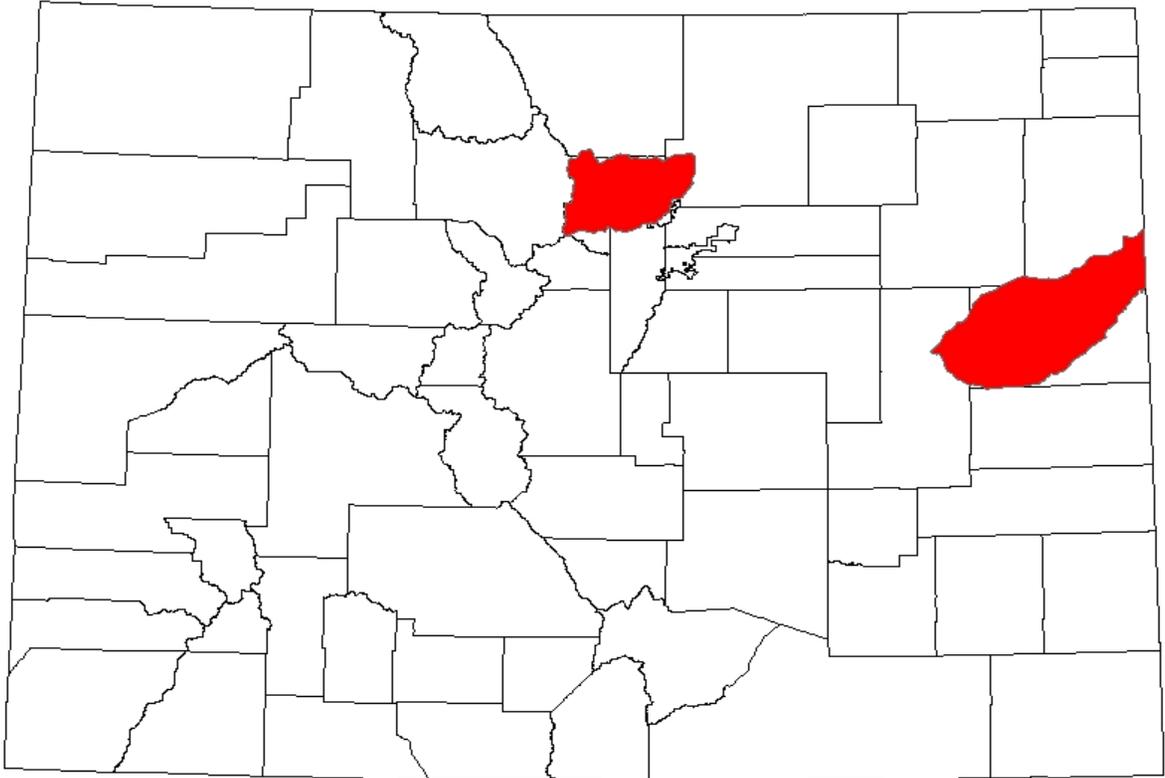


Figure 2.3: North American Stonecat distribution (native range).

Stonecat Distribution (Colorado)



Legend

- Historic Stonecat Presence (HUC8)
- Colorado Counties



Map Creator: Timothy D'Amico
Date: 18 June 2018
Projection: NAD 1983
Data Sources: USGS, Colorado State University

Figure 2.4: Historic Stonecat distribution in Colorado.

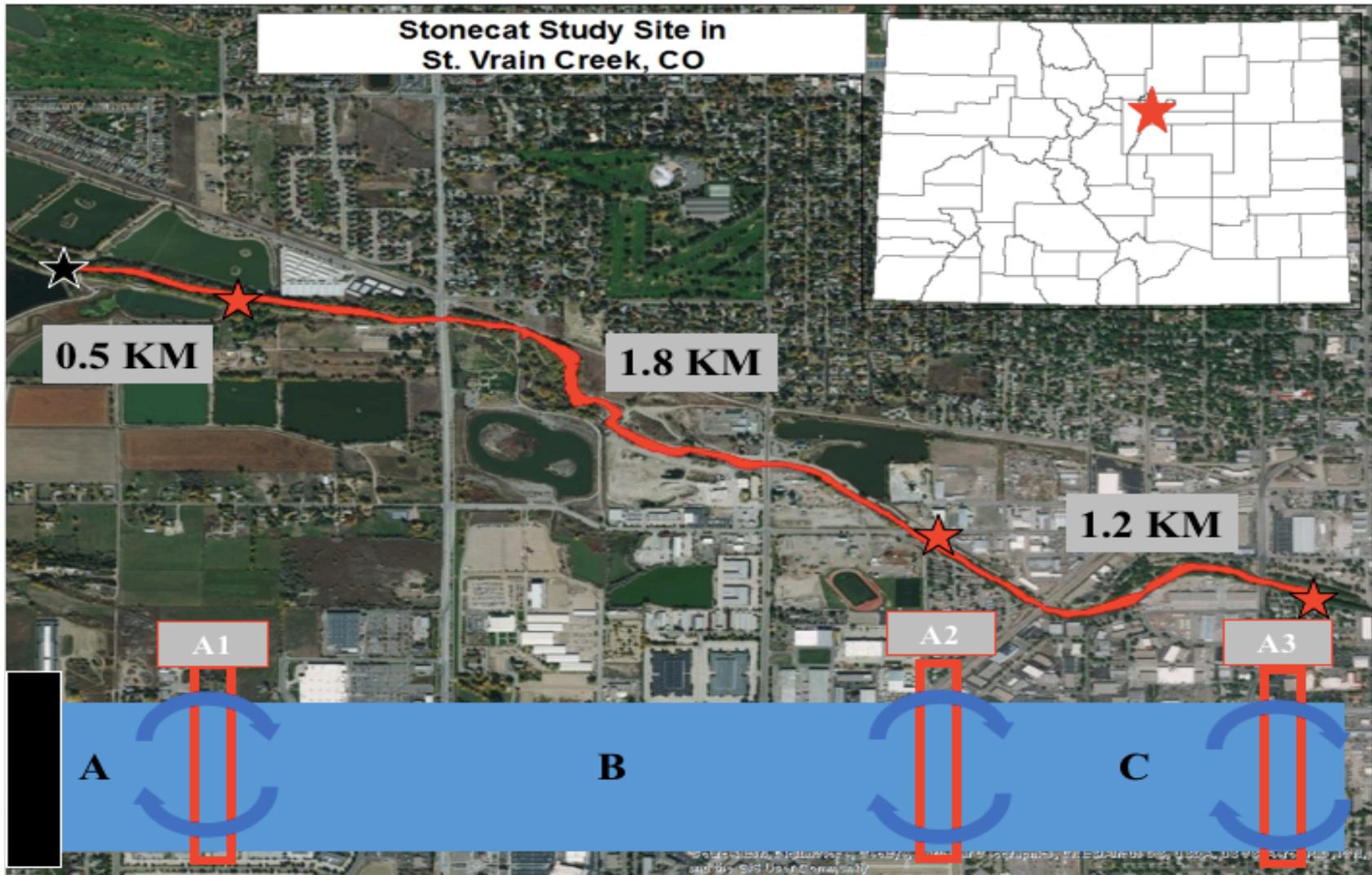


Figure 2.5: Static PIT tag antenna locations. The black star/bar represents the upstream terminus of the study site closed by a diversion structure. Antenna location 1 (A1) is located 0.5km downstream from the diversion structure. Antenna location 2 (A2) is located 1.8km downstream from A1. Antenna location 3 (A3) is located 1.2km downstream from A2. At each antenna location, there were two antennae to establish directionality of movement (shown by blue curving arrows).

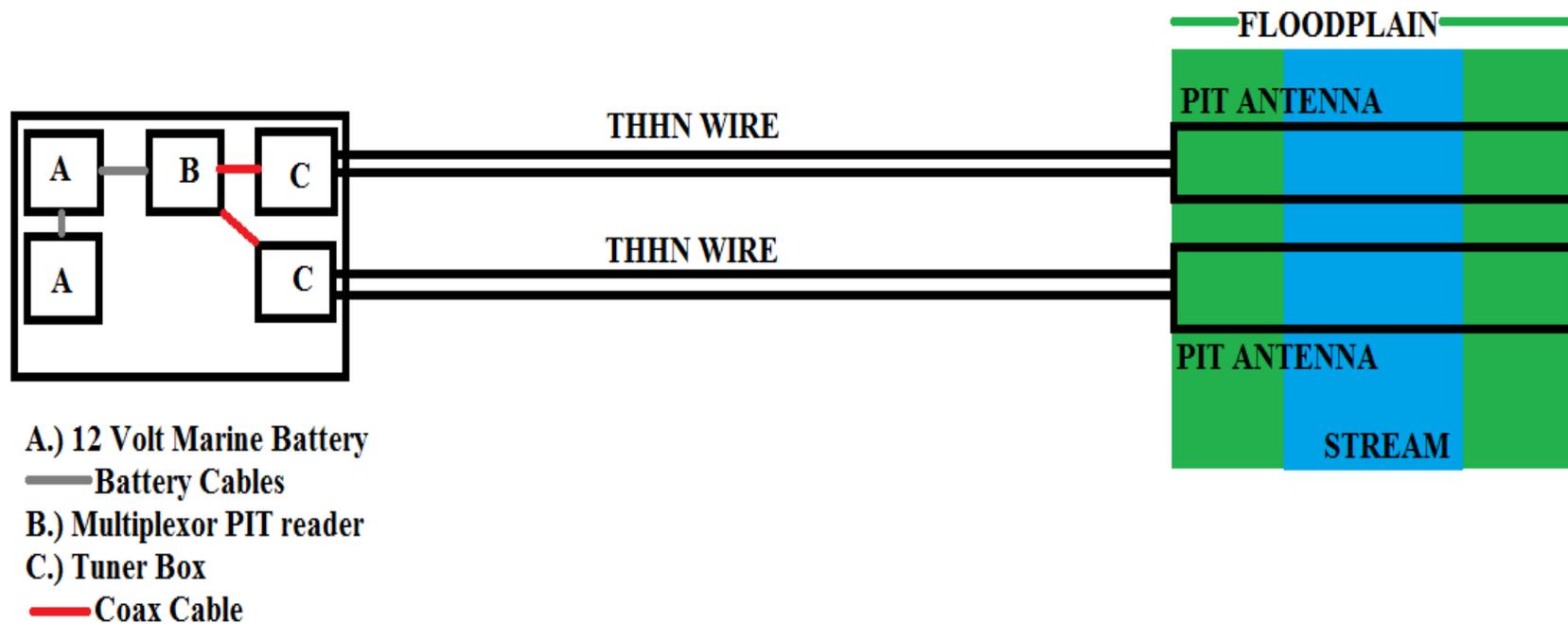


Figure 2.6: Static PIT antenna design for a single antenna at an antenna location (two antennae per location). PIT tag antennae were situated such that they included the entire stream width and any associated floodplain that may become inundated during typical high flows (i.e. seasonal runoff).

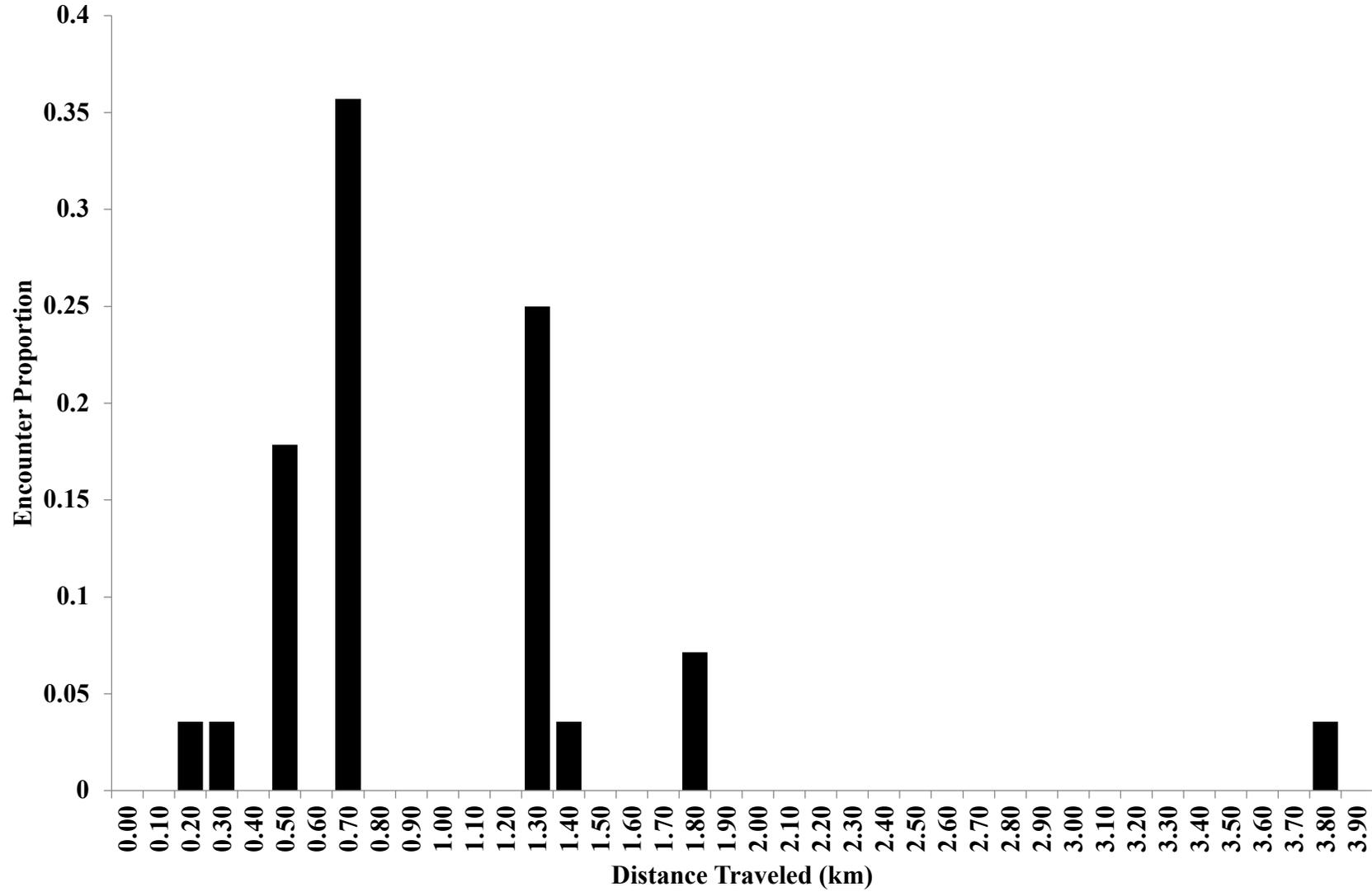


Figure 2.7: Encounter proportion as a function of Stonecat distance traveled (km) based on static PIT tag antenna data.

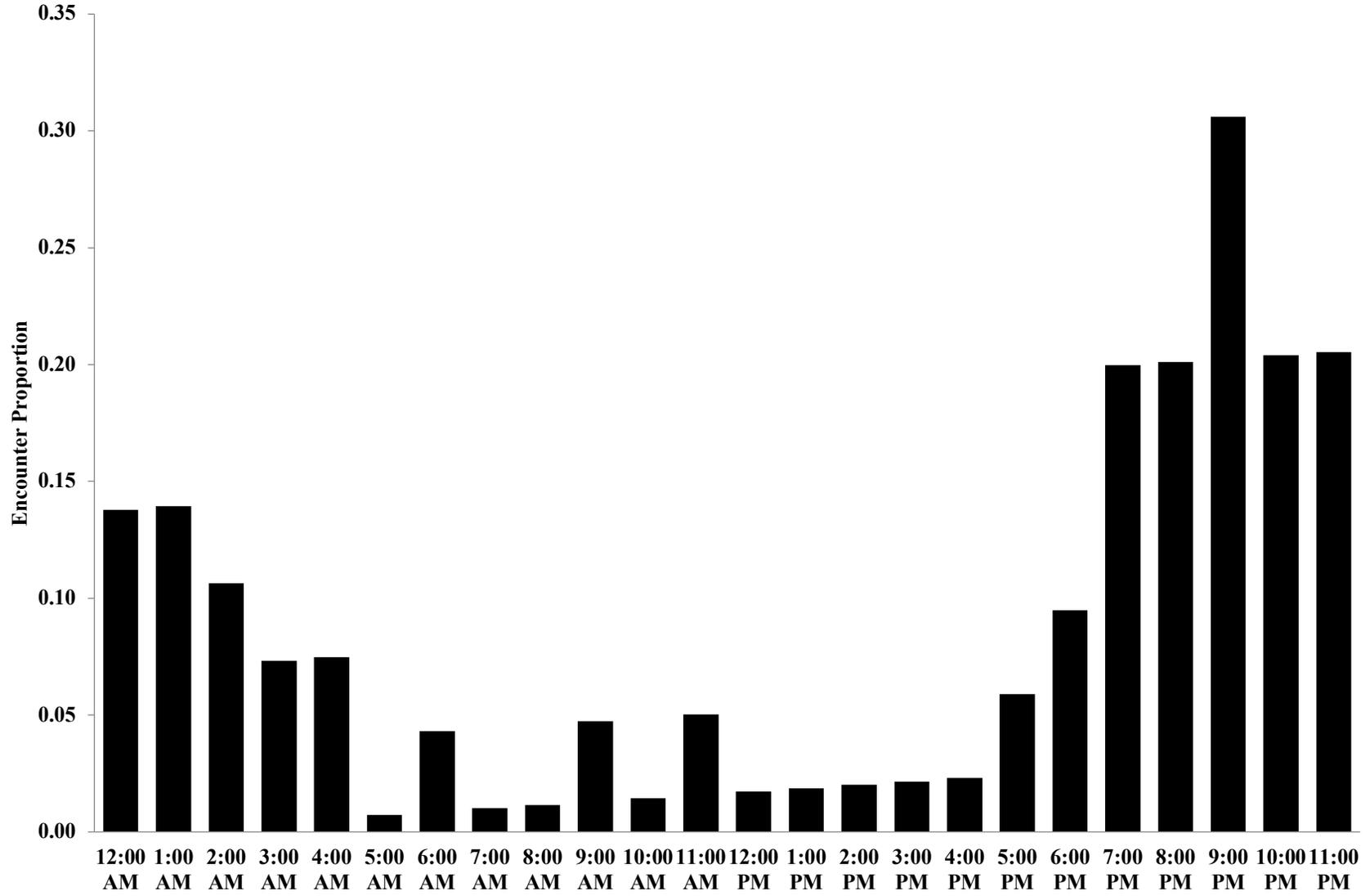


Figure 2.8: Encounter proportion of Stonecats by time of day based on static PIT tag antenna data.

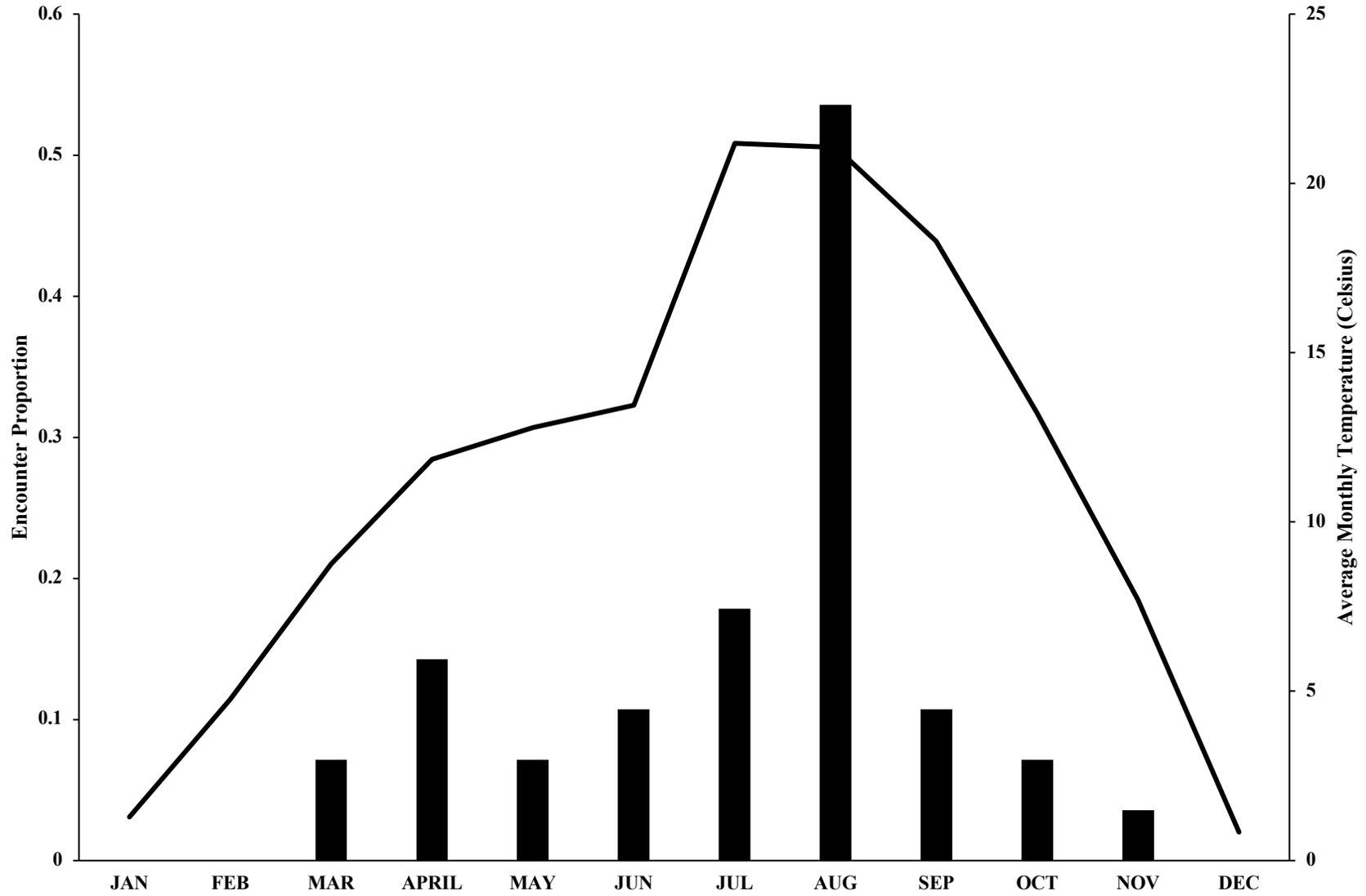


Figure 2.9: Encounter proportion of Stonecats by month of year with average monthly water temperature (Celsius) overlaid based on static PIT tag antenna data.

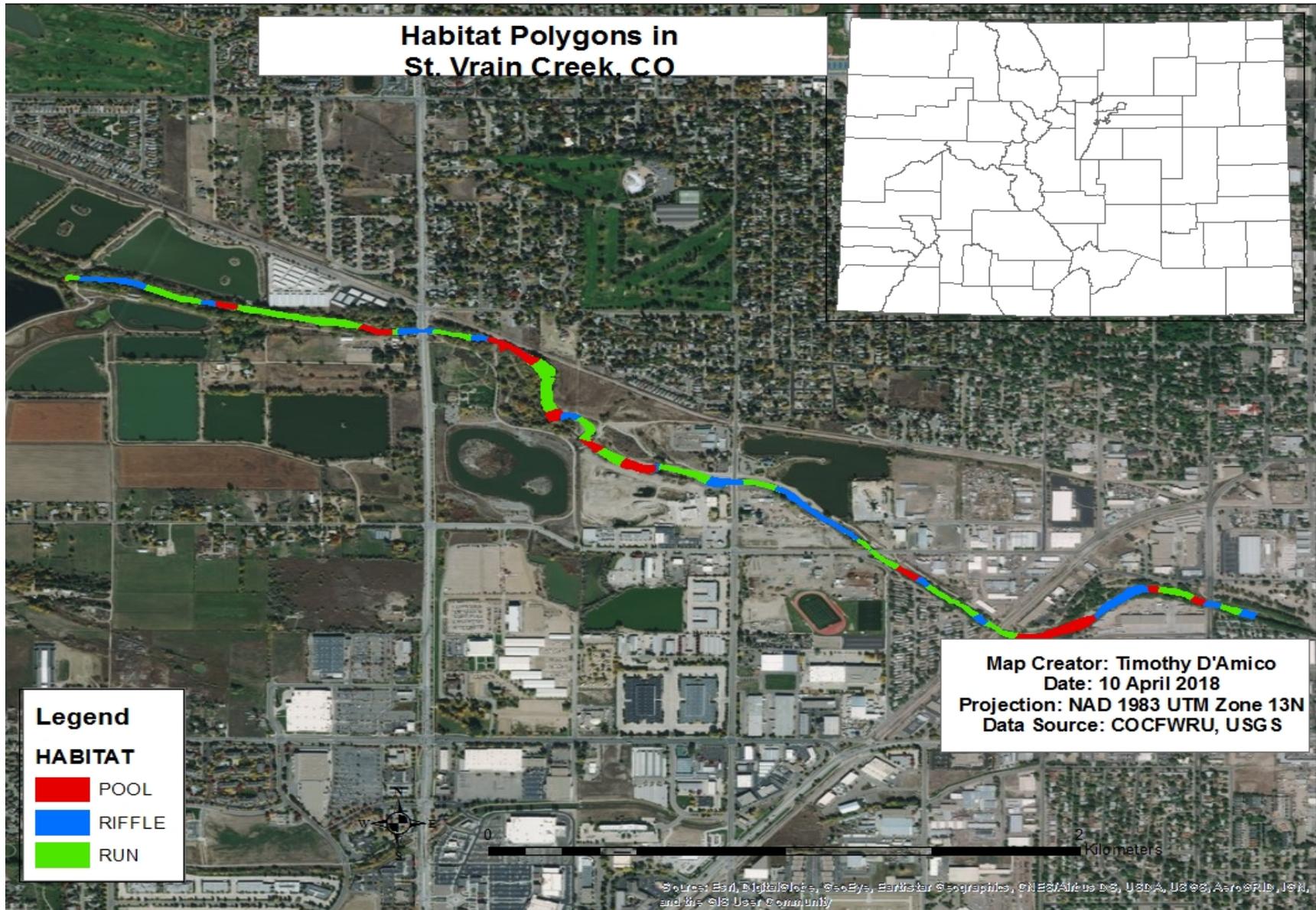


Figure 2.10: Meso-habitat polygons in St. Vrain Creek, CO.

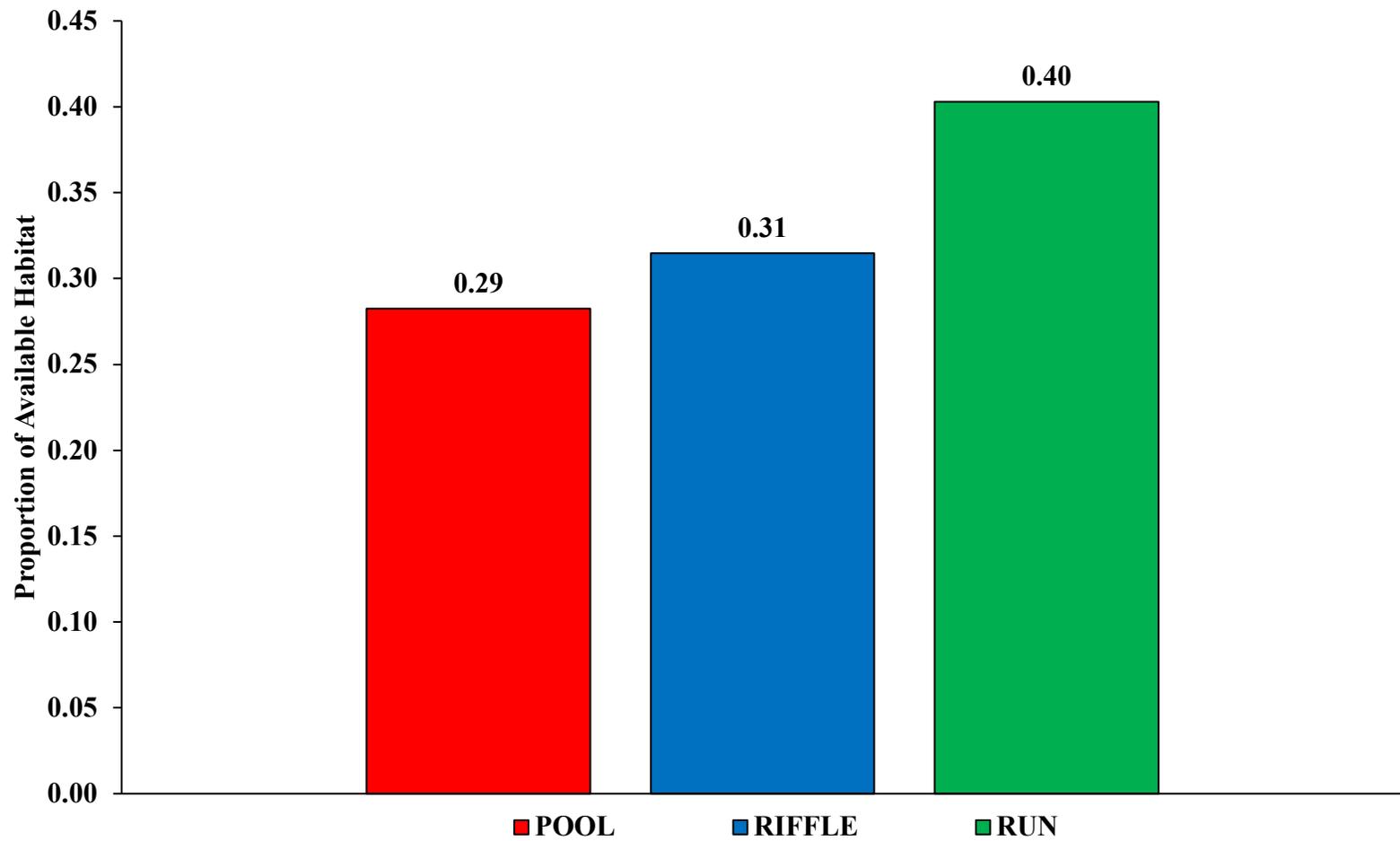


Figure 2.11: Proportion of available habitat by meso-habitat type in St. Vrain Creek, CO.

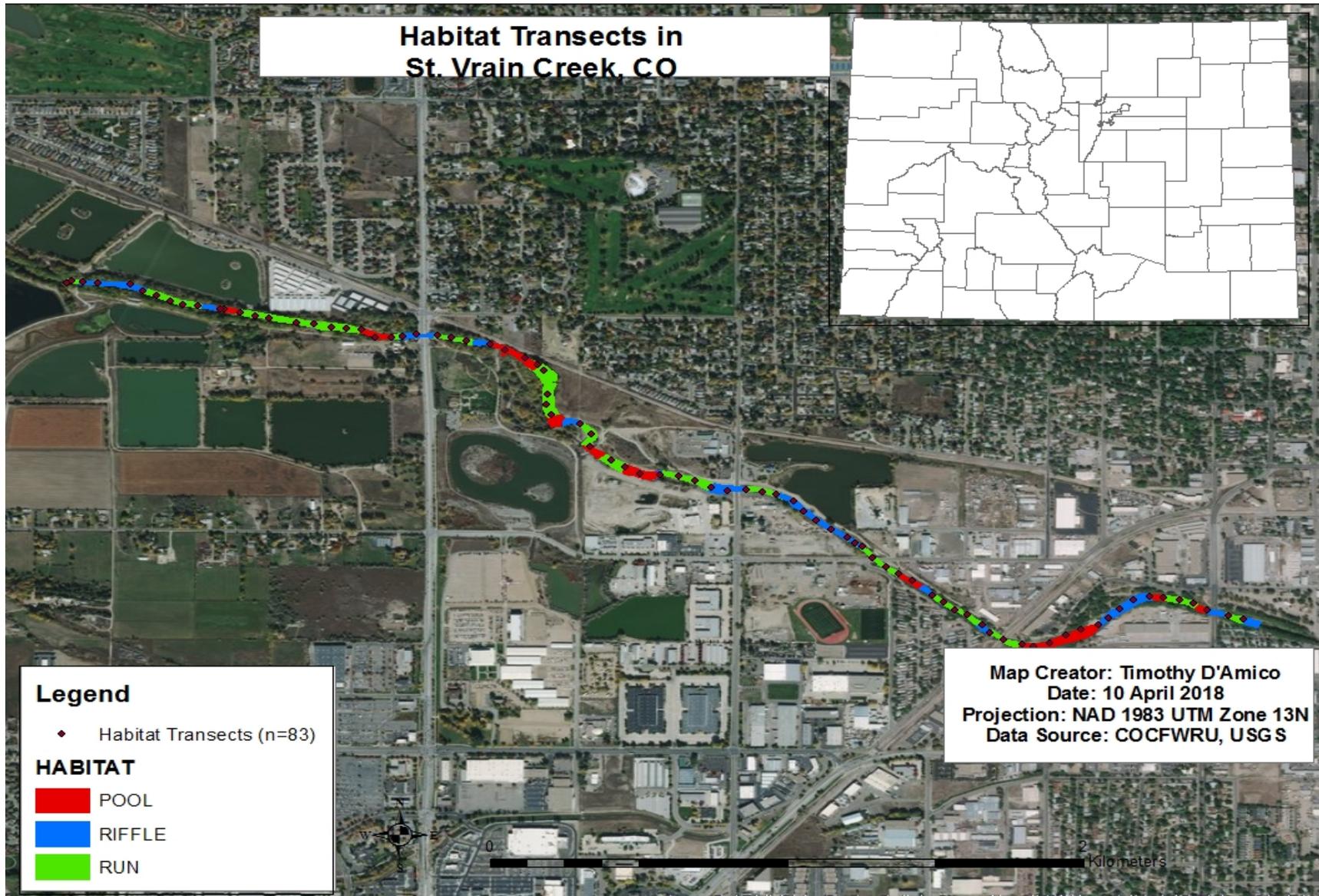


Figure 2.12: Meso-habitat polygons in St. Vrain Creek, CO with micro-habitat transect locations (n = 83) overlaid.

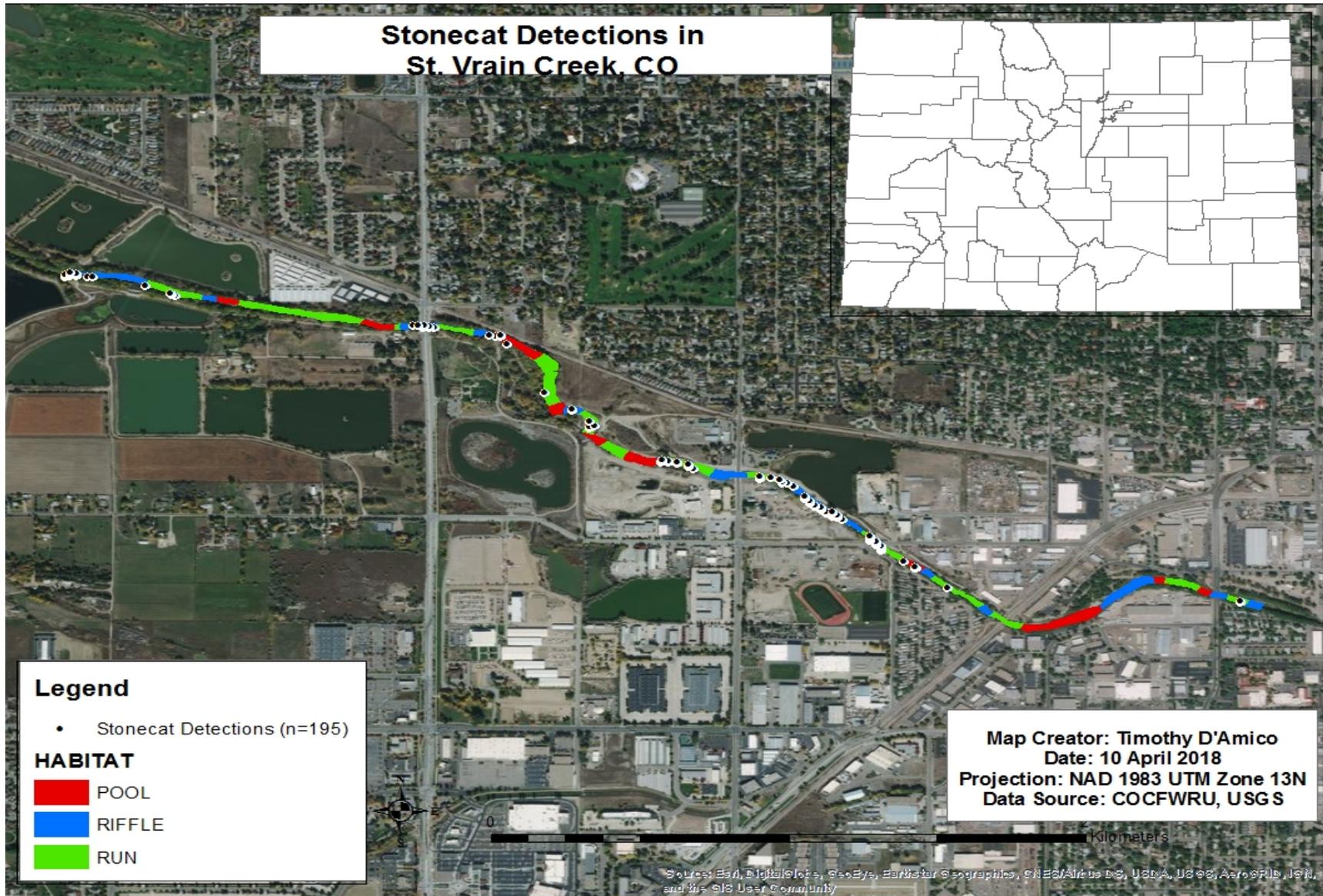


Figure 2.13: Meso-habitat polygons in St. Vrain Creek, CO with individual Stonecat detection locations (n = 195) overlaid.

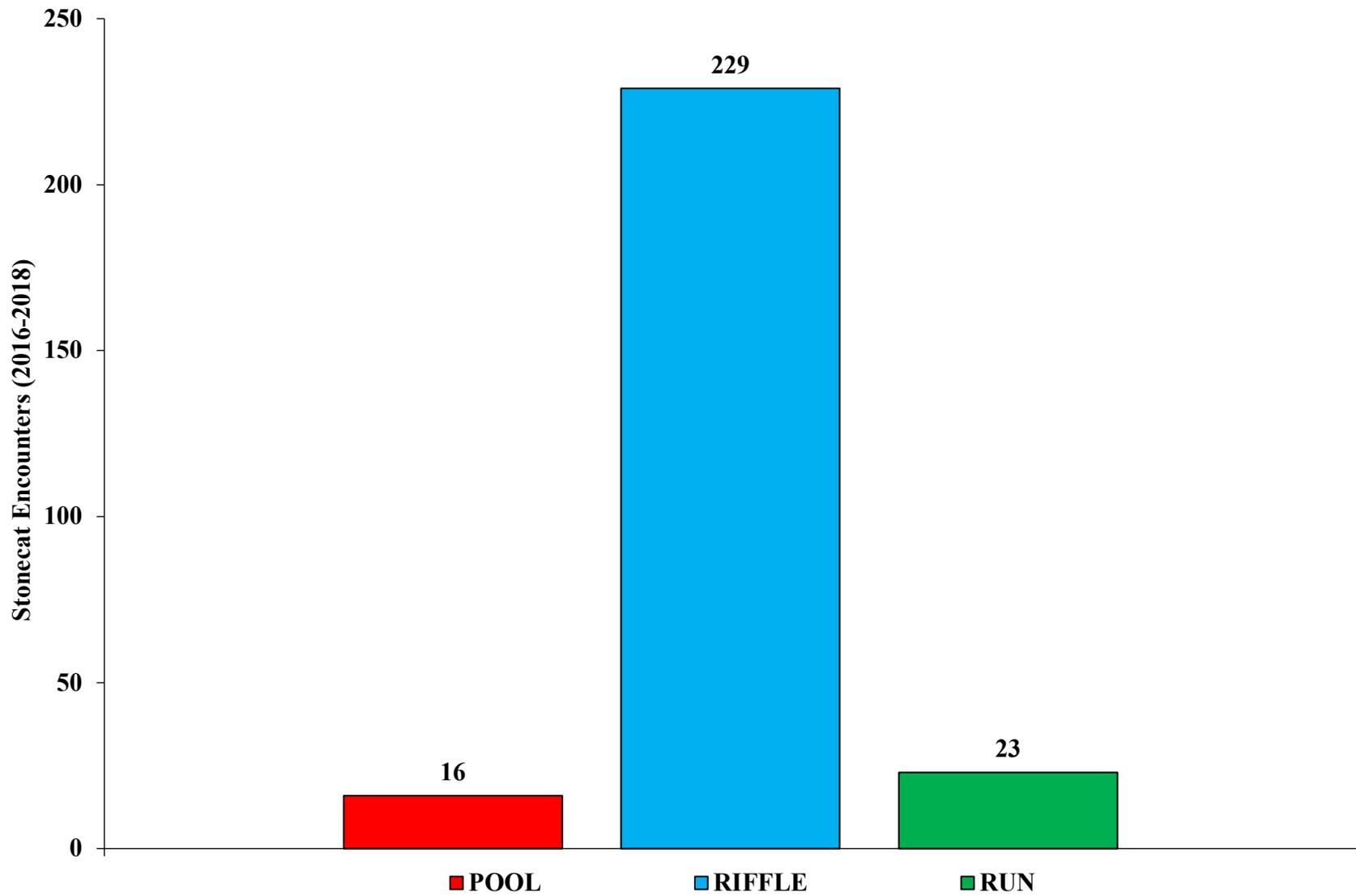


Figure 2.14: Stonecat encounters from mobile PIT antennae system for 2016-2018 by individual meso-habitat type.

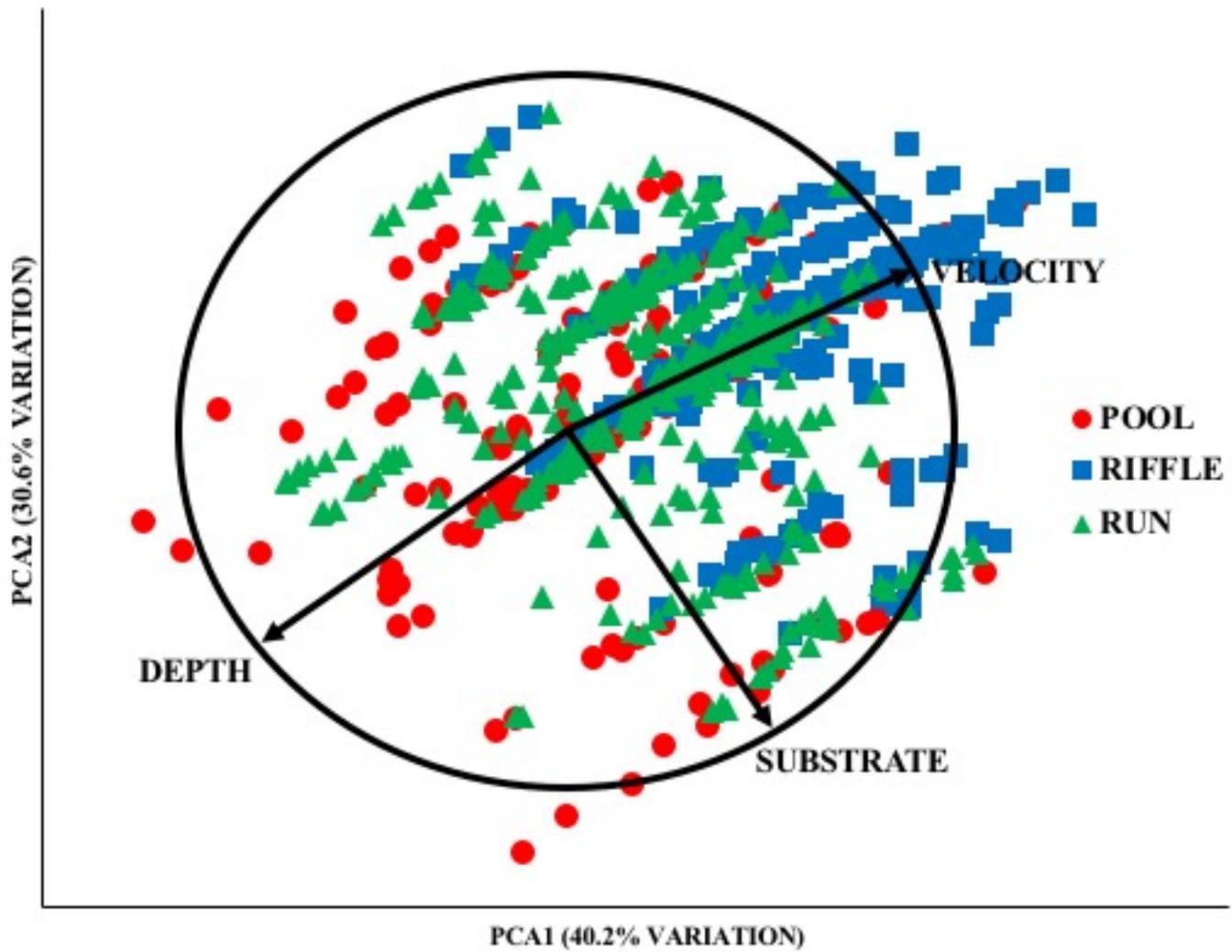


Figure 2.15a: PCA of meso-habitat type by micro-habitat characterization. The PCA1 axis explains 40.2% of the variation, while the PCA2 axis explains 30.6% of the variation.

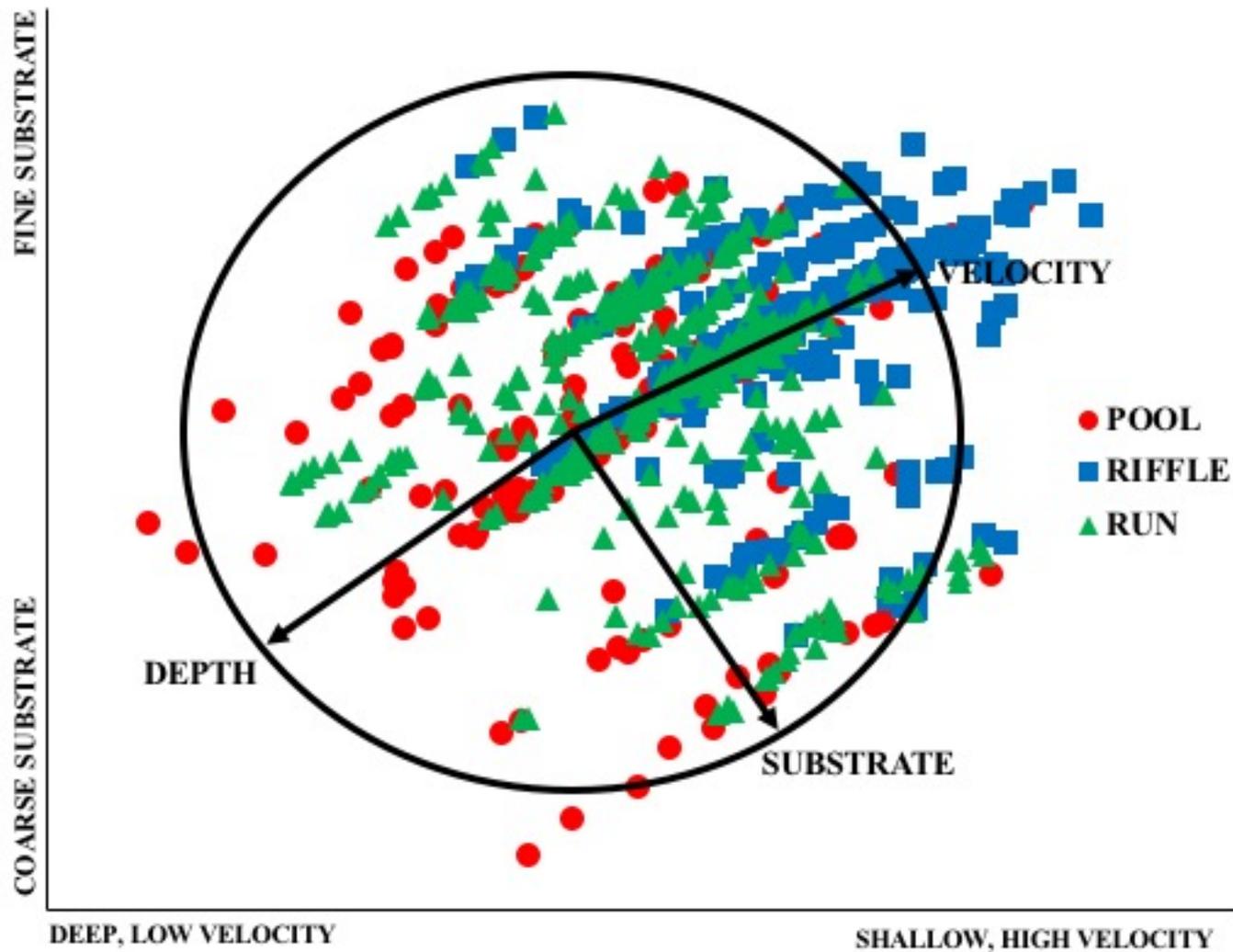


Figure 2.15b: PCA of meso-habitat type by micro-habitat characterization, now with micro-habitat drivers associated with PCA axis. PCA1 axis is driven by depth and velocity, while the PCA2 axis is driven by substrate.

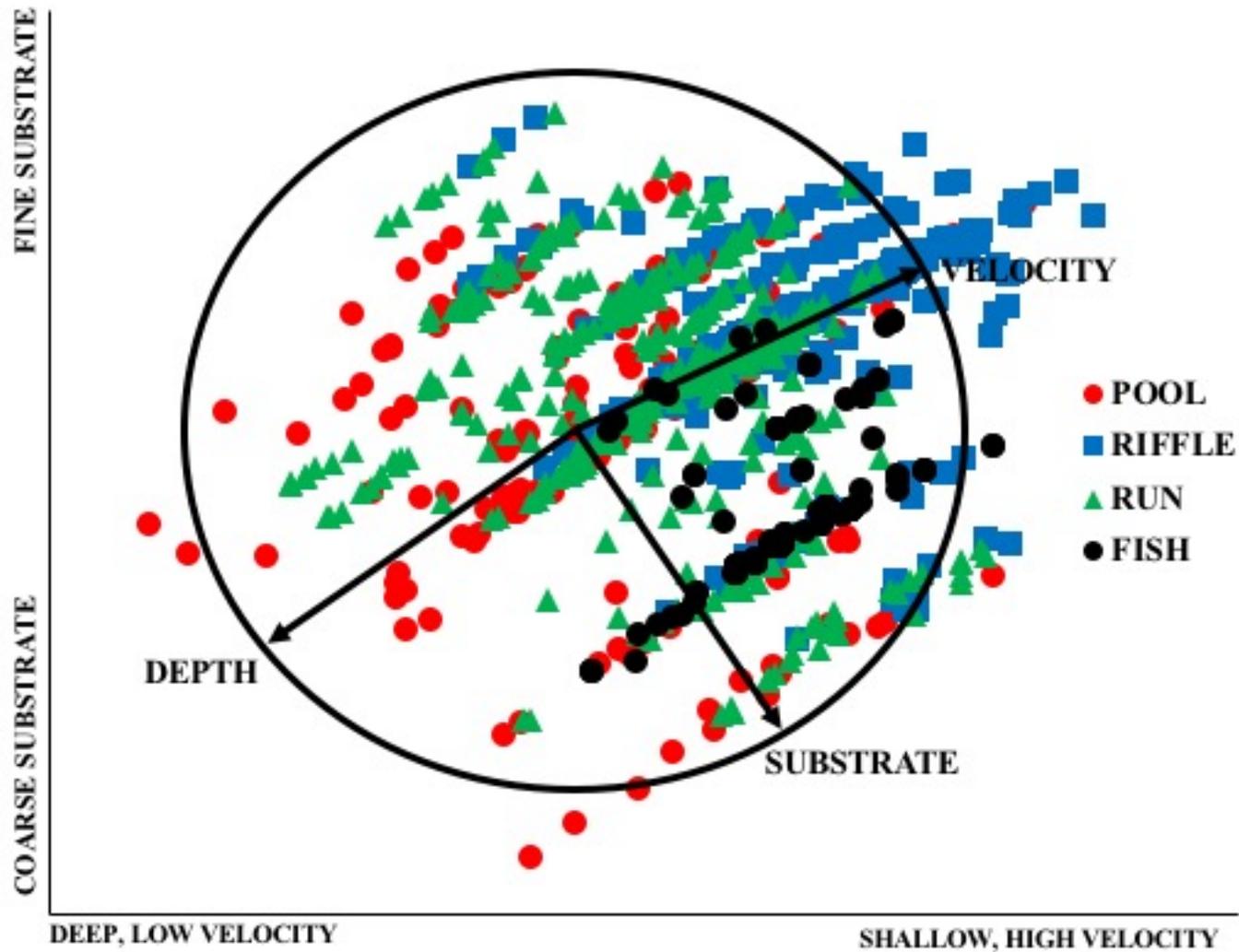


Figure 2.16: PCA of meso-habitat type by micro-habitat characterization, now with fish overlaid. As shown, Stonecats are found in shallower, high velocity habitat with coarse substrate.

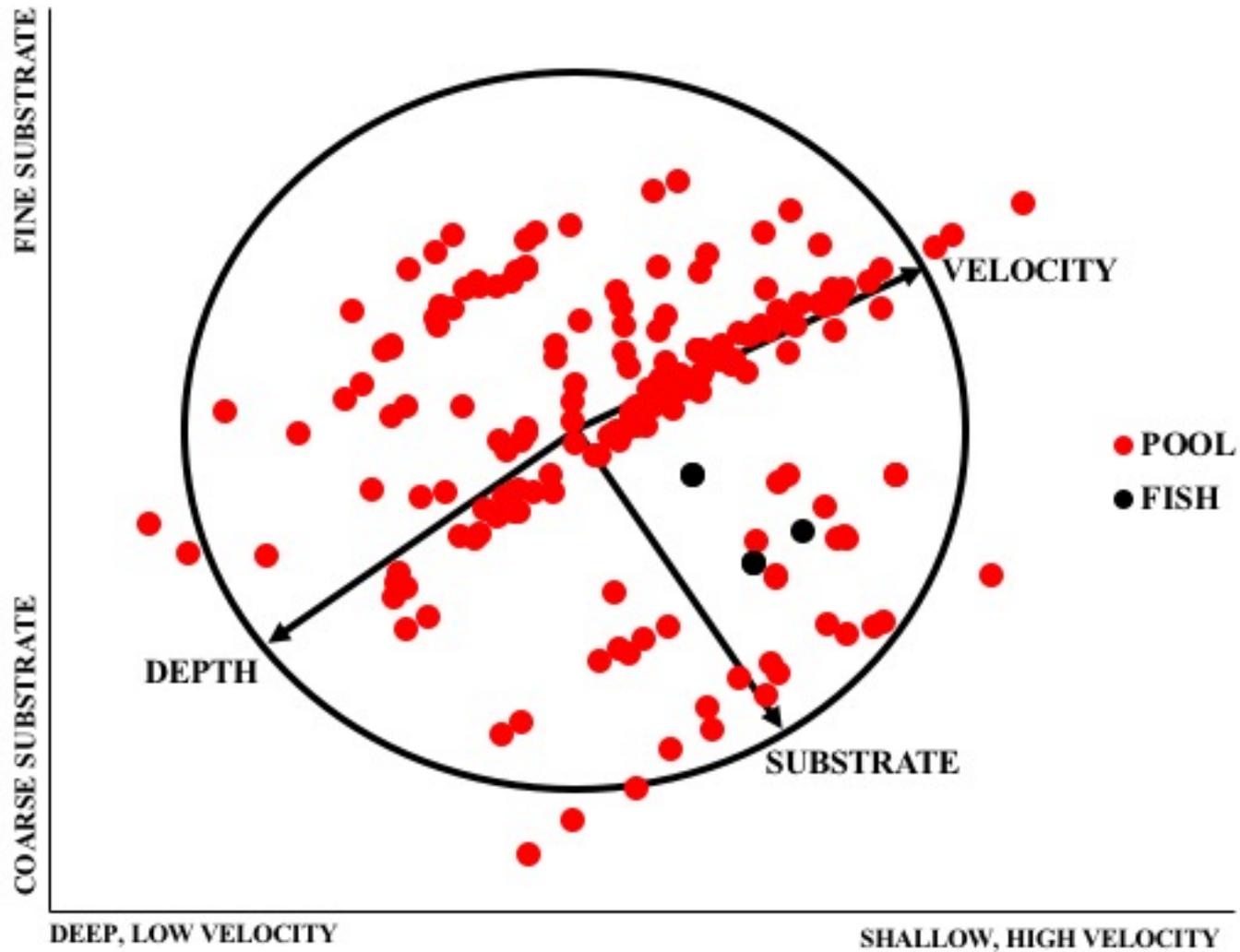


Figure 2.17a: PCA comparing Stonecat encounters within pool meso-habitat type. When I detected Stonecats within pools, they were associated with shallow, high velocity habitat (PCA1) and coarse substrate (PCA2).

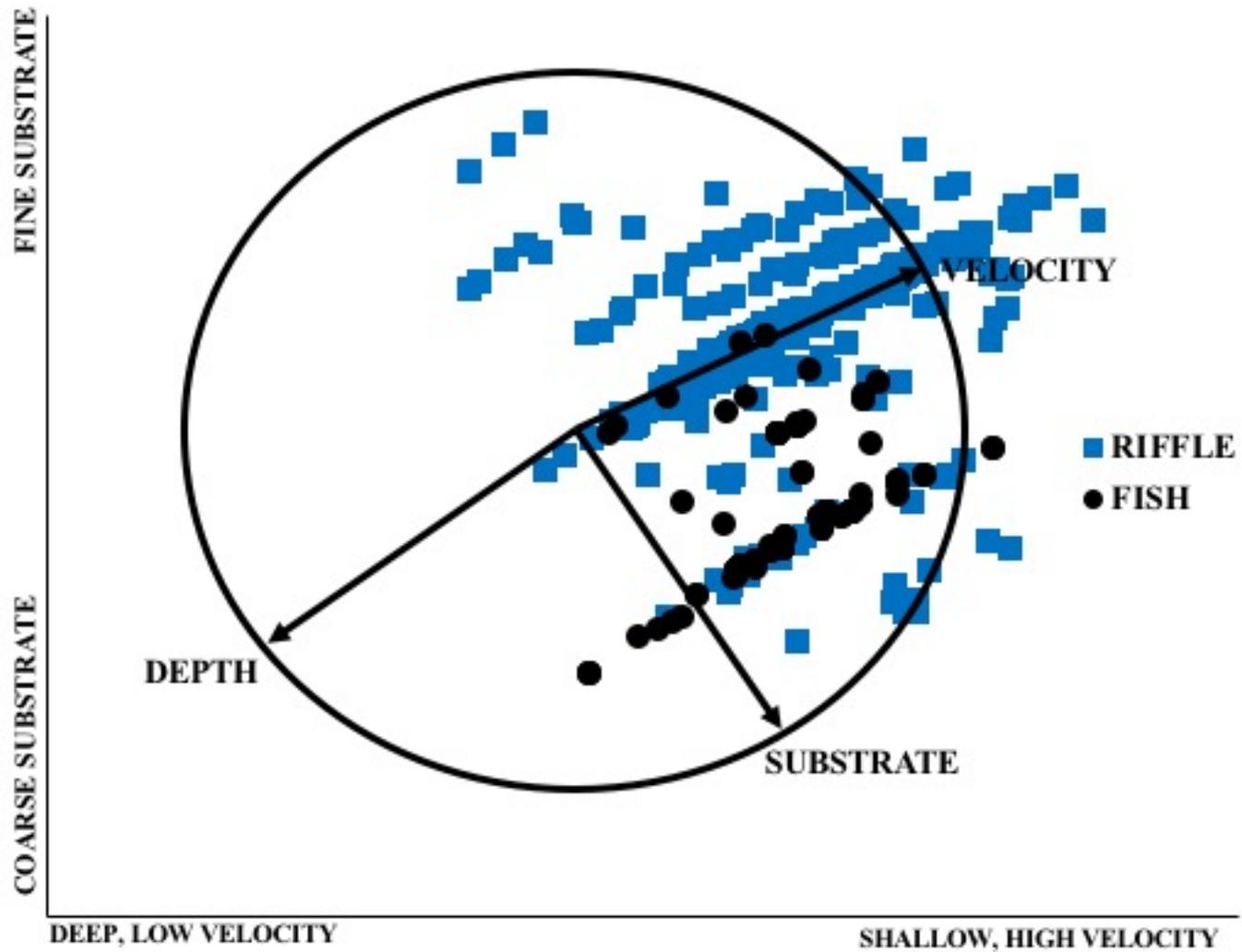


Figure 2.17b: PCA comparing Stonecat encounters within riffle meso-habitat type. When I detected Stonecats within riffles, they were associated with deeper, low velocity habitat (PCA1) and coarse substrate (PCA2).

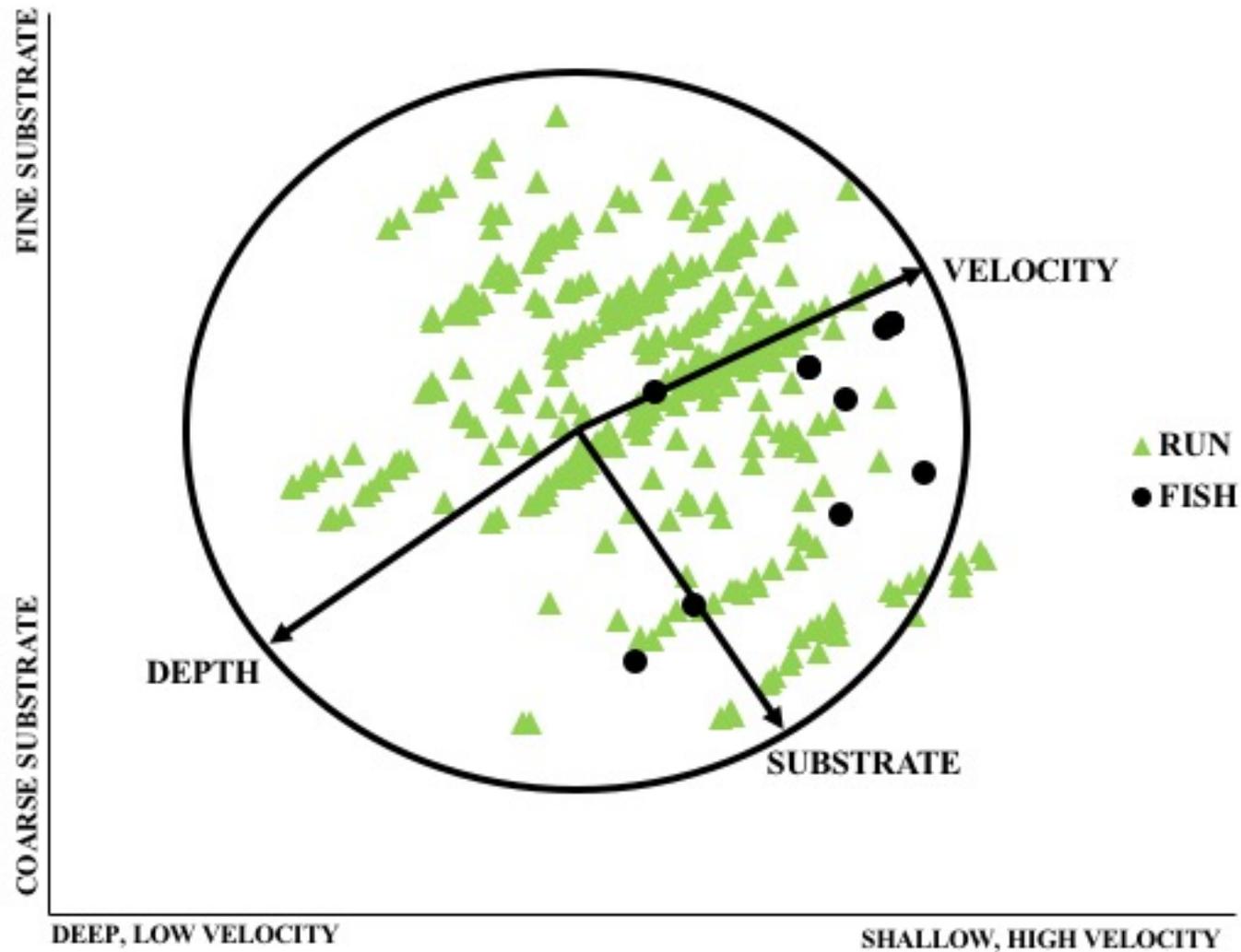


Figure 2.17c: PCA comparing Stonecat encounters within run meso-habitat type. When I detected Stonecats within runs, they were associated with shallow, high velocity habitat (PCA1) and coarse substrate (PCA2).

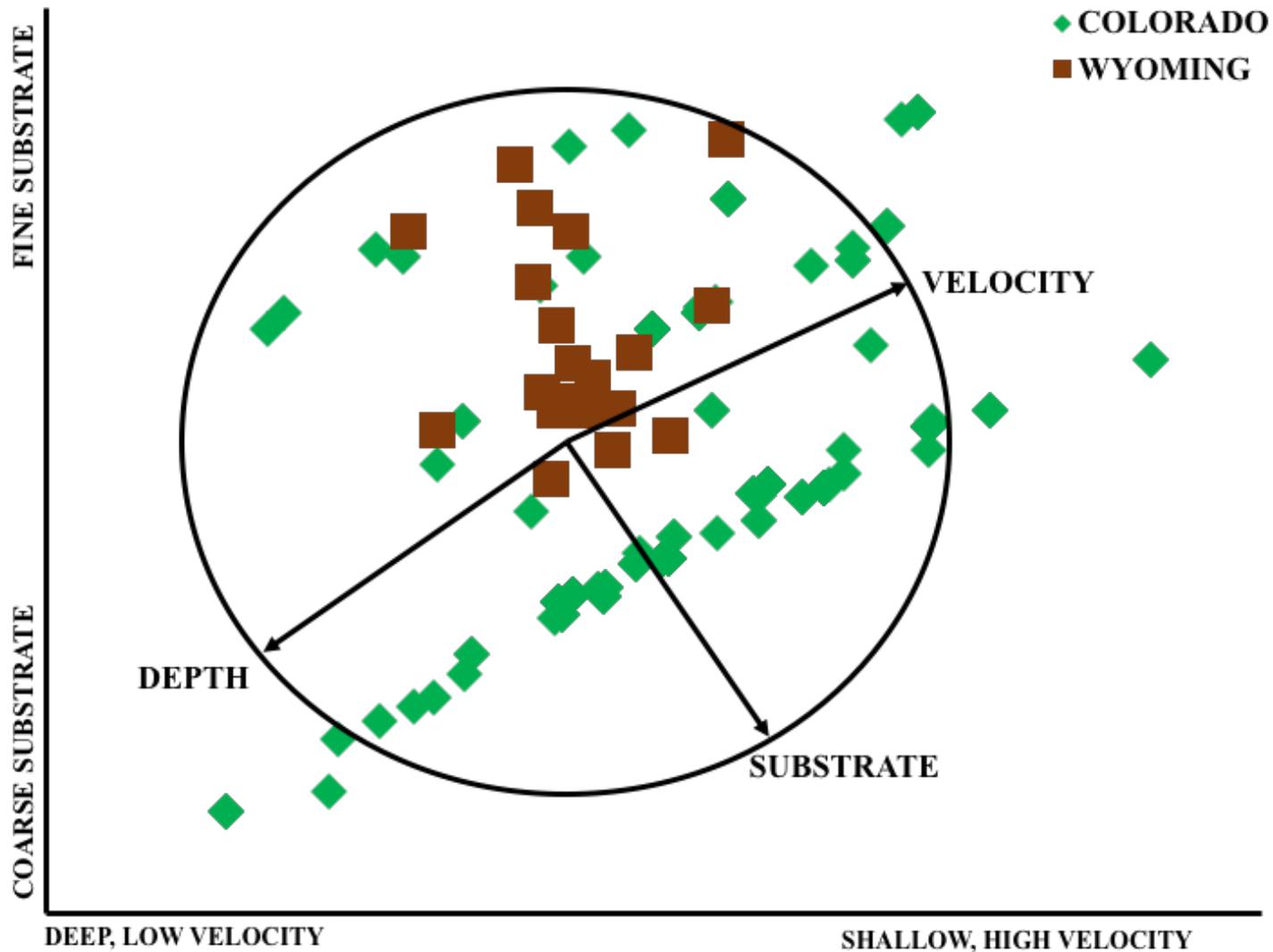


Figure 2.18: PCA comparing Stonecat encounters between Colorado and Wyoming. As shown, there is a high degree of overlap on the PCA1 axis in terms of velocity and depth. However, on the PCA2 axis, Colorado Stonecats are found in coarser substrate, which is that anthropogenic riprap.

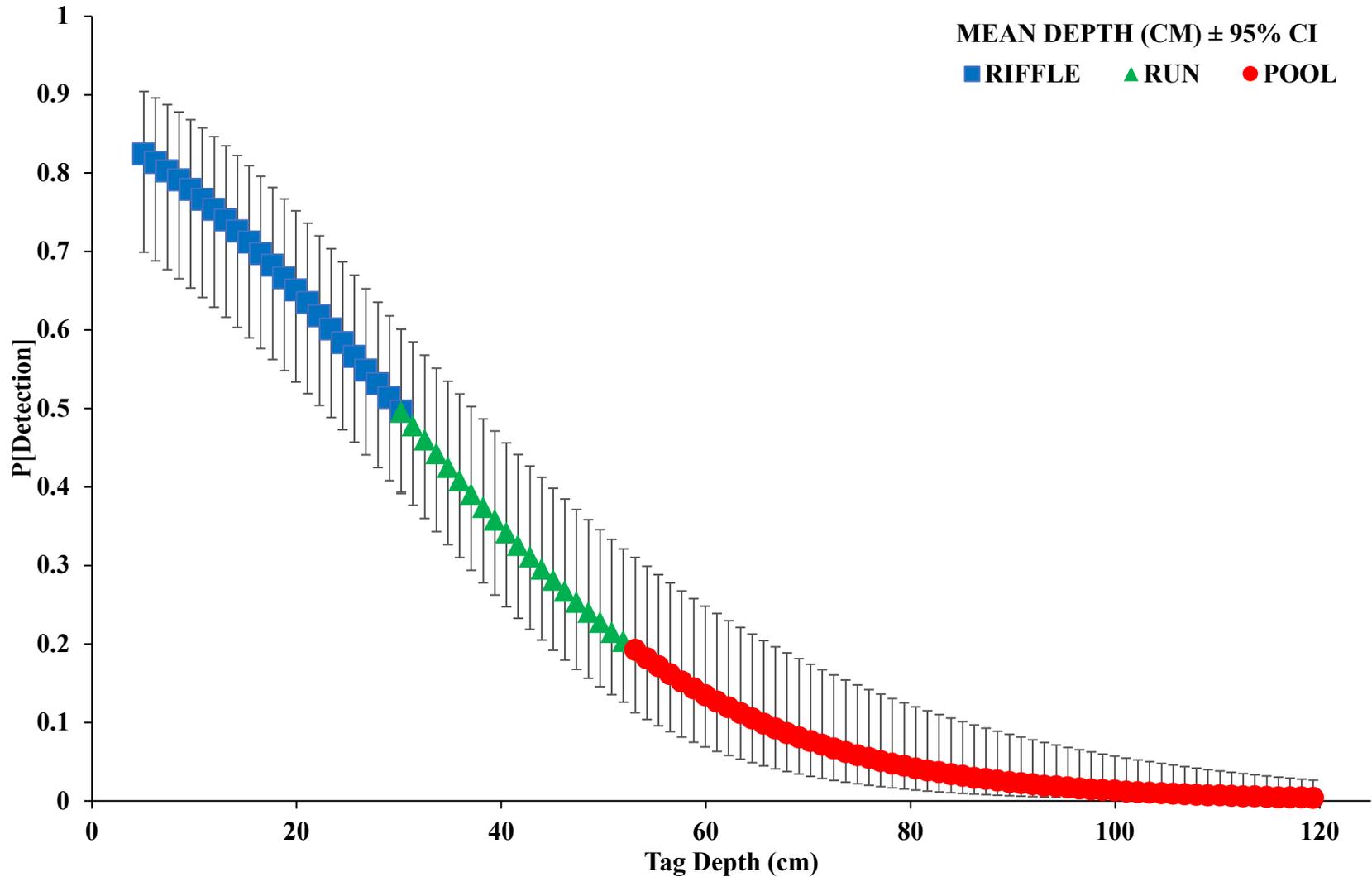


Figure 2.19: Detection probability as a function of depth for PIT tagged rocks with the mobile PIT antenna with mean depth (cm) and associated 95% confidence intervals of each meso-habitat type overlaid. As shown, detection probability decreases as tag depth increases. Furthermore, riffles have the highest detection probability followed by runs, and pools have the lowest detection probability.

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APPENDIX 1: ESTIMATING SURVIVAL

Introduction

From the mobile PIT tag data, I was able to not only characterize individual habitat selection preferences from the GPS locations of tag detection events, but I was able to estimate population demographic rates utilizing individual encounter histories in a Cormack-Jolly-Seber framework (Lebreton et al. 1992). Analysis in this framework allowed us to estimate apparent survival (ϕ) and detection (p).

Analysis

I analyzed the mobile PIT antenna data in a Cormack-Jolly-Seber model (Lebreton et al. 1992) in Program MARK utilizing the static PIT antennae to fulfill my physical closure assumption. I allowed apparent survival (ϕ) and detection probability (p) to vary by groups (summer vs. winter) as well as time (scan number 1-3). Since I evaluated known detection probability, I added this data as a third group to help inform how depth affects my parameter estimates. We also incorporated depth (cm) as a covariate. For fish that were detected, I used the depth at that individual mobile detection location. For fish that were not detected, I used an average depth from all the detections so not to bias the effect of the covariate.

Results

While I originally attempted to model survival across all four scans together, I quickly found that summer survival and over-winter survival needed to be modeled separately. My top model included apparent survival varying by season (summer vs. winter vs. known detection) and time (scan 1-3). The top model also included equal detection probability for all groups (summer, winter and known detection) and varying by time (scan 1-3). I included depth as a

covariate on both apparent survival and detection probability. My second-best model included apparent survival varying by season (summer vs. winter), time (scan 1-3) and modeled the known detection data the same as the fish. It also modeled detection probability the same for all groups, allowed it to vary by time (which pass in the multiscan survey) and included depth as a covariate on both apparent survival and detection probability. As shown, the third best model, without the depth covariate, has a $\Delta AICc > 82$ (Table A1). Based off my top model, I estimated summer monthly survival at 0.227 (95% CI: 0.177, 0.286) and monthly over-winter survival at 0.049 (95% CI: 0.027, 0.089). However, I found that apparent survival (ϕ) was affected by depth (beta estimate = -0.04; 95% CI -0.05, -0.03).

Discussion

My survival data indicate that summer apparent survival is higher than that in the winter (Figure A1, A2). I feel this is probably due to Stonecats moving into deeper pool habitat in the winter because shallow riffle habitat is less available because of low winter flows and ice formation. Furthermore, when I evaluate estimated apparent survival as a function of depth apparent survival decreases with an increase in depth and is more drastic during the overwinter period (Figure A1, A2). Brewer et al. (2006) suggested that Stonecats conducted pre-spawn activities in nearshore micro-habitats containing large substrates but probably moved to deeper habitats during summer to spawn. Their conclusion was based on a decrease in CPUE in shallow bedrock habitats but they did not directly sample deeper habitats. Brewer and Rabeni (2008) found little change in seasonal habitat use in Stonecats but did not sample deeper habitat. Apparent survival is the product of true survival, tag retention and study area fidelity (Burnham and Rexstad 1993). If I evaluate study area fidelity as shallow riffles (where the mobile antenna is able to detect them) and a temporary emigration to an undetectable state (beyond the read

range of the mobile PIT antenna), I feel that my apparent survival estimates are highly affected by inadequate sampling in deep pools. One way to estimate temporary emigration to deep habitats would have been to place static PIT antennae between riffle and pool habitats and monitor movement between habitat types. Unfortunately, based on my PIT antennae spacing and lack of encounters on the static PIT antennae, this was not possible.

Tables and Figures

Table A1: CJS model selection results for apparent survival.

Model	AICc	Delta AICc	AICc Weights	Model Likelihood	# of Parameters	Deviance
Phi(Season*Time,Rock.) p(t)}+DEPTH	984.01	0	0.64	1	11	961.68
Phi(Season*Time) p(t)}+DEPTH	985.18	1.1789	0.36	0.55	13	958.74
Phi(Season*Time) p(t)	1066.58	82.5738	0	0	12	1042.19
Phi(g*t) p(t)	1078.61	94.6079	0	0	21	1035.47
Phi(g*t) p(.)	1093.97	109.9619	0	0	19	1055.03
Phi(g*t) p(g)	1095.05	111.0415	0	0	24	1045.56
Phi(g*t) p(g*t)	1101.42	117.417	0	0	36	1026.07
Phi(t) p(g)	1117.49	133.4914	0	0	9	1099.28
Phi(g) p(g*t)	1124.82	140.8127	0	0	24	1075.33
Phi(t) p(g*t)	1125.03	141.0233	0	0	21	1081.88
Phi(g) p(t)	1125.03	141.0235	0	0	9	1106.81
Phi(.) p(g*t)	1166.02	182.0109	0	0	19	1127.08
Phi(t) p(.)	1182.79	198.7808	0	0	4	1174.74
Phi(t) p(t)	1182.85	198.8496	0	0	6	1170.75
Phi(g) p(g)	1183.71	199.7018	0	0	12	1159.33
Phi(.) p(g)	1189.21	205.2038	0	0	7	1175.07
Phi(g) p(.)	1200.86	216.856	0	0	7	1186.73
Phi(.) p(t)	1253.15	269.142	0	0	4	1245.09
Phi(.) p(.)	1354.74	370.7383	0	0	2	1350.73

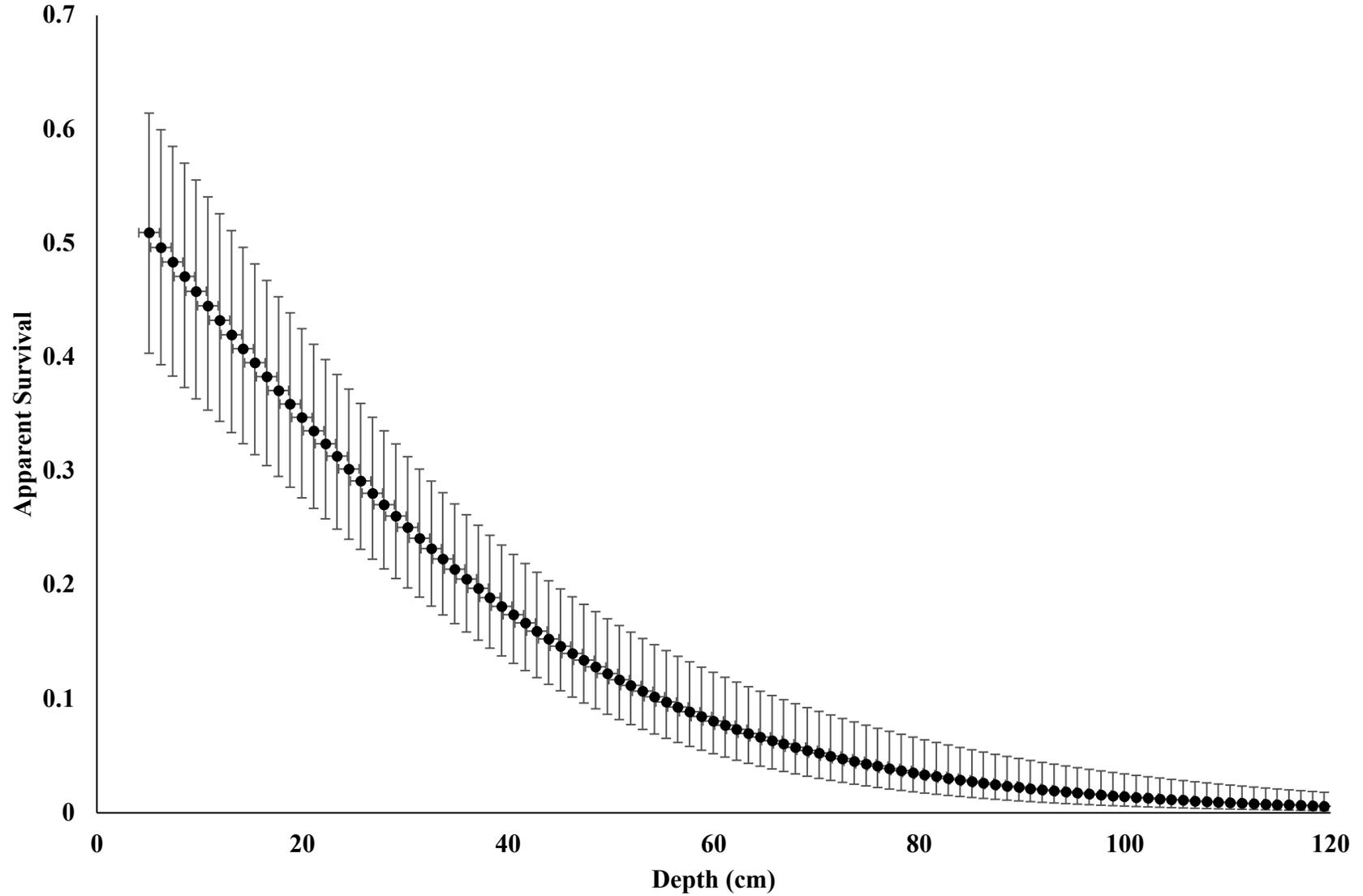


Figure A1: Apparent survival as a function of depth for Stonecats detected on the mobile PIT antenna during summers 2016, 2017. As shown, apparent survival decreases as tag depth increases.

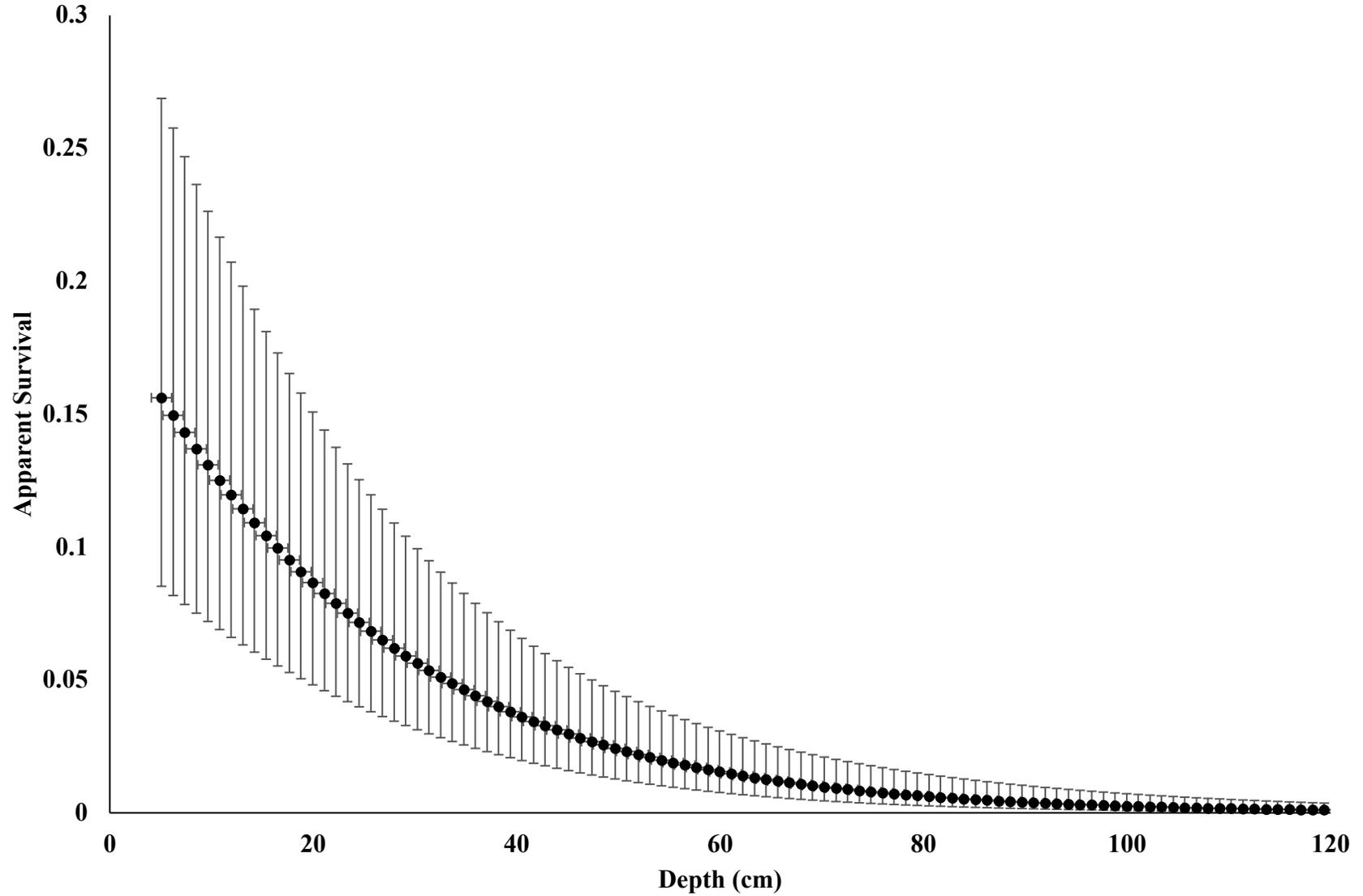


Figure A2: Apparent survival as a function of depth for Stonecats detected on the mobile PIT antenna during winters 2016, 2017. As shown, apparent survival decreases as tag depth increases.

APPENDIX 2: FISH PASSAGE

I was able to make a short-term evaluation (three weeks) of flood mitigation/whitewater park structures that have fish bypass channels included. I tagged Stonecats (n = 101) and placed them downstream of one of these structures. At the structure, I placed four static PIT antennas in the fish bypass channel to monitor passage progress, one static PIT antenna in the spillway (to confirm it was indeed a velocity barrier for Stonecats) and one static PIT antenna downstream (to establish physical closure). Although I only evaluated this on a very short time-scale and had limited encounters on the PIT antennas within the bypass structure, of the 19 Stonecat that entered the fish bypass channel, 60% of those (15/19) successfully navigated the entire bypass structure. Furthermore, the construction of the bypass channel was designed with large boulder substrate, which afforded habitat for Stonecats, which was manifested by numerous sequential detections of individual Stonecats at a single antenna within the bypass structure.