MOVEMENT AND FLOW-ECOLOGY RELATIONSHIPS OF GREAT PLAINS PELAGOPHIL FISHES

By

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Abstract: Alterations to freshwater systems are a leading cause of reduction in diversity and abundance of freshwater fishes. Pelagophils have been negatively affected by anthropogenic decreases in the frequency and magnitude of high-flow events and changes in baseflow conditions, but prescriptive flows based on pelagophil ecology are not available. Therefore, the goal of my thesis was to determine the flow components important for pelagophil persistence in the southern Great Plains. My first objective addressed how flow regime shapes pelagophil distributions over long periods of time by relating Arkansas River Shiner *Notropis girardi* and Plains Minnow *Hybognathus placitus* occupancy to flow regime components over cool and warm climatic periods. I found that pelagophils benefit from consistency in the number of high flow events and unpredictability in seasonal timing of high flow events. The relationships of Plains Minnow occupancy and predictability of seasonal flooding was dependent of the climatic regime, and may show that Plains Minnow is better able to adjust to warmer climate conditions. Disturbance and river fragmentation were negatively related to the occupancy of Arkansas River Shiner and Plains Minnow, respectively. To examine drivers of finer spatio-temporal dynamics and reveal the underlying factors affecting distributions, my second objective quantified movement of Arkansas River Shiner and Emerald Shiner *Notropis atherinoides* in the non-breeding season. Arkansas River Shiner and Emerald Shiner had a downstream movement bias in the winter and early spring. Long-distance upstream movements appear to be initiated in the spring by both species. I found evidence that a proportion of the Arkansas River Shiner population were residents by recapturing individuals in their tagging site after up to a year. If the goal is to conserve these species, water management strategies maintaining average annual (1983-2017) high-flow frequencies while promoting high-flow unpredictability may benefit these species. Land-use mitigation such as construction or recovery of wetland and riparian buffers may beneficial. Additionally, water management strategies improving river connectivity in late winter through the spawning season may benefit migratory individuals. However, improving lateral river connectivity in reaches with resident individuals (near river km 250) may increase recruitment from that proportion of the population.

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CHAPTER I

INTRODUCTION

Abiotic and biotic alterations to freshwater systems are leading causes of reduction in diversity and abundance of freshwater fishes. Although freshwater ecosystems account for only 0.01% of the Earth's water, > 6% of the world's species occupy these habitats, making freshwater rivers biodiversity hotspots (Strayer and Dudgeon 2010; Vörösmarty et al. 2010; Dudgeon 2019). More than 20% of all freshwater fishes are listed as threatened or endangered (Naiman et al. 2002; Collen et al. 2014; Arthington et al. 2016). There are at least five major threats to freshwater ecosystems: overexploitation, water pollution, flow modification, habitat degradation, and the introduction of non-native and invasive species (Dudgeon et al. 2006; Arthington et al. 2016). Flow modifications interact with the other threats often working synergistically to negatively affect freshwater ecosystems (Brook et al. 2008; Perkin et al. 2015b; Stoffels et al. 2018; Palmer and Ruhi 2019). For example, bankfull or higher discharge conditions maintain many channel features used as habitat by stream fishes (e.g., scouring of pools, deposition of sediment on floodplains, Gordon et al. 2004; Bray and Dunne 2017; Gibson and Shelley 2020). When flows are reduced, habitat complexity declines (Vanzo et al. 2016; White et al. 2018; Souza-Cruz-Buenaga et al. 2019), reducing available habitat for native species (Arthington et al. 2016; Mierau et al. 2018; Lear et al. 2019) while creating conditions often suitable for non-native species (Palmer and Ruhi 2019; Rogosch et al. 2019; Tonkin et al. 2020). Unfortunately, flow alteration of rivers is common worldwide and is a major contributor to the decline of freshwater fishes.

Globally, flow regimes (i.e., long term discharge patterns) of rivers are affected by a variety of human alterations including dams and direct water extraction. More than half of the world's rivers are dammed (Nilsson et al. 2005; Lehner et al. 2011; Best 2019) and damming disproportionately affects large rivers (Tharme 2003). More than half of our available surface water is appropriated (Tharme 2003; Abbott et al. 2019b, 2019a) with some reservoirs and rivers being over-appropriated via water permitting (Brewer et al. 2016; Perkin et al. 2017; Abbott et al. 2019b). Further, groundwater pumping has changed flow patterns by depleting deep alluvial aquifers that contribute to base flows (Pringle and Triskat 2000; Fox et al. 2011; Fan et al. 2013; de Graaf et al. 2019). Over 75% of discharge of northern hemisphere rivers is strongly or moderately affected by flow alteration (Dynesius and Nilsson 1994; Tharme 2003; Lehner et al. 2011). Altered flow patterns affect a variety of physicochemical factors that directly and indirectly affect aquatic biota (e.g., sediment transportation, Milhous 1998; Poff and Zimmerman 2010; habitat composition, Poff and Allan 1995; Lake 2000; Bêche et al. 2006; water quality and temperature, Poff et al. 1997; Hughes 2005; Carlisle et al. 2016).

Although several attributes of fish ecology are altered due to flow modifications (Poff et al. 2007; Poff and Zimmerman 2010; Tonkin et al. 2018), changes in the ability of fishes to move and the corresponding changes to distributions and abundances are disproportionately affected by water withdrawals. River fragmentation by man-made barriers can reduce movements by stream fishes and prevent completion of the life cycle (Perkin et al. 2015b; Rodeles et al. 2020; Zambaldi and Pompeu 2020). Movement reduction may be related to a variety of factors including swimming endurance (Leavy and Bonner 2009; Knapp et al. 2019; Schumann et al. 2019), jumping ability (Prenosil et al. 2016; Rahel and McLaughlin 2018; Schumann et al. 2019), and simply lack of connected habitat (Marshall et al. 2016; Neufeld et al. 2018; Schumann et al. 2019). For example, reduced river fragment length has been related to the decline and extirpation of pelagic-broadcast spawning species (see overview by Worthington et al. 2018) (e.g., Plains Minnow *Hybognathus placitus*, Arkansas River Shiner *Notropis girardi* (hereafter ARS), Perkin and Gido 2011; Perkin et al. 2015a; Sharpnose Shiner *Notropis oxyrhynchus*, Smalleye Shiner *Notropis buccula,* Wilde and Urbanczyk 2013). Flow reductions can decrease river connectivity longitudinally (Bunn and Arthington 2002; Cooke et al. 2016; Grill et al. 2019), laterally (Cooke et al. 2016; Stoffels et al. 2016; Shao et al. 2019), and vertically (Ward 1998; Cooke et al. 2016; Grill et al. 2019). Loss of connected longitudinal habitat can result in incomplete migrations (Brönmark et al. 2013; O'brien et al. 2019; Rodeles et al. 2019), reduced recruitment (Falke et al. 2010; Humphries et al. 2020; Stuart and Sharpe 2020), reduced biodiversity (Liermann et al. 2012; Shao et al. 2019; Rodeles et al. 2020), and decreased population persistence (Perkin and Gido 2011; Wilde and Urbanczyk 2013; Hopper et al. 2020). Alternatively, river-floodplain connectivity increases available food sources (Junk et al. 1989; Stoffels et al. 2015; Arantes et al. 2019), provides spawning and rearing habitat for a variety of fishes (e.g., Alligator Gar *Atractosteus spatula*, Kluender et al. 2017; Common Carp *Cyprinus carpio,* Jones and Stuart 2009; Smallmouth Buffalo *Ictiobus bubalus* and Black Crappie *Pomoxis nigromaculatus*, Koel and Sparks 2002), and provides flow refuge to fishes during extreme floods (Schwartz and Herricks 2005; Bolland et al. 2012; Arantes et al. 2019). Long term or repeated drying of stream segments may prevent recolonization by fishes, resulting in reduced abundances or truncated ranges (Jonsson 1991; Perkin et al. 2015b; Worthington et al. 2018). Improving flow conditions may be a key to improving conditions for many threatened and endangered freshwater species, including pelagic-broadcast spawning fishes.

Freshwater pelagic-broadcast spawning fishes (hereafter pelagophils) belong to a declining reproductive guild of freshwater fishes of the Great Plains, including the federally-threatened ARS. Pelagophils comprise about twenty small-bodied minnows thought to migrate long distances to spawn (Durham and Wilde 2008a; Worthington et al. 2014a; Chase et al. 2015) by releasing non-adhesive semi-buoyant eggs and sperm into pelagic areas of rivers (Moore 1944; Platania and Altenbach 1998; Worthington et al. 2018). The propagules (i.e., eggs and larvae) require long reaches of river because they drift in suspension during development to the free swimming stage (Moore 1944; Platania and Altenbach 1998; Dudley 2004; Durham and Wilde 2008a; Perkin and Gido 2011; Worthington et al. 2014a). The required length of river for complete development and maintenance of stable populations was estimated at 458 ± 137 river km by Perkin and Gido (2011) and 468–592 km by Albers and Wildhaber (2017).The current distributions of freshwater pelagophil species are truncated compared to their historical ranges (reviewed by Worthington et al. 2018). The ARS is an emblematic pelagophil fish due to its threatened listing status. Originally endemic to the Arkansas and Canadian river basins of AR, OK, and TX, current remaining native populations are limited to two segments of the South Canadian River (Worthington et al. 2014b). Remnants of a native populations might remain in the Cimarron River (Bestgen et al. 1989; Wilde 2002), though it is considered unlikely (Mollenhauer et al. 2018).

Several impediments prevent development of meaningful conservation and management strategies for freshwater pelagophils. First, to complete the life cycle, pelagophils are thought to migrate upstream long distances to spawn; however, direct evidence is lacking. Based on otolith microchemistry Pecos Bluntnose Shiner (*Notropis simus pecosensis*) includes both migratory and resident subsets of populations (Chase et al. 2015). Moreover, there are no data available on migration timing and relevant data are based on just a few recaptures during summer (Wilde 2016). Research attention has focused primarily on the late-spring and summer spawning seasons. We lack information on overwintering and movements across seasons. To further complicate matters, flow conditions that allow adult fish movements and successful development of offspring, are critical to completion of the life cycle. Pelagophils have been negatively affected by anthropogenic decreases in the frequency and magnitude of high-flow events (Bonner and Wilde 2000; Worthington et al. 2014b, 2016) and changes in baseflow conditions (Perkin et al. 2015b, 2019; Worthington et al. 2016), but prescriptive flows based on pelagophil ecology are not available. Understanding relationships between pelagophil presence and longer-term flow patterns during both the breeding and non-breeding season would facilitate a better understanding of the flows needed to support pelagophils. Examining movement behavior during winter and leading up to the spawning period is a critical information need. Therefore, the overarching goal of my thesis is to determine the flow components important for pelagophil persistence in the southern Great Plains (Figure 1) thereby aiding the development of conservation actions for pelagophil fishes.

I will accomplish my goal via two research objectives:

1. Determine the relationships between flow regime characteristics and occurrence of two pelagophil fishes; and
2. Quantify movement of two pelagophil fishes during the non-breeding, winter season.

My first objective addresses how flow regime shapes pelagophil distributions over long periods of time, whereas my second objective builds on the first by examining the drivers of finer spatio-temporal pelagophil dynamics and revealing the underlying factors affecting their distributions.

CHAPTER II

FLOW AND LANDSCAPE METRIC RELATIONSHIPS WITH THE OCCUPANCY OF TWO PELAGOPHIL FISHES ACROSS COOL AND WARM CLIMATIC PERIODS

ABSTRACT

Anthropogenic flow alteration is hypothesized to contribute to the decline of pelagophils within the Great Plains; however, the influence of flow conditions on riverine fishes can be context dependent. Therefore, my objective was to determine relationships between flow and landscape metrics in cool and warm climatic regimes and river segment occupancy of Arkansas River Shiner *Notropis girardi* (ARS) and Plains Minnow *Hybognathus placitus*. I used existing fish assemblage data collected from the species ranges within Oklahoma, Texas, and New Mexico and discharge data from U.S. Geological Survey stream gages to examine river segment occupancy. I compared average annual temperature to the long-term average over the employed years (1983-2017) and identified a cool regime (1983-1995) and a warm regime (1998-2017). I selected flow metrics representing five major flow-regime components (magnitude, frequency, duration, timing, and rate of change) that I hypothesized to be ecologically important to pelagophils. I included annual precipitation and a land-use disturbance index to account for general range restrictions and other human perturbations. Distance to the nearest upstream dam was included to account for river-fragment length. Using a hierarchical backward selection process, I developed an occupancy model to determine metrics important to species occupancy. Occupancy was negatively related to higher predictability of seasonal flooding during the cool regime for both

species; however, Plains Minnow showed a positive relationship with flood predictability during the warm regime. This reversal suggests Plains Minnow may be better able to adjust to warmer climate conditions. Plains Minnow had a strong negative relationship with variability in frequency of high-flow events over both regimes, despite pelagophils being considered tolerant of flow variability. Plains Minnows occupancy was positively related to larger river-fragment lengths. ARS occupancy was not related to fragment length in my study area but there was high variation around this coefficient. There was a negative relationship between land-use disturbance and ARS occupancy. If the goal is to conserve these species, water management strategies maintaining average annual (1983-2017) high-flow frequencies while promoting high-flow unpredictability may benefit these species. Land-use mitigation such as construction or recovery of wetland and riparian buffers may be beneficial.

INTRODUCTION

The influence of flow alterations (i.e., changes to long term discharge patterns) and responses to prescriptive flows by riverine fishes can be context dependent. For example, as the magnitude of flow alteration increases, there tend to be general reductions in the abundance and diversity of aquatic organisms (Poff and Zimmerman 2010); however, the extent of the changes observed varies by species, habitat, and physicochemical conditions (Poff and Zimmerman 2010; Walters 2016; Rosenfeld 2017; VerWey et al. 2018). As a generic example, decreases in abundance, growth, and reproduction are typically less extreme in more generalist species compared to species classified as specialists (Walters 2016; Rosenfeld 2017). Perkin et al. (2019) found pelagic-broadcast spawning fishes experienced recruitment failure during extreme drought conditions, but fishes using other reproductive modes did not. The availability and quality of habitat can mediate or exacerbate the effects of flow alteration on fishes (Walters 2016). The presence and connectivity of refuge habitat in altered systems increases fish resilience by allowing dispersal and recolonization and improving species survivorship and redundancy (Magoulick and Kobza 2003; Walters 2016; Van Looy et al. 2019). Species responses to flow alterations may also vary with other physicochemical conditions (e.g., temperature). For example, habitat offering thermal refuge becomes more important as stream temperatures increase (Aunins et al. 2015; Ebersole et al. 2020). Thus, flow alterations that reduce thermal refuge availability are more harmful to fishes when thermal regimes are also altered (Ebersole et al. 2020). Understanding how threatened and endangered species respond to changing flows under context-dependent conditions will allow us to develop more informed conservation actions under a variety of conditions.

Managing discharge patterns can improve physicochemical conditions for many stream fishes. Streamflow management sometimes attempts to mimic natural flow patterns and has been successful in increasing native fish abundances. For example, native fish density increased in the San Juan River, Colorado after dam releases were structured to mimic the timing of spring snowmelt (Propst and Gido 2004). In Lower Putah Creek, California, the range and abundance of native fishes increased following changes in flow magnitude to mirror historical patterns (Marchetti and Moyle 2001; Kiernan et al. 2012). In many cases, streamflow management benefits native fishes indirectly through restoration of critical habitats (e.g., connecting floodplains, Koel and Sparks 2002; Bowen et al. 2003) and by reducing non-native species abundance (Propst and Gido 2004; Marks et al. 2009; Kiernan et al. 2012). Moreover, improvements to flow conditions can both improve in-channel water temperature conditions (Santiago et al. 2016; Bair et al. 2019) and help sustain groundwater flows beneficial to some species (Perkin et al. 2017; Grover 2019; Mollenhauer et al. 2020). Native biodiversity increases with prescriptive flows based on natural flow patterns (Poff and Zimmerman 2010; Yarnell et al. 2015), but flows that mimic natural patterns can be difficult to achieve in highly altered landscapes (Poff 2018).

Effective flow prescriptions can be challenging in heavily altered regions where river channels can no longer support historical flow patterns (Jackson and Pringle 2010; Brewer et al. 2016; Conallin et al. 2018). Flow patterns have changed over time due to groundwater pumping (Falk et al. 2011; Scanlon et al. 2012), agricultural water use (Stohlgren et al. 1998; Scanlon et al. 2012; Maupin et al. 2014), dams (Graf 2001; Poff et al. 2007; Poff and Olden 2017), and other diversions (Milhous 1998; Perkin and Gido 2011). In conjunction with changing flow patterns, extensive ecosystem changes (e.g., climate; land use) over long temporal scales may not be reversible (Roberts et al. 2013; Perkin et al. 2015b). For example, historical floodplains of many rivers are urbanized for multiple uses (Tockner and Stanford 2002; Tockner et al. 2009; Brewer et al. 2016) and historical flow prescriptions could create a human health hazard. Altered flow patterns have reshaped channels, often reducing the capacity of rivers to carry water and sediment (Brewer et al. 2016; McManamay et al. 2016; Van Appledorn et al. 2019). Emblematic examples of channel miniaturizing are Great Plains rivers that have become narrow and incised since damming (Friedman et al. 1998; Brewer et al. 2016; Worthington et al. 2018); these rivers are not able to convey the historical flow volume (see Fig. 3, Brewer et al. 2016). Water availability in semi-arid regions such as the Great Plains is not sufficient to match historical flow volumes due to over allocation of water permits (Dodds et al. 2004; Perkin et al. 2017) making alternative flow prescriptions necessary (Acreman et al. 2014; Poff 2018). Using flow prescriptions that promote fish persistence over space and time may be necessary to recover species of conservation concern and be a more achievable goal than attempting to restore the full complement of historical flow patterns (Tharme 2003; Anderson et al. 2006; Poff 2018).

Freshwater pelagophils (i.e., a reproductive fish guild that reproduces in open water and relies on drifting propagules, Worthington et al. 2018) of the Great Plains were once considered emblematic Great Plains species tolerant of harsh environmental conditions (Matthews 1988; Matthews et al. 2005; Worthington et al. 2018), but now many of these species are of conservation concern (Worthington et al. 2018). Freshwater pelagophils have wide tolerances to temperature, dissolved oxygen, salinity, and suspended solids (Ostrand and Wilde 2001; Worthington et al. 2018). Great Plains river conditions have changed significantly over the past century because of land-use disturbances and climatic changes (Perkin and Gido 2011; Perkin et al. 2015a; Worthington et al. 2018). These changes have led to declines in pelagophil abundances and truncated distributions (Hoagstrom et al. 2011; Worthington et al. 2014b, 2018). Five taxa of Great Plains pelagophils are currently listed as federally threatened or endangered in the United States: Rio Grande Silvery Minnow *Hybognathus amarus*, Smalleye Shiner *Notropis buccula*, Arkansas River Shiner *Notropis girardi* (hereafter ARS), Sharpnose Shiner *Notropis oxyrhynchus*, and Pecos Bluntnose Shiner *Notropis simus pecosensis* and two taxa are extinct: Phantom Shiner *Notropis orca* and Rio Grande Bluntnose Shiner *Notropis simus simus* (Jelks et al. 2008; Worthington et al. 2018). Others are of conservation concern (e.g., Prairie Chub *Macrhybopsis australis;* Jelks et al. 2008; Worthington et al. 2018) or their status is unknown (Worthington et al. 2018).

Changes to the flow regime, combined with drought and other processes of fragmentation, are hypothesized to be a leading cause of the decline in pelagophils (Bonner and Wilde 2000; Worthington et al. 2014b, 2018; Perkin et al. 2015a). Yet, prescriptive flows to benefit pelagophils are generally lacking except for the federally endangered Rio Grande Silvery Minnow (Platania and Dudley 2003; U.S. Fish and Wildlife Service 2016; Valdez et al. 2019). Climate is one of the primary drivers of species distributions (Hynes 1975; Comte et al. 2013) and species tolerances and flow needs may differ in prolonged periods of cooler versus warmer temperatures. Therefore, my first thesis objective was to determine relationships between pelagophil occupancy and flow and landscape metrics during cool and warm climatic regimes. I hypothesized flow-ecology relationships for freshwater pelagophils would vary with climate and landscape factors. Specifically, I was interested in seven pelagophil fishes of the Great Plains

(Worthington et al. 2018): ARS, Plains Minnow, Speckled Chub *Macrhybopsis aestivalis*, Smalleye Shiner, Rio Grande Shiner *Notropis jemezanus*, Sharpnose Shiner, and Bluntnose Shiner *Notropis simus*. Although each of these species was identified in ≥ 20% of surveys within their range from relatively recent fish surveys in the southern Great Plains (see methods), only ARS and Plains Minnow had enough surveys for the modeling process.

METHODS

Study Area

My study area was the southern portion of the Great Plains ecoregion (EPA Level I, Commission for Environmental Cooperation Working Group 1997) including New Mexico, Oklahoma, and Texas (hereafter Southern Great Plains; Figure 1). The climate of the southern Great Plains is variable, with a west to east annual precipitation gradient from 55 cm to 98 cm (Woods et al. 2005). Southern Great Plains streams are relatively wide and shallow with braided channels dominated by sand or mud substrates (Matthews et al. 2005). These streams are characterized by extreme flood and drought events (Matthews 1988; Dodds et al. 2004). Stream water temperatures vary closely with ambient temperatures and intermittent channels are common in western reaches during dry periods (Woods et al. 2005). All large rivers and many smaller rivers of the region are impounded (Matthews et al. 2005; Roberts 2015). The dominant land use of the Southern Great Plains is agriculture where irrigation is common, but dewatering is also extensive (Woods et al. 2005) where water is extracted for human use (e.g., Oklahoma City). Alterations to these systems have resulted in dramatic declines in abundance and truncated distributions of native fish populations such as pelagophils (Perkin et al. 2015a, 2017; Worthington et al. 2018).

Fish Assemblage Data and Spawning Period

I compiled and organized existing fish assemblage data from 10 databases (Table 1). I retrieved data from six online databases covering my study area. I also obtained data from three fish databases covering portions of Texas from The Nature Conservancy. Lastly, I included fish data where the species were identified by and data compiled by the Brewer lab at Oklahoma State University, where the U.S. Fish and Wildlife Service conducted sampling in both Oklahoma and Texas (Table 1; Figure 3). Data were organized in an occupancy modeling framework where sites contained spatially replicated surveys with replacement (Kendall and White 2009) conducted within a defined sampling season (*see* next section). I considered discrete sampling events at a site within the same sampling season to be a survey and all surveys within a sampling season at a site to collectively be a sample. Surveys without coordinates were georeferenced in ArcMap (10.2.1, ESRI, Red Lands, California) using location descriptions. Sampling conducted outside of the study area, incomplete data (no date or location information), and duplicate events were removed from the database (Figure 1). I identified duplicate sampling by locating data within close proximity (4 km) in ArcMap and manually inspecting each database to determine if data should be combined as a unique sampling event or omitted (duplicate data). Each survey in the resulting database was assigned to its referenced site using ArcMap. Capture histories were developed for each site and sampling season where surveys included detection (1) and non-detection (0) events for each species. I developed capture histories only for sites within the documented historical range of each species (Worthington et al. 2018).

Surveys were assigned to either the spawning or non-spawning period to account for movement associated with reproduction (i.e., meet the closed system assumption, MacKenzie et al. 2002, 2009). A large portion of pelagophil populations is thought to migrate long distances to spawn (Durham and Wilde 2008a; Worthington et al. 2014a; Chase et al. 2015), making a closed-system assumption unreasonable across the spawning and non-spawning periods. Surveys conducted May through August were assumed to reflect spawning fish, whereas fishes sampled

October through March were considered non-spawning fishes. I omitted data from April and September, treating them as representing transitions to spawning and non-spawning periods, respectively. Southern Great Plains pelagophils begin gonad recrudescence in April (e.g., Plains Minnow, Taylor and Miller 1990; ARS, Bonner 2000) and spawning may continue into September (Bonner 2000).

Sampling seasons

My study period was split into two sampling seasons (MacKenzie et al. 2005) based on climatic regimes to account for changing temperature patterns over time. I compared average annual temperature to the long-term average (i.e., 1895-2018) to determine cool and warm climatic regimes where flow-ecology relations might differ. Average annual temperatures and the long-term average were retrieved from the National Oceanic and Atmospheric Administration (NOAA National Centers for Environmental information 2019). A cool climatic regime occurred 1983–1995 (hereafter cool season) and a warm climatic regime occurred 1998–2017 (hereafter warm season) (Figure 2). I considered 1996–1997 a transitional period and did not include these data in my analyses.

Site Selection

I selected sites based on spatial changes in discharge and available U.S. Geological Survey stream gage data. As with similar efforts in arid and semi-arid regions, modeling daily discharge within the Great Plains has proved problematic for developing flow-ecology relationships (*see* Worthington et al. 2019); therefore, I relied on stream gages for flow data. My sites were stream segments containing a U.S. Geological Survey stream gage with at least 20 years of flow data (Gan et al. 1991; Olden and Poff 2003). Each stream segment was delineated using the shorter distance of either fifth order tributary confluences (Strahler 1957) or a 20-river km (rkm) segment extending upstream and downstream 10 rkm from an associated stream gage.

These distances were chosen by examining differences in discharge spatially between stream gages and treating ≤ 15% as acceptable for recognition of a stream segment. The cutoff at 15% was based on preliminary calculations of percent change in discharge between gages at varying distances from one another and with confluences of differing stream orders. Sites intersected by a major dam (National Anthropogenic Barrier Dataset, Ostroff et al. 2013) were omitted. In three cases, two stream gages occurred within 10-rkm and those sites were split into two segments at the midpoint between the gages.

Detection and Occupancy Covariates

To account for detection differences due to major changes in either water temperature or flow conditions, I created a ranked variable to reflect conditions associated with sampling months across each annual period. Due to the inconsistent and often incomplete nature of fish assemblage databases (Singer et al. 2020), most commonly used detection variables (e.g., gear type, collector) were not available. Most sites (>85%) were approximately the same length (i.e., 20 rkm); therefore, a site length variable would not be useful as a detection parameter. I used sampling month as a detection variable where similar weather conditions between months would be closer in value than those further apart. I assigned a 1 to June and July, then numbered months sequentially from July to December (1-6) and from June to January (1-6). This ranked variable allowed me to account for the similarity in sampling conditions between sampling months (i.e., rather than numbering sequentially 1-12). Treating sampling month as continuous also allowed me to decrease the number of parameters included in the model when compared to a sequential month factor.

I compiled 1983-2017 daily discharge data from stream gages within the study area to calculate flow metrics at each site (Figure 1). I trimmed the database to ensure the gages included provide the longest period of continuous data common to all gages while also avoiding periods of major dam construction (i.e., 1970s). Discharge data available via USGS gages were used to calculate ecological flow metrics using Eflow Stats (https://github.com/USGS-R/EflowStats). Eflow Stats works well with the USGS data format, has a convenient R interface, and calculates 171 hydrologic indices and seven fundamental streamflow statistics, the "Magnificent Seven" (Archfield et al. 2014).

I reduced the 171-flow metrics to 13 metrics for my analysis by examining correlations and developing ecological hypotheses based on the life history of the fishes. First, I omitted the flow metric "variability in the number of zero-flow days" (DL19) because it relies on having at least 1 zero-flow day in the flow record; thus, the metric cannot be calculated for perennial streams with continuous flow. I also omitted "number of zero-flow months" (DL20) because the value was zero at every site. Next, I calculated Pearson's product moment coefficient for the remaining flow metrics (N = 169) to examine multicollinearity. I removed correlated flow metrics using a cutoff of $|r| = 0.50$ (Supplemental Table S1). Moderate correlation among variables is considered acceptable and may improve predictive power (Allison 1999; Grewal et al. 2004). I retained more general flow variables that tended to be correlated with numerous other metrics to reduce possible spurious results. For example, mean annual flow was selected over mean monthly flow because they were highly correlated, and it would be misleading to suggest importance of any particular month. Lastly, I retained flow metrics based on knowledge of the life history of pelagophil fishes (Moore 1944; Worthington et al. 2016, 2018). The final database had 13 flow metrics and served as the hydrologic foundation for model development (Appendix Table A.1).

In addition to flow metrics, I also quantified occupancy covariates describing precipitation, land-use disturbances, and river fragmentation to account for other major drivers of species distributions. I determined average annual precipitation (1 cm) at each site using an annual precipitation gradient layer in ArcMap (Hijmans et al. 2005). Because land use can also be a driver of species occupancy (Allan 2004; Wenger et al. 2008), I quantified land use into a single metric (i.e., disturbance index). The disturbance index was calculated from annual summaries of historical land use and land cover (Sohl et al. 2016). I created a 10-km buffer centered on each site to quantify land-use disturbance. Number of pixels of each land-use category was quantified in each buffer using the zonal histogram tool in ArcMap (10.2.1, ESRI, Red Lands, California). Land-use categories were weighted using coefficients describing disturbance through modification of the landscape development intensity index (Brown and Vivas 2005). Brown and Vivas (2005) used finer land-use categories than were available via GIS in my study area. Therefore, I followed Mouser et al. (2018) and averaged coefficients, making it applicable to my available land-use data. For example, Brown and Vivas (2005) used both row crops (4.54) and agriculture – high intensity (7.00), whereas I only had a single category describing agriculture (i.e., cultivated land). Therefore, I averaged the two coefficients for the detailed agriculture classes, creating a single coefficient (5.77) to reflect agriculture. I developed a coefficient for each available land-use category. Coefficients range from 1.00 (undisturbed) to 8.32 (highly disturbed) (Table 2). The coefficients were then multiplied by the proportion of pixels of each land-use category within each buffer. The resulting weighted proportions were then summed across categories to obtain a single disturbance index for each year, and the annual values were averaged across each climatic season (cool or warm). Lastly, river-fragment length (rkm) was retrieved from Science Base (https://www.sciencebase.gov/catalog/item/58a60b88e4b057081a24f99d, accessed May 2019) to estimate river length available for propagule drift and migrations. These data were developed by Cooper et al. (2017) using large dam locations (n = 49,468) obtained from the National Anthropogenic Barrier Dataset. These data were spatially linked to the National Hydrography Dataset Plus version 1 (NHDPlusV1) to calculate fragment length (Cooper and Infante 2017; Cooper et al. 2017).

Occupancy Modeling

I examined correlations between my final flow and landscape metrics and made appropriate data transformations. I natural-log transformed most of the covariates (i.e., right skewed) except precipitation gradient, flow predictability (TA2), seasonal predictability of flooding (TA3), and number of day rises (RA5). All covariates were then standardized to a mean value of zero and a variance of one to improve model interpretation. Pearson's product moment coefficient $|r|$ was ≤ 0.49 between all occupancy predictor variables (Appendix Table A.2).

I modeled occupancy of pelagophils while accounting for variation in detection using the hierarchical framework described by MacKenzie et al. (2002). First, I determined if sampling month was an appropriate detection covariate by fitting the detection component of the model (hereafter detection model) using occupied sites (i.e., the species was captured in at least one of the surveys at that site). I treated climatic season (cool or warm) as an indicator variable with cool season as the reference (i.e., dummy variable, MacKenzie 2006). I began by fitting the most complex detection model, where the intercept and sampling month varied by species. I modeled species relationships as deflections around the group mean (Kruschke 2015) to focus on individual species rather than the differences between species (i.e., a traditional reference approach). However, with only two levels of a factor (i.e., species), the coefficients do not "shrink" towards the group mean, but the model estimates are identical to those using a traditional reference approach (Gelman and Hill 2007). Using this approach allowed me to identify significant species relationships, and the model output was interpreted as though I fit separate models for each species but with the benefit of including all data in a single model. This model structure is similar to the "random-slopes" model described by Jamil et al. (2013). The detection model can be written as

$$\text{logit}(p_{ij}) = \Sigma_{k=1}^{2}\alpha_{0k} + \beta_{1k}X_{1[ij]} + \beta_{2wk}CLIM_{w[i]}, \text{for } i = 1,2, \dots N, \text{ for } j = 1,2, \dots J$$

$$\beta_{nk} \sim N(\mu, \sigma^2),$$

where p_{ij} is species detection probability for survey j at site i, α_{0k} is the species k deflection from the group-mean intercept, β_{1k} is the species k deflection from the group mean for the sampling month slope, X_1 is the sampling month, β_{2wk} is the species k deflection from the group mean for the climatic season factor, and CLIM is climatic season. I began by fitting the most complex detection model and removing any species interaction coefficients having 95% highest density intervals (HDIs; Kruschke 2013; Kéry and Royle 2016) that overlapped zero. I then refit the model and removed any main effects (i.e., not included in interactions) using the same criteria (i.e., a hierarchical backward selection, Kleinbaum and Klein 2010). Retained main effects are interpreted as significant detection relationships that did not vary between species. The resulting detection parameters were included in every step of the occupancy model selection process using all sites.

I developed an occupancy model (MacKenzie et al. 2003; MacKenzie 2006) to determine if flow and other landscape metrics related to occupancy by ARS and Plains Minnow while accounting for imperfect detection. The 13 final flow metrics (Appendix Table A.1), annual precipitation gradient, disturbance index, and river-fragment length were considered occupancy covariates. I treated spawning period (spawning or non-spawning) and climatic season (cool or warm) as indicator variables with non-spawning period and cool season as the references (i.e., dummy variable, MacKenzie 2006). I also included two-way spawning period and climatic season interactions with each flow metric to examine different flow needs related to spawning behavior. Similarly to the detection model, I allowed relationships with predictor variables to vary by species (i.e., estimates modeled around the mean) to examine individual species relationships. The occupancy component of the model can be expressed as:

$$\mathrm{logit}(\Psi_i) = \Sigma_{k=1}^{2}\alpha_{0k} + \Sigma_{w=1}^{2}\Sigma_{k=1}^{3}\alpha_{1wk[i]} + \Sigma_{k=1}^{3}\alpha_{2k[i]} + \Sigma_{f=1}^{13}\Sigma_{k=1}^{2}\alpha_{3kf[i]}v_f +$$

$$\Sigma_{n=1}^{16}\Sigma_{k=1}^{2}\Sigma_{c=1}^{16}\beta_{nk[i]}v_n + \Sigma_{n=17}^{32}\Sigma_{k=1}^{2}\Sigma_{c=1}^{16}\Sigma_{w=1}^{2}\beta_{nk[i]}v_c CLIM_{w[i]}, \text{for } i = 1,2, \ldots N$$

$$\alpha_{0k}\text{–}\alpha_{3k}, \beta_{nk} \sim N(\mu, \sigma^2),$$

where Ψ_i is occupancy probability for site i, α_k is the species k deflection from the group-mean intercept, α_{1k} is the species k deflection for the climatic season factor, α_{2k} is the species k deflection for the spawning period indicator variable, α_{3kf} is the species k deflection for the spawning period-flow metric interaction term for flow metric *f*, v_f is a flow metric, β_{nk} is the species *k* deflection from the group-mean for coefficient *n*, v_n is a landscape occupancy covariate, and CLIM is climatic season. I determined the final model using the same backward-selection process described for the detection model by first examining interactions and then main effects that had HDIs that did not overlap zero.

I fitted models using the program JAGS with the jagsUI package (Kellner 2019) in the statistical software R (version 3.5.3; R Development Core Team 2019). The model was fitted using a binomial distribution and logit link function. The logit link function transforms the estimated probability of occupancy to a value between 0 and 1. I used vague uniform priors for model coefficients and vague gamma priors for their associated standard deviations (Kéry and Royle 2016). Posterior distributions for coefficients were estimated using Markov chain Monte Carlo methods with a 40,000-iteration burn-in phase and 100,000 iterations. Convergence was assessed using the Brooks-Gelman-Rubin statistic (Gelman and Rubin 1992) where values < 1.1 for all model parameters indicate adequate mixing of chains (Gelman et al. 2004). A goodness-of-fit test (Kéry and Royle 2016) for the most complex model indicated a lack of overdispersion (i.e., $\hat{c} = 1.00$; MacKenzie and Bailey 2004).

RESULTS

Fish Assemblage Data and Spawning Period

The final fish assemblage database contained 211 samples with 655 surveys. Plains Minnow and ARS were the most commonly sampled fishes with 146 and 65 samples, respectively. Plains Minnow was detected in one or more surveys at 73 sites and ARS was detected at 17. The number of replicated surveys at each site varied from 1–11 with 74 sites having only a single survey for a climatic season. Each indicator variable category (spawning/non-spawning period and cool/warm season) was represented in ≥ 25% of samples and surveys for each species.

Sites

After evaluating the stream gage data, I had 80 sites for my analyses. There were 950 stream gages within the Southern Great Plains, but 138 gages did not record discharge (i.e., gage height or reservoir area were recorded instead). Of the gages that did measure discharge, 389 were active for < 20 years, leaving 423 gages considered useful for developing flow patterns over time (Gan et al. 1991; Olden and Poff 2003). Only 80 of 423 gages recorded data during the study period (1983-2017) and were within the range of my study species (Figure 1; Appendix Table A.3). Each of the river segments containing these gages was a site used in occupancy modeling.

Detection and Occupancy Covariates

The ranked sampling-month covariate was skewed toward warmer months across surveys (mean ± standard deviation = 2.41 ± 1.43). As expected, June and July (coded as 1) were the months with the most surveys (N = 327), followed by May and August (coded as 2; N = 263). March and October (coded as 4) had 126 surveys and February and November (coded as 5) had 32 surveys. December and January (coded as 6) were the least sampled months across surveys (N =19).

I selected 13 flow metrics to consider in occupancy modeling based on my hypothesized ecological importance to pelagophil species (Appendix Table A.1). The flow regime was represented by 4 magnitude metrics, 3 frequency metrics, 2 duration metrics, 2 timing metrics, and 2 rate of change metrics. The magnitude metrics were the most variable, whereas the timing and rate of change metrics were the least variable (Table 3).

The landscape occupancy covariates were moderately variable across surveys (Table 3). The precipitation gradient of the region ranged 37.5–125.0 cm (Table 3). The highest possible values of my disturbance index were not represented in my sites. Rather, the disturbance index ranged 1.00–4.67 (Table 3). River-fragment length ranged 23.8–1154.4 rkm (Table 3).

Occupancy Modeling

I included sampling month in the detection portion of my model and the relationship between month and detection varied by species (Table 4). Detection was not related to climatic season for either species; thus, the climatic season indicator variable was not retained in the final model. The average detection probability (mean levels of coefficients) was similar for ARS (0.73; 0.53–0.88, 95% HDIs) and Plains Minnow (0.69; 0.57–0.79, 95% HDIs). ARS had a negative relationship with sampling month, indicating detection was higher in colder months than in warmer months over the study period. Sampling month was not a significant detection variable for Plains Minnow.

The final occupancy model included river-fragment length, disturbance index, variability in high flow pulse count (FH2), seasonal predictability of flooding (TA3), and a warm season interaction with TA3; all significant relationships were species specific (Table 4). Occupancy was not related to spawning period for either species; thus, the spawning period indicator variable was not retained in the final model. The average occupancy probability (at mean levels of coefficients) during the cool season was twice as high for ARS (0.61) than for Plains Minnow (0.33), although confidence intervals broadly overlapped (95% HDI, 0.16–0.96 and 0.09–0.72, respectively). Plains Minnow occupancy probability was positively associated with river-fragment length (Table 4) and negatively related to variability in high flow pulses (FH2, Table 4; Figure 4). The probability that ARS occupied a site decreased with increasing levels of disturbance. During the cool season, both ARS and Plains Minnow were less likely to occur at a site as seasonal flooding (TA3) became more predictable. The relationship between ARS and TA3 remained unchanged during the warm season; however, Plains Minnow was more likely to occur at a site during the warm season as seasonal flooding became more predictable (Table 4; Figure 5, 6).

DISCUSSION

Persistence of riverine fishes is typically related to components of the flow regime; however, I found that these relationships were context dependent for Plains Minnow but not the federally threatened ARS. The flow regime is considered a 'master variable’ in rivers by controlling the structure and function of aquatic ecosystems (Power et al. 1995; Poff et al. 1997; Boltz et al. 2019). However, climate change is altering riverine ecosystems globally (Kakouei et al. 2018; VerWey et al. 2018; Walker et al. 2020), and some fish responses to flow alterations and flow prescriptions may change based on this overarching background condition (i.e., climate). It was interesting that relationships between flood predictability and variability in high flow pulses and occupancy changed for only one of the pelagophil fishes as related to climate, a reminder that fishes sharing select guilds are not responding in the same manner to some human threats. Moreover, this is the first evidence that suggests Plains Minnow shows an adaptation to select flows during these dry periods that may facilitate persistence over time.

The negative relationship between both ARS and Plains Minnow occupancy and seasonal flood predictability during relatively cool periods may reflect their evolution in flashy flow regimes. Pelagophils evolved physiological and behavioral adaptations in response to the naturally flashy hydrology of the Great Plains (Lytle and Poff 2004; Hoagstrom and Turner 2015; Worthington et al. 2018). High flow events are likely cues for behaviors (e.g., movement upstream, spawning) adapted to flashy flow regimes (Bonner 2000; Durham and Wilde 2014; Hoagstrom and Turner 2015). These adaptations are thought to give pelagophils an advantage in rivers with relatively unpredictable flows. Pelagophils have protracted spawning seasons, likely reflecting a bet-hedging strategy where multiple spawning events increase the opportunities for propagule success in a variable environment (Albers and Wildhaber 2017; Caldwell et al. 2019). Worthington et al. (2016) found that higher flows during the spawning season explained persistence of ARS. The mismatch between the spawning season for these species (April – September) and seasons used to calculate flood predictability may explain why this metric was negatively related to occupancy in my study. Seasonal predictability of flooding is calculated in six, 2-month “seasons” (i.e., December and January, February and March, etc.). Up to three of these “seasons” overlap the protracted spawning seasons of ARS and Plains Minnow. Intermittent flooding throughout the spawning season may enable pelagophils to opportunistically take advantage of high-water events for spawning (Rodger et al. 2016). Additionally, unpredictable higher flow events may deter the establishment and success of introduced non-native species (Rahel and Olden 2008; Pool et al. 2010) that might compete with native pelagophils. As conditions have changed over time, it appears that only one of the two species (Plains Minnow) might have adapted to the increasing predictability of flows.

Occupancy modeling indicates Plains Minnow might adjust to warmer climate conditions better than ARS. During the warm season, Plains Minnow occupancy was positively related to seasonal flood predictability, whereas ARS showed a negative response. Seasonal flood predictability is a measure of the dispersion of flood events across the defined two-month seasons (i.e., high values mean flooding is seasonally concentrated from year to year). Climate change and other human-induced stressors reduce the duration of high flow events and increase the frequency of low flow events in Great Plains streams (Mittal et al. 2016; Chatterjee et al. 2018; Walker et al. 2020). As these systems change, the ability of species to adapt to novel flows will be important to population persistence (Lande 2009; Ruhi et al. 2018; Nadeau and Urban 2019). The ability to adapt to novel flow environments depends on the context of species attributes (e.g., size, behavior, morphology, trophic level). For example, larger fishes are generally more susceptible to extirpation from low-flow events in riverine systems (McCargo and Peterson 2010; Walters 2016; Fabré et al. 2017). Additionally, organisms associated with high-velocity habitats are more susceptible to low flows associated with drought conditions (Buchanan et al. 2017; Patterson et al. 2017). Context dependency of flow-ecology relationships has been evident in other studies (Tonkin et al. 2011; Walters 2016; Walker et al. 2020). Tonkin et al. (2011) found that rises in river height late in the season increased larval and juvenile Australian smelt *Retropinna semoni* growth in years with no floodplain inundation; however, growth was unaffected by river rises in years where the floodplain was inundated. Alternatively, there may be some other factor that was not included in modeling for this study, but is correlated with this flow metric and is the true driver behind this relationship. Although Plains Minnow appears to have adjusted to high seasonal predictability of flooding during the warm season, the species does not appear very tolerant of variability in frequency of high flow events, indicated by a significant negative occupancy relationship with this metric.

Although pelagophils are considered quite tolerant of flow variability, Plains Minnow had a strong negative relationship with variability in the frequency of high-flow events. This relationship suggests an optimum range in frequency of high floods. High flows are necessary for maintenance of habitat complexity and connectivity (Stanley et al. 2010; Bestgen et al. 2017). Periods with exceptionally few high flow events result in habitat homogenization and fragmentation (Poff et al. 2007; Rolls et al. 2012; Bestgen et al. 2020). Alternatively, periods with many high flow events result in channel incision and homogenization through increased substrate mobilization and bank hardening (Best 2019; Bestgen et al. 2020; Gibson and Shelley 2020). This is rapidly facilitated by increasing areas of bank armoring due to invasion by Saltcedar *Tamarix* spp. in riparian areas (Kui et al. 2017; Lightbody et al. 2019; Mayes et al. 2019). High flow events also are thought to facilitate synchronous spawning (Worthington et al. 2018; Perkin et al. 2019; Urbanczyk et al. 2019). Years with few high flow events may result in low recruitment due to fewer spawning opportunities (Dudley and Platania 2007; Durham and Wilde 2008b; Perkin et al. 2019; Urbanczyk et al. 2019). However, a higher than average frequency of high-flow events may wash drifting propagules out of the river channel (e.g., into the reservoir) or into unfavorable habitats (Dudley and Platania 2007; Worthington et al. 2014a), thereby preventing adequate juvenile growth and survival.

The relatively wide HDIs around my flow metric estimates (Figure 4, 5) suggest uncertainty in the flow-ecology relationships. Because my flow and fish data cover a large region (i.e., Southern Great Plains) over several years (1983-2017), there are other biotic and abiotic factors (e.g., co-occurring species, water quality parameters) unaccounted for in my model (Poff et al. 2010; Rosenfeld 2017). My study area covers a variety of stream class types (e.g., intermittent flashy, harsh intermittent, Poff 1996) and lithology (Woods et al. 2005). I could not include these factors as variables in my model because both species did not occur in all levels of the factors (i.e., rank deficiency, Hunter and Caswell 2009). Lastly, there were fewer samples available during extreme flow periods (i.e., particularly times with high frequencies of floods) thereby increasing uncertainty in the resulting flow–ecology relationships (Buchanan et al. 2017).

The relationship between pelagophils and river fragmentation is commonly reported, but there remains a substantial amount of uncertainty in necessary fragment lengths for different species. Increased river-fragment length is considered important for the occupancy of many pelagophil fishes (Wilde and Urbanczyk 2013; Worthington et al. 2014b; Perkin et al. 2015a, 2019). Perkin and Gido (2011) found Plains Minnow required a minimum fragment length of approximately 115 rkm for based on historical fragment extirpations. Others have associated Plains Minnow with relatively long fragment lengths ranging > 100-425 rkm (Dudley and Platania 2007; Perkin et al. 2010). I found no relationship between ARS and fragmentation length, but I recognize the results of these studies depend on a variety of factors. First, all regression-based approaches are dependent on the data used and the variables included in the model. My study is the first to account for incomplete sampling detection while establishing flow-ecology relationships for pelagophils (i.e., other studies assumed 0 was a true absence). Detection of pelagophils can be variable and has been reported as < 0.10-0.58 in the spawning season (Mollenhauer et al. 2018). Differences between studies would be expected given the scale of observations for both sites and surveys; however, both outcomes emphasize a need to account for detection when interest lies in uncovering important ecological relationships. Minimum fragment lengths for ARS have been estimated at > 100 (Dudley and Platania 2007) and 217 rkm (Perkin and Gido 2011) based on historical fragment extirpations, 220 rkm for reproductive cycle completion (Bonner and Wilde 2000), and 360 rkm for drifting propagules (Platania and Altenbach 1998). The range in values provided by other investigators also highlights uncertainty in our understanding of fragment lengths needed. It may be more important for some species of pelagophils and under particular environmental conditions. Some ARS move large distances and some are non-migratory residents (see Chapter 3 results and discussion). The presence of a non-migrant fraction might buffer the effect of river fragmentation on population persistence. River-fragment length may become increasingly important for some species under drought conditions when the ability to recolonize upstream reaches is limited to higher flow events (Radinger et al. 2018; Schumann et al. 2019), unless such events are frequent enough to offset the effect of fragmentation. Alternatively, river fragments may become unimportant under extreme drought periods, where species may be simply unable to reproduce regardless (Perkin et al. 2019). Some climate models predict more intensive flooding (Arnell and Gosling 2016; Jimenez-Martinez et al. 2016; Toosi et al. 2020) which could be beneficial unless the frequency of high flow events is excessive. Like fragmentation related to major dams, disturbance related to land use has been implicated as a factor related to the decline of pelagophil fishes.

Land use disturbances affect a variety of freshwater organisms including pelagophil fishes. Many negative relationships have been established between land-use disturbances and freshwater organism truncated distributions (Pugh et al. 2016; Mouser et al. 2018; Wilkinson et al. 2018) and changes in abundance (Pugh et al. 2016; Joy et al. 2019), but responses are often species specific. For example, most Amazonian stream fishes were negatively related to deforestation hypothesized to relate to habitat loss; however, responses to deforestation varied by species (Brejão et al. 2018). I found ARS occupancy was negatively related to disturbances that included urbanization and agricultural land uses. Human development and agriculture are common land-use disturbances in the Great Plains (Matthews et al. 2005; Woods et al. 2005). Environmental degradation in the forms of habitat destruction and pollution (i.e., nutrients, sediment, salts, and other agricultural and industrial pollutants) is caused or exacerbated by land-use disturbances (Arthington et al. 2016; Best 2019; Britton et al. 2019) and is linked to the decline of pelagophil species (Hoagstrom et al. 2011; Perkin et al. 2015a; Worthington et al. 2018). Urbanization transforms natural habitats by replacing them with impervious surfaces (Calderon et al. 2014; Miller et al. 2014) and increasing soil compaction and runoff (Chen et al. 2017; Sofia et al. 2019). Agricultural land use is a leading cause of water contamination via nutrients (Sharpley et al. 2015; Lupi et al. 2019), organic matter (Wen et al. 2017; Humbert et al. 2019), and sediments (Foucher et al. 2015; Evans et al. 2019). Although the mechanism is unknown, this is an important area of future research. If the negative effects to these fishes are related to contamination or sedimentation, mitigation using constructed wetland and/or riparian buffers may be helpful (Almuktar et al. 2018; Cooper et al. 2019, 2020). Currently, it is difficult to recommend a promising strategy without a better understanding of this linkage.

The use of surrogate species to represent taxa whose ecological relationships are poorly known is a popular strategy (Meurant et al. 2018; Yamaura et al. 2018; Ward et al. 2019), yet the appropriateness of surrogate species may be limited (Jones et al. 2016; Ilg and Oertli 2017; Stewart et al. 2018). Surrogate species are typically selected because they share a perceived ecological trait, guild, or habitat affiliation (Meurant et al. 2018; Yamaura et al. 2018). For example, Shovelnose Sturgeon *Scaphirhynchus platorynchus* is often selected as a surrogate for the federally endangered Pallid Sturgeon *Scaphirhynchus albus* because they share similar life-history traits (e.g., diet, Gosch et al. 2018, 2019; morphology, Jordan et al. 2016; Pracheil et al. 2016). However, species sharing traits or a guild may not respond similarly to an environmental change (Yamaura et al. 2018; Saylor et al. 2020). Although they share a reproductive guild, Plains Minnow and ARS had different relationships with landscape and flow metrics. Other studies have shown that closely related species and species with similar attributes often respond differently under the same circumstances (Banks et al. 2017; Miller et al. 2019b; Saylor et al. 2020). Although sharing forest foraging habitat, Northern Brownbul *Phyllastrephus strepitans* and Green Barbet *Cryptolybia olivacea* respond differently to the presence of leaf litter (Banks et al. 2017). Assuming similar responses to environmental change among fishes sharing a life-history strategy (e.g., pelagophil reproduction) can lead to poor conservation and management decisions (Banks et al. 2017; Ward et al. 2019).

Managers should be alert to context dependency of conservation and management actions. Maintenance of the natural flow regime may be beneficial to fishes under a variety of settings (Poff et al. 1997; Tonkin et al. 2018; Palmer and Ruhi 2019); however, the natural conditions may not be possible to replicate in highly disturbed ecosystems or areas where human water needs are high (Kopf et al. 2015; Brewer et al. 2016; Conallin et al. 2018). Heavily altered river channels that can no longer support natural flow magnitudes may still be able to mimic flow patterns deemed important (e.g., timing and frequency of events, García de Jalón et al. 2019; Palmer and Ruhi 2019; Bestgen et al. 2020). Historical conditions may also not be beneficial for species with flow-ecology relationships that have shifted with climate and other landscape changes (Poff et al. 2016; Poff 2018; Horne et al. 2019). Flow prescriptions are most beneficial when they account for both the present species needs balanced with human water needs (McManamay et al. 2016; Chen and Olden 2017; He et al. 2019). Higher water releases at varying times but average frequencies over the spawning season from year to year can maintain the unpredictable nature of Great Plains ecosystems and reduce opportunities for non-native fishes to establish. Future efforts to test experimental flows and determine a frequency and magnitude of flow events that maintain these populations while minimizing water loss from municipalities and agricultural needs and encouraging the use of water conservation strategies would be advantageous to avoid additional species listings and extirpations.

TABLES

Table 1. Fish assemblage data sources with descriptions. The acronym of the database is listed as the source. The description of each database includes the full name of the database, the type of fish data available (occurrence, abundance, or both), sample years, and the website where these data were obtained.

Source	Description	Date Accessed
BISON	Biodiversity Information Serving Our Nation, Species occurrence database from 1847 – 2016. Accessed from https://bison.usgs.gov/#home	6/13/2018
FOTX	University of Texas, Fishes of Texas, Species occurrence and abundance database from 1900 – 2017. Received from The Nature Conservancy[1].	9/20/2018
GBIF	Global Biodiversity Information Facility species occurrence and abundance database with data from 1853 – 2018. Accessed from https://www.gbif.org/occurrence/search	6/13/2018
MARIS	Multistate Aquatic Resources Information System, Species occurrence and abundance database from 1916 – 2013. Accessed from https://www.sciencebase.gov/catalog/item/51c45ef1e4b03c77	4/22/2018
OWRB	Oklahoma Water Resources Board, Species abundance database from 2003 – 2016. Accessed from https://home-owrb.opendata.arcgis.com/datasets?t=fish	4/22/2018
SNM	Sam Noble Museum, Species occurrence and abundance database from 1921 – 2009. Accessed from http://samnoblemuseum.ou.edu/collections-and-research/ichthyology/ichthyology-database/	4/22/2018

TAMU	Texas A&M University, Species abundance database from 2013. Received from The Nature Conservancy[1].	9/20/2018
TPWD	Texas Parks and Wildlife Department, Species abundance database from 1954 – 2016. Received from The Nature Conservancy[1].	9/20/2018
USFWS	U.S. Fish and Wildlife Service, Species abundance database from 2010 – 2018. Data can be requested from USFWS (Daniel Fenner, contact) or Shannon Brewer (Shannon.brewer@okstate.edu).	NA
VertNet	VertNet (Vertebrate Networks), National Science Foundation, Species occurrence and abundance database from 1916 – 2016. Accessed from http://vertnet.org/resources/datatoolscode.html	4/22/2018

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Table 2. Land-use categories and their associated coefficients used for calculating the disturbance index. Original land-use categories and coefficients were obtained from Brown and Vivas (2005). My land-use categories (Revised land-use categories) were based on available historical land use summaries of Sohl et al. (2016). The final coefficients were calculated by averaging the original coefficients from subgroups within each class (e.g., coefficient values for row crops and high intensity agriculture were averaged to represent agriculture, and the coefficient of different pasture types were averaged to represent a single pasture category).

Original land-use categories	Original coefficient	Revised land-use categories	Final coefficient
Industrial	8.32	Mining	8.32
Row crops	4.54	Agriculture	5.77
Agriculture - high intensity	7.00		
Woodland pasture (with livestock)	2.02	Pasture	2.99
Improved pasture (without livestock)	2.77		
Improved pasture - low-intensity (with livestock)	3.41		
Improved pasture - high-intensity (with livestock)	3.74		
Recreational / open space – low-intensity	1.83	Developed	7.83
Recreational / open space – high-intensity	6.92		
Single family residential – low-density	6.90		

Single family residential – medium density)	7.47		
Single family residential – high density	7.55		
Mobile home (medium density)	7.70		
Highway (2 lane)	7.81		
Low-intensity commercial	8.00		
Institutional	8.07		
Highway (4 lane)	8.28		
Mobile home (high density)	8.29		
Multi-family residential (low rise)	8.66		
High-intensity commercial	9.18		
Multi-family residential (high rise)	9.19		
Central business district (average 2 stories)	9.42		
Central business district (average 4 stories)	10.00		
Natural system	1.00	Barren	1.00
		Deciduous forest	1.00
		Evergreen forest	1.00
		Grassland	1.00

Herbaceous wetland	1.00
Ice/snow	1.00
Mixed forest	1.00
Shrubland	1.00
Water	1.00
Woody wetland	1.00

Table 3. Summary of occupancy covariates including the mean, standard deviation (SD), minimum, and maximum values. Flow covariate definitions are defined in Table A1.

Covariate	Mean ± SD	Minimum	Maximum
*MA7	63.56 ± 210.47	1.28	1300.00
*MA36	51.52 ± 142.47	7.74	1353.08
*ML19	5.95 ± 8.37	0.00	57.60
*MH20	1.01 ± 1.95	0.00	14.90
*FL3	5.16 ± 4.60	0.00	28.00
*FH2	43.88 ± 14.09	14.09	86.60
*FH11	0.49 ± 0.74	0.00	3.50
*DH15	9.41 ± 4.07	2.90	28.00
*DH16	79.71 ± 34.37	15.94	222.42
*TA2	53.09 ± 12.75	12.75	87.75
*TA3	0.26 ± 0.09	0.06	0.68
*RA4	489.61 ± 196.42	139.43	1249.53
*RA5	0.32 ± 0.06	0.06	0.47
Precipitation gradient (cm)	64.4 ± 23.7	37.5	125.0
Disturbance index	2.04 ± 0.92	1.00	4.67
River-fragment length (rkm)	426.1 ± 283.5	23.8	1154.4

* MA7: range in daily flows; MA36: variability across monthly flows; ML19: baseflow conditions using median flow; MH20: specific median annual maximum flow using median flow; FL3: frequency of low pulse spells using median flow; FH2: variability in high pulse count; FH11: flood frequency using median flow; DH15: high flow pulse duration; DH16; variability in high flow pulse duration; TA2: predictability; TA3: seasonal predictability of flooding; RA4: variability in fall rate; RA5 number of day rises

Table 4. Coefficients associated with detection and occupancy flow-ecology variables for Arkansas River Shiner *Notropis girardi* and Plains Minnow *Hybognathus placitus* in the Southern Great Plains. Each parameter indicates the associated species and variable. Average detection parameters are estimated at mean sampling month levels. Average occupancy parameters are estimated at mean levels of continuous variables and referenced to the cool period. Coefficients were reported from the final model and are on a logit scale. Highest density intervals (HDIs; 95%) were reported from the final model. Asterisks indicate parameters that were found to be significant. Detailed descriptions and calculations of flow metrics FH2 (variability in high pulse count) and TA3 (seasonal predictability of flooding) are provided in Kennen et al. (2007).

Parameter	Coefficient	Lower and upper limits for 95% HDI
Detection		
Arkansas River Shiner		
Average detection	0.986	0.106, 2.016*
Sampling month	-0.396	-0.828, -0.046*
Plains Minnow		
Average detection	0.808	0.287, 1.333*
Sampling month	0.135	-0.089, 0.370
Occupancy		
Arkansas River Shiner		
Average occupancy	0.441	-1.629, 3.112

River-fragment length	-0.423	-1.493, 0.369
Disturbance index	-0.992	-2.149, -0.179*
Warm season	0.063	-1.641, 1768
FH2	-0.114	-0.846, 0.639
TA3	-2.661	-8.203, -0.735*
Warm season * TA3 interaction	1.258	-1.369, 6.28
Plains Minnow		
Average occupancy	-0.693	-2.329, 0.931
River-fragment length	1.19	0.623, 1.816*
Disturbance index	0.55	-0.078, 1.234
Warm season	-0.012	-1.066, 1.105
FH2	-1.072	-1.803, -0.483*
TA3	-1.17	-2.171, -0.276*
Warm season * TA3 interaction	1.575	0.437, 3.102*

FIGURES

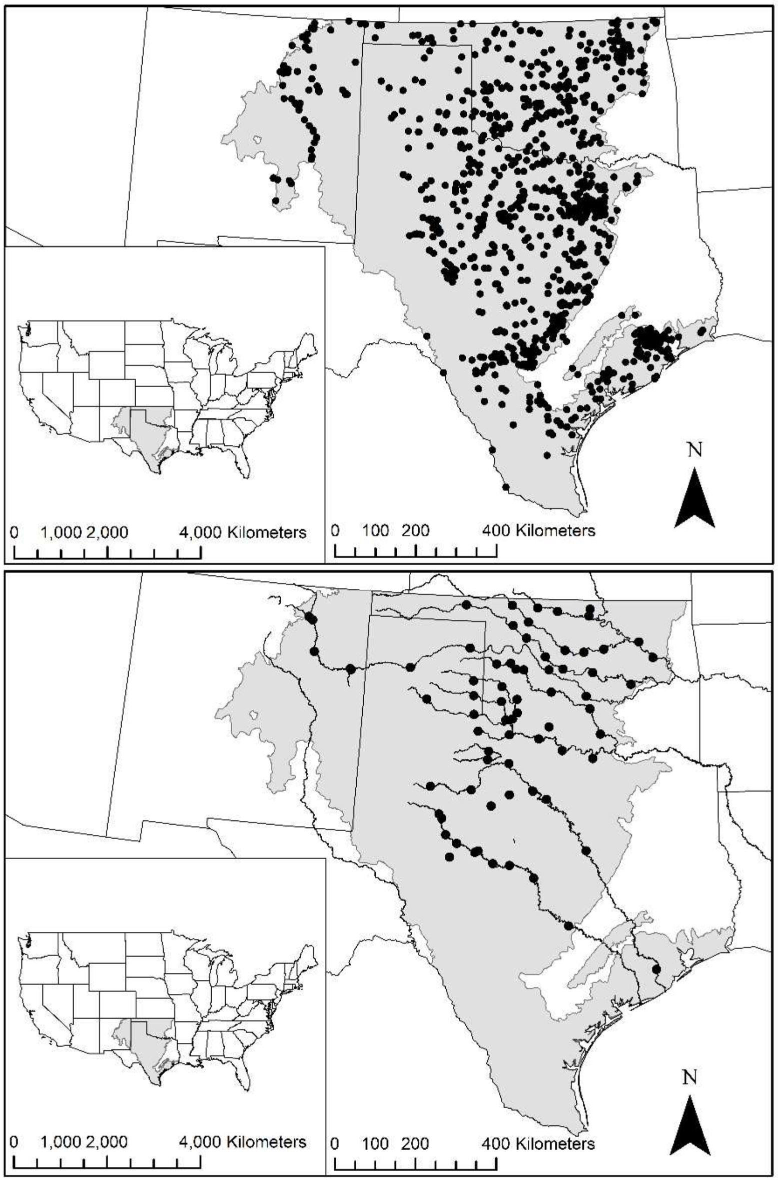

Figure 1. Distribution of U.S. Geological Survey stream gages (black dots) within the southern portion of the Great Plains ecoregion (Southern Great Plains; gray). The top panel is all of the USGS stream gages within the Southern Great Plains. The bottom panel shows major rivers with all of the USGS stream gages within the ecoregion that were retained for analysis based on the ranges of Arkansas River Shiner *Notropis girardi* and Plains Minnow *Hybognathus plactus*, historical fish records, and 20+ years of discharge data available.

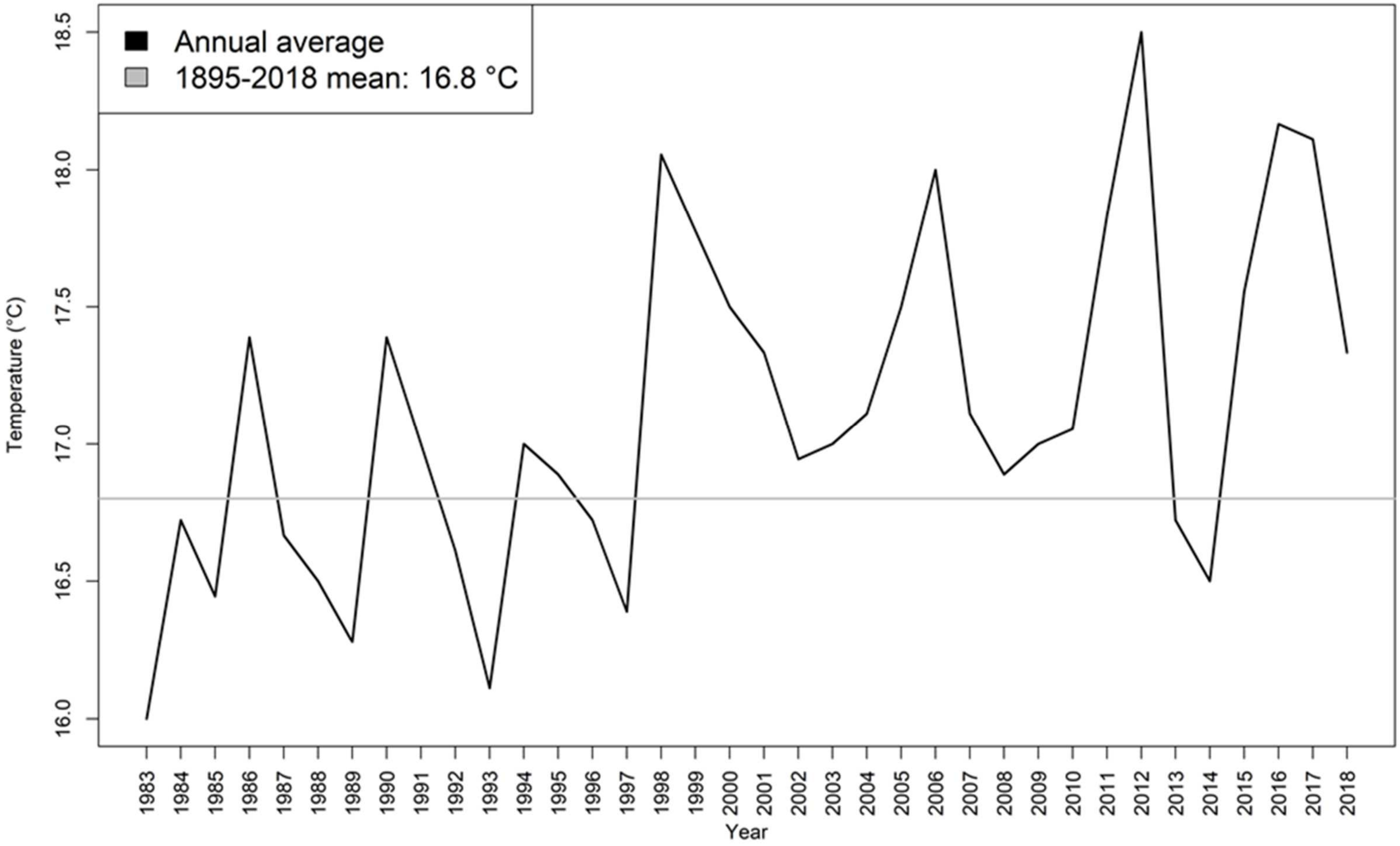

Figure 2. Annual average surface temperatures (black line) in the Southern Great Plains from 1983-2018. The solid gray line indicates the long-term average (16.8°C; 1895-2018). These data were retrieved from National Oceanic and Atmospheric Administration's National Centers for Environmental information from https://www.ncdc.noaa.gov/cag/.

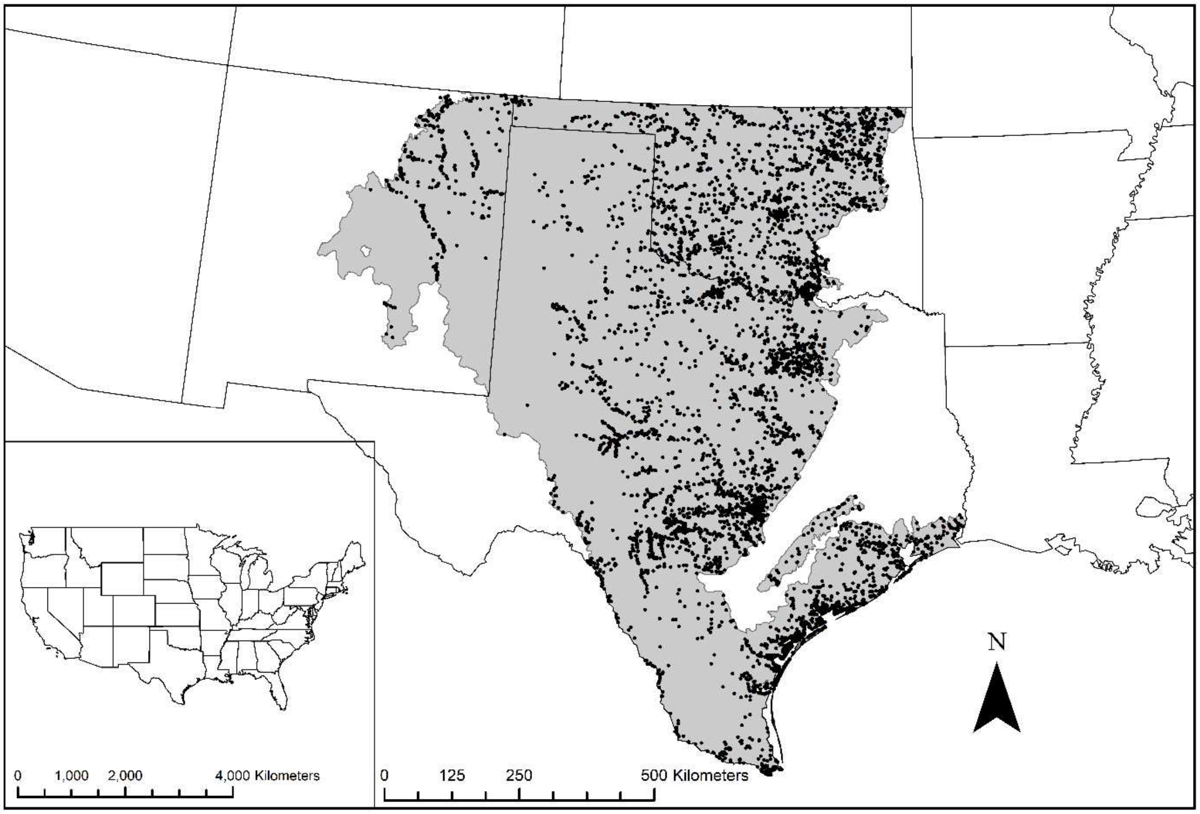

Figure 3. The locations of fish assemblage surveys (black dots) in the Southern Great Plains (light grey) before data were truncated by removing duplicate surveys, incomplete data (i.e., missing date or locality information), and matching these data to the temporal extent of the flow data (see also methods).

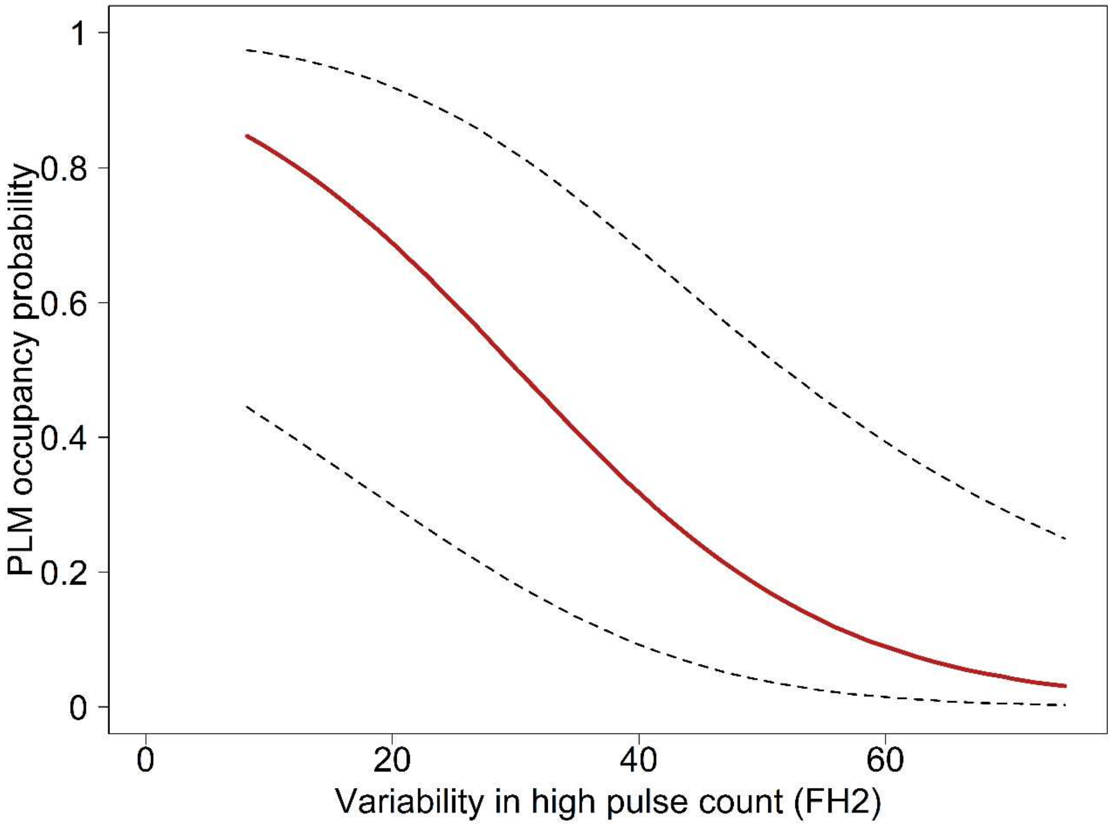

Figure 4. The modeled relationship between variability in high pulse count (FH2, where high pulse is defined by the 75% percentile discharge over the entire flow record; see also Appendix Table A.1) and occupancy probability of Plains Minnow (PLM) after accounting for imperfect detection. At FH2 = 0 there is no variability in the annual number of high pulses over the flow record (i.e., the same number of high flow events occur every year in the flow record). Occupancy probabilities were estimated with other covariates held at mean levels.

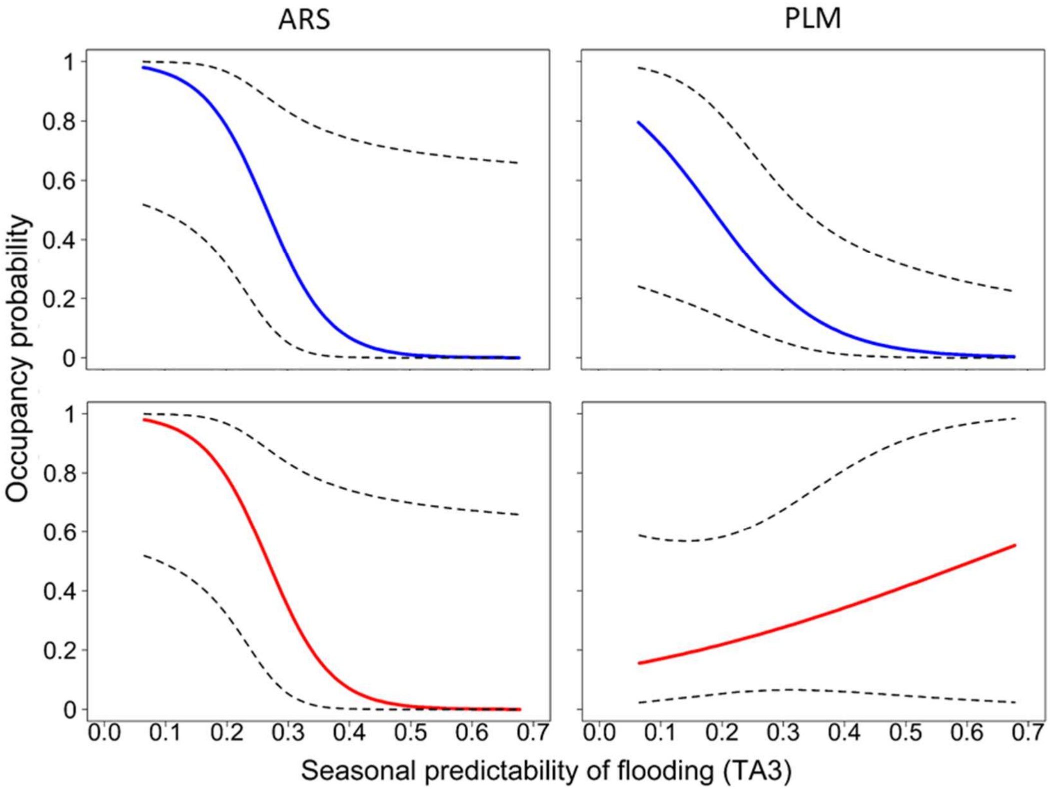

Figure 5. The modeled relationship between seasonal predictability of flooding (TA3, where flooding is defined by the flood that occurs every 1.67 years on average; see also Appendix Table A.1) and occupancy probability of Arkansas River Shiner (ARS; left panels) and Plains Minnow (PLM; right panels) over the cool (blue) and warm (red) seasons after accounting for imperfect detection. At TA3 = 0.0 flooding is completely unpredictable (i.e., random) seasonally. Occupancy probabilities were estimated with other covariates held at mean levels.

CHAPTER III

WINTER MOVEMENT OF TWO PELAGOPHIL FISHES WITH NOTES ON SPRING MIGRATION

ABSTRACT

River fragmentation is hypothesized to contribute to the decline of pelagophils within the Great Plains, but seasonal movement patterns and migration timing are unknown. Therefore, my study objective was to quantify movement of Arkansas River Shiner *Notropis girardi,* and Emerald Shiner *Notropis atherinoides,* during the non-breeding, winter season (November-March). I tagged fish at several sites along the Canadian River in Oklahoma using Visible Implant Elastomer and Passive Integrated Transponder tags the winter of 2018-2019 and p-Chips the winter of 2019-2020. A laboratory study indicated that p-Chips were the most suitable for tagging these species and tag retention and survival were high. I sampled to recapture tagged individuals weekly in the winter. I also sampled 1-2 times monthly in the spring (April-June) to determine when upstream migration is initiated. Using a model ranking and averaging process, I developed a linear mixed model to describe mean daily displacement (distance moved between tagging and recapture divided by the number of days between) of fish recaptured within 14 days post-tagging in winter. Modeling predictor variables included were fish total length, average temperature and discharge of the 10 days prior to recapture events, and photoperiod. I examined net movement distributions and directionality. Recapture rates indicated higher survival and retention using p-Chips in the field. Mean daily displacement relationships were not species specific. Fish moved downstream with increasing temperature, discharge, and photoperiod during winter. Larger individuals had a higher frequency of upstream movement but moved smaller distances than smaller fish. Net movements over the entire study period revealed a downstream movement bias before upstream migration is initiated in the spring by both species. Arkansas River Shiner and Emerald Shiner had leptokurtic and mesokurtic net movement distributions, respectively. The first upstream migration was found in late May. I found evidence that a proportion of Arkansas River Shiner are residents. If the goal is to conserve these species, water management strategies improving river connectivity in late winter through the spawning season may benefit migratory individuals. However, improving lateral river connectivity in reaches with resident individuals (near river km 250) may increase recruitment from that proportion of the population.

INTRODUCTION

Understanding the movement patterns of fishes can inform conservation and management strategies. Use of fish locations over multiple spatial and temporal extents can provide important information on critical fish habitat and river connectivity needs, and allows us to predict the effects of human alteration. For example, man-made barriers fragment critical habitat and interrupt colonization by small-bodied fishes (Perkin and Gido 2012; Perkin et al. 2015b; Zarnetske et al. 2017), and Pennock et al. (2018a) demonstrated that fishways can be constructed to facilitate upstream passage of diminutive fishes through these barriers. Understanding fish movement patterns and use of habitat can prevent poor conservation and management decisions and outcomes (see Cooke et al. 2016 for an overview). For example, biological assessments without considering species movement patterns can underrepresent or omit critical life-history stages (e.g., juveniles or reproducing adults) leading to erroneous conclusions about reproduction or recruitment (e.g., cyprinids and darters, Schlosser 1987, 1991). Fishes move for several reasons including spawning (Falke et al. 2010; Rasmussen and Belk 2017; Morán-López and Uceda Tolosa 2018), juvenile rearing (Brewer and Rabeni 2008; Pavlov and Mikheev 2017; Pavlov et al. 2019), feeding (Brönmark et al. 2008; Nunn et al. 2010; Manning et al. 2019), and accessing refuge environments (e.g., floods and droughts, Schwartz and Herricks 2005; Costa et al. 2019; Ebersole et al. 2020) and other critical habitats (Jones and Stuart 2007; Garwood et al. 2019; Schall et al. 2019).

Fishes make both short and longer-distance movements to successfully complete their life histories. Many populations of fishes include individuals that disperse from their hatching location and resident individuals (e.g., Pecos Bluntnose Shiner *Notropis simus pecosensis,* Chase et al. 2015; Iberian Barbel *Luciobarbus bocagei,* Branco et al. 2017; Rainbow Trout *Oncorhynchus mykiss,* Kelson et al. 2019). The resident portion of a population is variable among species (e.g., 20% Pecos Bluntnose Shiner residents, Chase et al. 2015; 89% Iberian Barbel residents, Branco et al. 2017), by season (Brodersen et al. 2008; Hansen et al. 2020), may depend on climate conditions (Brodersen et al. 2011; McCann et al. 2018), and is an important consideration for management. Fine-scale movements (e.g., short distances within a stream segment) are influenced by habitat patches (microhabitat) offering optimal survival and fish growth (Gowan and Fausch 2002; Bozeman and Grossman 2019; Miller et al. 2019a). Alternatively, coarse-scale movements (e.g., immigration/emigration, long distances over several stream segments or between main channels and tributaries) are important for colonization of unoccupied habitat patches (Matthews and Schaefer 2001; Perkin et al. 2015b; Schumann et al. 2019) and gene flow (Meldgaard et al. 2003; Ruzich et al. 2019; Zhai et al. 2019), and tend to be shaped by long-term evolutionary processes (Lucas and Baras 2008; Cooke et al. 2016; Hegemann et al. 2019). For example, some cyprinids migrate seasonally to increase their long-term fitness (e.g., Roach *Rutilus rutilus,* Brodersen et al. 2008; White Bream *Blicca bjoerkna,* Common Bream *Abramis brama,* Brönmark et al. 2008, 2013). Understanding movement patterns of fishes provides insight into how fishes respond to both natural (e.g., drought) and human perturbations (e.g., introduced species, road crossings) including those factors related to fragmented habitat (Schaefer et al. 2003; Perkin et al. 2015b; Cooke et al. 2016).

River fragmentation is hypothesized as a primary factor associated with the decline of many small-bodied fishes. This seems particularly true for pelagophils (i.e., fishes that reproduce in open water and rely on drift dynamics for development, Perkin and Gido 2011; Wilde and Urbanczyk 2013; Worthington et al. 2018), but seasonal responses to fragmentation are unknown. Although the effects of habitat fragmentation on larger fishes have been relatively well studied (e.g., Morita and Yamamoto 2002; Gosset et al. 2006; Torterotot et al. 2014; Van Leeuwen et al. 2018; Flitcroft et al. 2019; Zambaldi and Pompeu 2020), relationships between diminutive fishes and fragmentation have been slower to emerge (e.g., darters, Blanton et al. 2019; Hubbell et al. 2020; minnows, Pennock et al. 2018a; Hubbell et al. 2020). Fragmentation is thought to prevent spring and summer pelagophil fish dispersal. Bonner (2000) found a higher proportion of larger, sexually mature Arkansas River Shiner *Notropis girardi* (hereafter ARS) in upstream portions of the Canadian River throughout the year suggesting upstream movement over time. At broader spatial and temporal extents, dams and associated reservoirs fragment available habitat where pelagophils were historically abundant, but are now likely extinct (Perkin and Gido 2011; Worthington et al. 2014b; Perkin et al. 2015b). Most movement studies have focused on the breeding season (i.e., April-September) because of the importance of reproduction and prevalence of summer stream drying (e.g., Durham and Wilde 2009; Wells et al. 2017; Pennock et al. 2018). However, the non-breeding season may also be important because these periods are often dry (Mesonet 2018), and winter is a harsh period for fishes (Cunjak et al. 1987; Hurst 2007; Fernandes and McMeans 2019). Moreover, overwinter survival by fish contributing to the next breeding season (i.e., recruitment bottleneck, Schlosser 1998; Hurst 2007; Weber and Brown 2019) is critical to short-lived populations.

Understanding the timing of fish migrations is critical to developing effective management plans. Despite the listing status of many pelagophil fishes, the migration timing of many species has often been assumed or inferred based on indirect or limited evidence (Wilde 2016; Worthington et al. 2018). Previous studies assumed juvenile pelagophils immediately begin upstream migrations because their swimming ability is well-developed (Chase et al. 2015; Wilde 2016). A proportion (82%) of Pecos Bluntnose Shiner were found to move upstream during their first year by Chase et al. (2015), but they were only able to evaluate broad-scale movement (i.e., monthly or seasonal movement over distances >55 km). Ruppel et al. (2020) used monthly occurrences of age groups to indirectly infer upstream movement by Prairie Chub *Macrhybopsis australis* may be driven by refuge habitat instead of reproduction. It is also unknown if pelagophil movements reflect round-trip migration or a one-way dispersal (Archdeacon et al. 2018; Ruppel et al. 2020). Movement of ARS, in particular, during the non-breeding season is largely unknown. Previous work attempted to evaluate ARS movement during the breeding season (April-September), but recaptures were rare (< 2% of n = 1505 over two years, Wilde 2016), leaving a large knowledge gap about the movement patterns of this and related species (Wilde 2010, 2016). Therefore, my second thesis objective was to quantify movement of two pelagophil fishes during the non-breeding, winter season (November-March). I hypothesized that larger pelagophils would be more likely to migrate upstream, and movement would be positively related to environmental cues (i.e., photoperiod, temperature, discharge). I focused on examining winter movement by two pelagophil fishes of the Great Plains: ARS and Emerald Shiner *Notropis atherinoides*. Although both species are pelagophils, they have somewhat different reproductive strategies (see below) and offer an interesting comparison.

METHODS

Target Species

I selected two pelagic minnows as my target species for quantifying movement patterns: ARS and Emerald Shiner. ARS is federally threatened and has declined across most of its historical range (Bonner and Wilde 2000; Worthington et al. 2018). In constrast, Emerald Shiner is a common minnow species and belongs to a similar reproductive guild (i.e., lithopelagophilic broadcast spawning). Lithopelagic fishes release eggs and sperm in open water over substrate, and eggs may be initially adhesive (Simon 1999; Worthington et al. 2018). I included Emerald Shiner to provide movement data for a common reference species for comparison to ARS.

Study area

All sampling was conducted in the Canadian River in the Central Great Plains ecoregion (i.e., level-three ecoregion, Woods et al. 2005) (Figure 1). The Canadian River begins in eastern New Mexico, flows east through the Texas Panhandle, and terminates in Lake Eufaula, Oklahoma (Figure 1). The river is characterized by a relatively wide and shallow sand-bed channel and dynamic abiotic conditions (see Study area of Chapter 2 for a more detailed description). The river flows through rural areas comprising mixed-grass prairie and agriculture upstream of Oklahoma City but is heavily influenced by human water-use activities including withdrawals of groundwater and large impoundments. The Canadian River has had a > 75% reduction in annual discharge from both Ute and Sanford dams (Bonner and Wilde 2000; Woods et al. 2005). The most common land uses in the upper Canadian River basin are grazing, farming, and oil and gas extraction, whereas the lower basin is heavily urbanized (Matthews et al. 2005; U.S. Environmental Protection Agency 2017).

I tagged fish at four primary sites on the Canadian River to quantify movement by ARS and Emerald Shiner; however, I added additional locations as needed to increase recaptures and continue sampling during periodic floods. My tagging site selection was based on: 1) the presence of ARS from recent surveys, 2) collection of adult fish, and 3) landowner permissions. Using U.S. Fish and Wildlife Service (USFWS) fish sampling data, I identified several sites on the Canadian River where ARS have been collected over the previous 5 years. I examined the TL of collected ARS and removed sites where few or no adult ARS were collected (i.e., downstream of Mustang, OK). I contacted several landowners at each of the remaining sites for permission to access the river on private lands. My four primary tagging sites were: the highway 4 bridge crossing south of Mustang, the South Banner Road bridge crossing southeast of Union City, the bridge crossing at south Ranch Road between Union City and Bridgeport, and Fire Canyon north of Bridgeport (Figure 1). Initial re-sampling of tagged fish occurred at the four primary tagging sites, but I added sites at additional locations (Figure 1; Appendix Table B.1). All sites were ≈1 km in length and split into 10 equally sized bins (i.e., 100-m sections of the site) that were used to examine movement within a site when individual identification was not available (i.e., tagged using Visible Implant Elastomer, VIE).

Preliminary tagging

Tagging fish in the laboratory was used to determine the best approach to mark fishes. Preliminary tagging of Emerald Shiner, a common species, as a surrogate in the laboratory suggested passive integrated transponder (PIT) tags would be a feasible option for larger Arkansas River Shiner (Appendix C). Following low recaptures of marked fishes during my first field season, the tagging study was repeated in the lab using Arkansas River Shiner. The second laboratory trial revealed low retention (50%) and survival (53%) by ARS after PIT tagging fish under anesthesia (Appendix C). After testing a new, smaller tag (i.e., p-Chips) with higher retention (72%) and survival (87%) in the laboratory (Appendix C), I changed tag types during my second field season.

Field tagging

I seined weekly during autumn and winter (November-February) to collect fish for tagging. I used a 3.5 X 1.2-m seine with 3-mm mesh to sample fishes. The seine was pre-soaked in VidaLife (Western Chemical Inc., Ferndale, WA) to reduce handling stress on fishes (i.e., reduces friction). Sampling began at the downstream end of each site. Seine hauls ≈10-m long were completed within a variety of habitats, targeting those where pelagophil species were likely to be found (slow slackwater and deep non-moving pools; Matthews and Hill 1980; Worthington et al. 2018). Seining was conducted using standardized sampling techniques (Rabeni et al. 2009) where two people pulled the seine in the downstream direction faster than the current and swept the seine upward to capture fishes. Collected fishes were held in aerated stream water in a cooler until they were anesthetized for tagging.

For the duration of my first field season (November 2018 – April 2019), I tagged fish with fluorescent VIE and PIT tags to examine movement. VIE tags allowed tagging of many individuals of a variety of sizes to examine movement trends over the non-spawning season. After the elastomer was properly mixed, it was injected subcutaneously forming a line in two of six possible locations. The six possible tagging locations were behind the head on the dorsal surface (nape), the dorsal surface in front of the caudal peduncle, parallel to the base of the dorsal fin on either side, and on the lateral surface of the caudal peduncle on either side (Figure 2). Using two tags in each individual allowed me to establish a code for each bin within a site. I used four VIE colors: red, yellow, orange, and blue, to represent the four selected tagging sites. I chose these colors because they are the easiest to distinguish from one another once fish are tagged

(Northwest Marine Technology Inc. 2017a). This tagging scheme allowed me to examine longitudinal movements of 100 m or more when fish move out of their original tagging bin. ARS of sufficient size (≥50-mm TL, based on the Emerald Shiner study above) were also PIT tagged. PIT-tagged individuals were double tagged (i.e., VIE and PIT tag) to allow these fish to be used for all analyses while providing a way to observe tag retention in the field. PIT tags are associated with high retention and low mortality in many species, including select small-bodied cyprinids (Ward et al. 2015; Musselman et al. 2017). PIT tags allowed individual identification so I could relate movement to individual size (TL), but could only be used for larger fishes.

During my second field season (November 2019-March 2020), I tagged fish with p-Chips in an attempt to increase recaptures. Due to a low recapture rate over the first field season (0.7%), I suspected issues with retention or mortality due to tagging (Appendix C). Using P-Chips allowed me to tag many individuals of a variety of sizes and provide individual identification of all recaptured individuals. I could also observe finer movements (≥20 m) using GPS (Garmin eTrex Vista C, Lenexa, Kansas) data for tagging and recapture locations (i.e., rather than relying on batch tagged fishes in bins).

I anesthetized and tagged fish using the same methods outlined in the preliminary tagging efforts (Appendix C) before releasing them to their capture location. Fish were anesthetized using tricaine methanesulfonate (MS-222) at 100 mg/L buffered with sodium bicarbonate to match the pH of the stream water (≈200 mg/L). I kept each fish in the anesthesia solution until it lost equilibrium and operculum movements slowed (i.e., approximately 1-2 min). Then, I recorded its total length (TL; to nearest 1.0 mm) and tagged the fish as described in Appendix C. The VIE was injected subcutaneously into each fish according to manufacturer guidelines (Northwest Marine Technology Inc. 2017a). Following Musselman et al. (2017), I injected a small (8 X 1.4 mm) full-duplex (FDX) PIT tag (Oregon RFID, Portland, OR, USA) into the peritoneum of each ARS ≥ 50-mm TL using a 1.6-mm diameter injection needle. Over the second field season, I injected each fish with a p-Chip subcutaneously left of the base of the dorsal fin using a 0.8-mm diameter injection needle according to manufacturer guidelines (Pharmaseq Inc 2020). All fish were immediately placed into a recovery cooler after tagging. Once all normal behavior resumed, fish were released at their capture location (i.e., the sampled habitat in the 100-m section of stream where they were captured). To reduce stress on fish, water-to-water transfers were made among all holding containers and during release (Matthews et al. 1986; Flagg and Harrell 1990).

Recapture

I seined weekly between November and March to recapture tagged fish to attain information on both relatively fine (≥20 m within sites) and coarse (i.e., among sites) movement patterns. More sites were sampled upstream of the tagging sites due to extensive downstream flooding. Each site was divided arbitrarily into 100-m sections (i.e., 10 bins per site) to detect local movements (100 m or more) of batch tagged fish within the sites. To maintain consistency, I established bin boundaries on the first site visit and returned to these bins throughout the study. I selected and seined transects using the same standardized sampling methods described for tagging. Transects were placed 10-m apart, perpendicular to stream flow within each bin I randomly selected 8 of the 10 transects within each bin to seine on each sampling event. I conducted seine hauls 10-m apart along each selected transect. Further, my initial sampling location on each transect was chosen using a random number between 1 and 10 where each number represented the initial sampling location (meters) from the access. Each seine haul thereafter was 10-m from the first to ensure I was not chasing fish to different locations. My sampling interval (i.e., weekly) was selected to increase recapture rates as previous research attempts only conducted sampling 1-2 times a month (~1% recapture, Wilde 2016). Although I attempted to sample sites weekly, this was not always possible due to weather conditions, river discharge, and landowner permissions.

Additional sampling was conducted 1-2 times per month April-June determine the timing of long-distance fish movements (i.e., among sites). I did not anticipate recapturing enough long-distance movements for quantitative analyses. Rather, I wanted to qualitatively assess when long-distance migrations occur. I sampled with the same standardized sampling methods described in the previous paragraph.

I carefully checked recaptured fish for VIE and PIT tags or p-Chips while avoiding excess handling stress during identification. I transferred recaptured fish to a prepared cooler of stream water with aeration. Captured fish were removed from the cooler individually to avoid PIT tag interference and reduced handling time for each individual. Each fish was visually inspected for VIE tags and scanned with a portable PIT tag reader or visually inspected for a p-Chip. If a p-Chips was located, it was scanned with the handheld laser reader. A shallow dish was used to prevent removing the fish from water unnecessarily during inspection. The identity of individuals and the bin and GPS location where the fishes were recaptured were recorded. After checking for tags, individuals were placed into a second aerated cooler to avoid repeat handling. All fish were released at the approximate capture location (i.e., sample transect) after processing.

Environmental variables

I collected temperature, discharge, and photoperiod data to relate to fish movement. Stream temperature can be an environmental cue for fish movement including migration (Jonsson 1991; Falke et al. 2010). Discharge is commonly related to movement in freshwater fishes but directionality appears to vary by species with some showing positive relationships (Wilde 2016; Wells et al. 2017), whereas other species appear to have negative relationships (Albanese et al. 2004; Beesley et al. 2019). Photoperiod is also commonly linked to warmwater fish movement and often related to reproduction (Fritz et al. 1975; Falke et al. 2010; Krabbenhoft et al. 2014; Valdez et al. 2019).

Discharge was obtained from the U.S. Geological Survey (USGS) stream gage nearest to each recapture site (Figure 1). Temperature was collected at each site using Hobo Pendant MX Water Temperature Data Logger (Onset Computer Corporation, Borne, Massachusetts). Temperature data were recorded hourly and downloaded on each site visit. Sunrise and sunset times were retrieved from the nearest Mesonet station to each site (Figure 1) to calculate photoperiod.

Data analysis

Environmental Metrics

I calculated environmental metrics to relate to movement of pelagophil fishes. Because movement cues may occur several days before I recaptured my tagged fish, I averaged my environmental metrics over a 10-day period before each recapture event. First, I used hourly discharge and temperature values to calculate average daily discharge. Next, I used daily temperature and discharge for the 10 days prior to fish recapture and calculated a 10-day average. I calculated photoperiod as minutes between sunrise and sunset on the day of recapture.

Mean daily displacement

I used movement data to calculate mean daily displacement. Because fish may move in different directions over time, I only included individuals recaptured within 14 days of tagging or previous recapture events to reduce inaccurate calculations of mean daily displacement. I used data from fish tagged with p-chips because they allowed for individual identification. GPS data associated with fish captures were projected in ArcMap (10.2.1, ESRI, Redlands, California) and the longitudinal distance (10 m) between captures was measured using the Locate Features Along Routes tool. I calculated mean daily displacement as the distance between two consecutive captures, scaled by the number of days between captures (Bacheler et al. 2019; Miller et al. 2019a; Chan et al. 2020).

I made appropriate transformations, standardizations, and checked statistical assumptions associated with linear mixed models to prepare for model development. Photoperiod was natural-log transformed due to a right-skewed distribution. All covariates were standardized to a mean of zero and variance of one to improve model coefficient interpretation. The Pearson's product moment coefficient $|r|$ was ≤ 0.42 in all pairwise combinations of predictor variables (Appendix Table B.2). I confirmed that residuals were normally distributed by examining a histogram of residuals and a qq-plot (Kéry and Royle 2016). I plotted the residuals against predicted values to confirm the error variance was homoscedastic (Kéry and Royle 2016).

I developed a candidate set of 27 linear mixed models (Bates et al. 2015) to assess the relationship between movement (i.e., mean daily displacement) and my environmental variables (Table 1). I only used data collected via p-Chip tagged fish for these models because of the low number of recaptured PIT-tagged individuals. It was not reasonable to include a tag type in the model due to the few fish recaptured with PIT tags. My candidate model set included all possible combinations with ≤ 6 parameters of the fixed effects: species, fish TL (mm), average daily discharge (m^3/s), photoperiod (min), and average daily temperature (°C). Models with > 6 variables were not included due to the size of my dataset (i.e., number of recaptures). I evaluated models that included two-way interactions between species and predictor variables to examine species-specific relationships and models where all predictor variables were additive to determine if there were any common relationships among pelagophils (Table 1). A random individual fish effect was included in all models to account for unexplained variation between individuals, the lack of independence between repeat observations on the same fish, and unequal sample sizes among individuals.

I assessed the candidate model set to select the top-ranked model relating mean daily displacement to fish TL and environmental variables. I fitted all models using the program R version 3.6.2 (R Core Team 2019) using the "lme4" package (Bates et al. 2015), and ranked the models using AIC adjusted for small sample size (AICc, Sugiura 1978; Burnham, K. P. and Anderson D. R. 2001) in the "MuMIn" package (Barton 2020). I averaged models with AICc value differences less than 2 using the "MuMIn" package to calculate unconditional coefficients, standard errors, and relative variable importance values (RVI). Model averaging results in more robust estimates of fixed effects when models are ranked closely in model selection (Johnson and Omland 2004; Bolker et al. 2009; Burnham et al. 2011). The RVI values are calculated by summing the AICc values for all averaged models containing that variable (Barton 2020). I also used the "MuMIn" package to calculate R^2 values. I calculated marginal R^2 (R_m^2) to quantify the amount of variance explained by fixed effects and conditional R^2 (R_c^2) to quantify the amount of variance explained by both fixed and random effects in the candidate models (Vonesh et al. 1996; Nakagawa and Schielzeth 2013).

Net Movement and Directionality

I described net movement by all recaptured individuals (November-June) and then statistically determined movement directionality. Net displacement is the distance between the original tagging location and the recapture location. I used the bin midpoints for VIE tagged fish because they were batch tagged; thus, the resolution for these movements is coarser (≥100 m) than for PIT or p-Chip tagged fish (≥20 m). I constructed frequency histograms showing directional movement by species and tag type (i.e., batch tagged data were visualized separately from individual data) where negative values indicated downstream movement and positive values indicated upstream movement (Petty and Grossman 2004; Wells et al. 2017). Chi-square tests were used to test for movement directionality by ARS and Emerald Shiner (Skalski and Gilliam 2000; Branco et al. 2017; Wells et al. 2017) using the program R version 3.6.2 base package (R Core Team 2019). The normality and kurtosis of each species distribution were tested using D'Agostino's test for normality using the "fbasics" package (Wuertz et al. 2020) and Anscombe-Glynn's test of kurtosis using the "moments" package (Komsta and Novomestky 2015) respectively (Anscombe and Glynn 1983; Skalski and Gilliam 2000). Leptokurtic dispersal patterns occur when most of the population moves relatively little, but there is a small portion that moves long distances (i.e., a distribution with a high central peak and long tails on either side, Fraser et al. 2001). Leptokurtic dispersal patterns are often found in field movement studies (Fraser et al. 2001), making it a popular pattern within movement prediction models such as the *Fishmove* package created for R statistical environment (Radinger and Wolter 2014). Determining if pelagophil fishes follow a leptokurtic dispersal pattern is useful in making future movement predictions.

RESULTS

Field tagging

In my first field season, I tagged 3,122 fish across sites (Table 2). ARS were the dominant species tagged (70%; Table 2). I did not PIT tag Emerald Shiner because it was sensitive to PIT tagging stress during initial field tagging, despite the high survival in lab efforts (95%; Appendix Table C.3). Excess field tagging mortality was not apparent in ARS. Most ARS captured were 45-50 mm TL, with 28% of ARS large enough to be PIT tagged (Figure 3). Higher proportions of large ARS (≥50 mm TL) were tagged at upstream sites. Mean TL generally increased for both species farther upstream (Table 2). Differences in abundances of tagged fish among sites were mostly related to sampling effort at each site (i.e., more fish were tagged at the Caddo site, but I also had more opportunity to sample there due to discharge conditions and landowner approval; Table 2; Appendix Table B.1). Neither species was captured or tagged at the two most upstream sites, Roll and Camargo (Figure 1; Appendix Table B.1).

For my second field season, I tagged 2,898 fish across 4 sites (Table 3). Fish were captured in higher numbers at the 4 initial tagging sites relative to field season 1; thus, tagging was not conducted at additional sites during season 2. Similar to the first season, ARS were the dominant species tagged (70%; Table 3). Unlike the first field season, mean TL did not have an obvious trend among sites for either species (Table 3). The number of fish tagged was again related to sampling effort for each site (Table 3; Appendix Table B.1).

Recapture

Across field seasons, I had 137 recaptures comprised mostly of ARS (n = 96). ARS tagged in field season 1 were recaptured at a lower rate (1.09%; n = 24) than season 2 (3.56%; n = 72). Emerald Shiner also had a lower recapture rate in season 1 (0.44%; n = 4) than season 2 (4.22%; n = 37). Only 5 PIT-tagged ARS were recaptured in the first season. The majority of recaptured fish were found within their tagging site (≤885 m; Table 4, 5). However, 3 ARS and 1 Emerald Shiner recaptured late in the season moved extensive distances (30.23 – 100.77 km; Table 6). There were also 3 ARS that were only VIE tagged in the first field season and recaptured within their tagging site in the following field season (30 October 2019; 8 January, 2020; and 4 February 2020). Recaptured ARS non-directional movement distance (i.e., absolute value of movement distances) increased over the duration of the sample seasons more so than Emerald Shiner (Figure 4). However, larger individuals of both species moved shorter distances than smaller individuals (Figure 5). During the second field season, neither ARS nor Emerald Shiner were captured at the Thomas site until February 17 but numbers increased thereafter. Similarly, they were not captured at the Taloga site until March 3 when 1 individual was captured and numbers slowly increased at that site thereafter.

Environmental variables

The environmental covariates for mean daily displacement modeling varied across recapture events from November-March. The 10-day average discharge ranged from 4.90-12.19 m^3/s (mean = 7.59 m^3/s; SD = 1.72), which covered all the wadeable values over the duration of the second field season. The 10-day average temperature ranged 4.75-10.66 °C (mean = 6.83 °C; SD = 1.69). Photoperiod was right skewed, as expected, ranging 986-1177 minutes (mean = 999.22 min; SD = 25.50).

Mean daily displacement

Mean daily displacement ranged -210-205 m (mean = -15.47 m; SD = 59.27; n = 30) for ARS and -199-200 m (mean = -22.14 m; SD = 70.31; n = 30). Of the fish included in modeling, 2 ARS and 2 Emerald Shiner were recaptured twice. A single Emerald Shiner was recaptured 3 consecutive times. All of these individuals were recaptured at the same site (i.e., Fire Canyon, Figure 1; Appendix Table B.1) 1-4 days after their previous recaptures. These individuals were variable in size, ranging 40-54 mm total length.

Mean daily displacement was related to all of our predictor variables except species (i.e., temperature, discharge, total length, and photoperiod). Species was not in the top-ranked model or any of the averaged models, indicating ARS and Emerald Shiner have common relationships with all predictor variables. ARS and Emerald Shiner had a weak ($R^2 = 0.0051$) negative relationship with temperature in the top model (Table 1, 7). The random individual fish effect explained most of the variability in the model ($R^2 = 0.79$). These species also moved downstream more with higher discharge and longer photoperiod (Table 7). Larger fish also moved upstream more than smaller fish. Photoperiod (0.193) and total length (0.164) had the higher RVI values (Table 7).

Net movement and directionality

Frequency histograms showed that all groups had more downstream movements than upstream (Figure 6, 7). ARS had more large net movement distances compared to Emerald Shiner (i.e., longer tails on the frequency histograms). A greater number of ARS ($\chi^2 = 9.29$, $P = 0.002$) and Emerald Shiner ($\chi^2 = 4.76$, $P = 0.03$) moved downstream than upstream. D'Agostino Normality Tests indicated net distances were not normally distributed for ARS ($D = 148.18$, $P < 0.01$) and Emerald Shiner ($D = 7.71$, $P = 0.02$). The net movement distribution was leptokurtic in ARS (kurtosis = 65.43, $P < 0.01$), but mesokurtic (i.e., normal kurtosis) in Emerald Shiner (kurtosis = 3.66, $P = 0.20$).

DISCUSSION

There has been much debate over the drift compensation theory for pelagophil movements (Hoagstrom 2014; Wilde and Urbanczyk 2014; Ruppel et al. 2020). This theory predicts upstream movement to compensate for larval drift downstream. However, I found evidence of the reverse pattern, with greater numbers of downstream movement in both ARS and Emerald Shiner. Previous studies on ARS movement only recaptured individuals over the breeding season (i.e., spring and summer; Wilde 2010, 2016), leaving movement patterns over the non-breeding season unknown. Although this study confirmed long-distance upstream movements (30-100 km) by some individuals, these occurred in May and June, after the start of the breeding season. Moreover, both resident and migratory individuals were captured in breeding condition. For example, I recaptured a tagged ARS in breeding condition at the same site where it was tagged five months earlier (i.e., January to May 2020). This suggests that either the fish left and returned to the same site or the fish was a resident at that location. My study also provides the first evidence of Emerald Shiner making long-distance migrations in a river system.

Although pelagophils are associated with upstream movement, it appears that ARS and Emerald Shiner have a downstream movement bias during winter. Winter is harsh for fishes due to extreme abiotic conditions and low food availability (Cunjak et al. 1987; Hurst 2007; Fernandes and McMeans 2019). Fish energy expenditure is typically lower in the winter (Brownscombe et al. 2017; Cote et al. 2020). Expending the energy to move, particularly upstream, during this time may result in decreased fitness through overwinter mortality (Bonte et al. 2012; Fernandes and McMeans 2019) or decreased growth and reproductive potential (Kinnison et al. 2003; Bonte et al. 2012). Other highly mobile fishes also decrease movement in the winter to conserve energy (e.g., bonefish *Albula vulpes,* Brownscombe et al. 2017; Lake Sturgeon *Acipenser fulvescens,* Thayer et al. 2017). Fishes often resume moving long distances at the onset of spring (Verhelst et al. 2018; Tornabene et al. 2020). However, in contrast to speculation on pelagophils (e.g., Pecos Bluntnose Shiner, Chase et al. 2015), my results do not support upstream movement immediately following larval development.

My results are consistent with the expectation that pelagophils make long-distance upstream movements at the onset of spring. The relevant recaptures for ARS and Emerald Shiner (Table 6) occurred in May and June, after the expected onset of spawning (April) for these species. These recaptures were few and do not provide information on movement timing directly, but it is worth noting that they occurred after spring began despite lower sampling effort at that time. Further indicating upstream movement preparatory to spawning is the observation that ARS and Emerald Shiner were not captured at upstream sites until later in the field season and such captures occurred in increasing numbers over time. The long-distance movements detected for ARS are similar to other reports for the species (i.e., 51.7 km, Wilde 2010; 13.3-213.6 km, Wilde 2016). Although it is hypothesized that a proportion of all pelagophils migrate upstream for spawning (Wilde 2016; Worthington et al. 2018), Emerald Shiner was previously only thought to migrate between river and reservoirs based on sampling abundances and genetic similarities between river and reservoir populations (Campbell and Maccrimmon 1970; Lang 2016). My results are the first records of Emerald Shiner making extensive lotic migrations, suggesting lithopelagophilic fishes may have movement strategies similar to those of pelagophis.

Although I found long-distance movements, I also found evidence that a proportion of the ARS population comprises non-migratory residents. Several ARS individuals were recaptured at their tagging site, after long periods (e.g., over a year). There are life-history tradeoffs between migration and residency (Dermond et al. 2019; Tigreros and Davidowitz 2019). Migration may increase recruitment because propagules require some longitudinal distance to develop while drifting (Wilde and Urbanczyk 2013; Worthington et al. 2014a; Perkin et al. 2019); however, residents do not expend energy on migration, possibly resulting in better body condition, higher fecundity, and the capacity to spawn on more occasions over a single spawning season (Brosset et al. 2016; Tigreros and Davidowitz 2019). The overall effect might be a bet-hedging strategy that heightens overall recruitment (Albers and Wildhaber 2017; Caldwell et al. 2019). For example, migratory individuals may require greater river fragment lengths to persist over long time frames, but resident individuals may persist in relatively shorter fragments when connectivity is decreased (see discussion of Chapter 2 for an overview). The overall occupied space is the same during the spawning and non-spawning seasons for ARS (see results of Chapter 2), indicating that not all individuals are migratory. Although the proportion of residents could not be determined from this study, this is an important area for future research. In particular, it would be advantageous to determine survival associated with the two strategies under different environmental conditions (e.g., high versus low flows). The outcome could lead to more fruitful management strategies for a species occupying a semi-arid region.

Movements by ARS and Emerald Shiner are potentially underestimated in my study because data were limited by the ability to recapture individuals, especially at downstream locations. Fish may have moved upstream and downstream multiple times between capture events, causing mean daily movement estimates to be lower than the actual values. Because sites are several km apart, individuals that moved shorter distances are easier to capture than individuals that move out of their tagging sites (Rodríguez 2002; Albanese et al. 2003; Kanno et al. 2020). These estimates only use the values of recaptured individuals, causing long-distance movements to be underrepresented in my data. It is common for movement to be underestimated in mark-recapture studies using repeat sampling at relatively few sites (Albanese et al. 2003). This underestimation was most likely exacerbated in the first field season because the tags used were heavier and associated with higher mortality. Moreover, I was unable to sample at my most downstream sites on many occasions due to flooding. Consequently, it is not clear that these species do not make extensive downstream migrations.

P-Chips may be a suitable tagging method for small-bodied fishes, especially those species intolerant of other tag types (e.g., VIE, PIT). ARS and Emerald Shiner are associated with higher mortality when tagged with VIE and PIT tags (see Appendix C for an overview). VIE performed better than PIT tags, possibly because they are lighter and less invasive (i.e., injected subcutaneously instead of into the peritoneum). Some of this mortality was associated with handling stress, shown also in the control groups of my laboratory tagging efforts. To my knowledge, this is the first quantitative assessment of tagging retention and mortality in ARS. Moreover, my findings agree with others that Emerald Shiner are sensitive to handling stress

(Pennock et al. 2018a). Although the initial lab study for Emerald Shiner revealed high survival associated with PIT tagging, high mortality was evident during initial field tagging. Handling stress may have been exacerbated by field conditions (e.g., low temperatures, wind) that were not representative of conditions in the controlled lab setting. The lab setting may also provide optimal conditions for recovery, whereas conditions are much harsher in the field (e.g., strong currents, predators, parasites). This difference highlights the importance of using both lab and field studies to assess physiological responses (Speers-Roesch and Norin 2016; Colotelo et al. 2017). When tagging with p-Chips, fish can be held in a shallow dish to reduce handling stress. Additionally, p-Chips are very small (0.5 mm X 0.5 mm X 0.1 mm) and lightweight, providing quicker healing times and reduced tag loss. P-Chips increased recaptures over the second field season to allow mean daily displacement modeling.

The variables I included in my mean daily displacement models were weakly associated with winter movement. Discharge, temperature, photoperiod, and fish size were all marginally related to daily movement, but random variation among individuals accounted for most of the variation in my data. High individual variation in movement is common among fish species (Skalski and Gilliam 2000; Petty and Grossman 2004; Miller et al. 2019a), and may be related to a variety of factors (e.g., individual attributes, Skalski and Gilliam 2000; Miller et al. 2019; local environmental conditions, Gilliam and Fraser 2001; Petty and Grossman 2004). Although larger fish were associated with higher frequency of upstream movement, they moved shorter distances than smaller fish during the winter months. Larger individuals are typically associated with larger movements (Monnot et al. 2008; Rasmussen and Belk 2017; Miller et al. 2019a). There may be some other important factor related to movement that was not included in my models such as food availability (Brochier et al. 2018; Crook et al. 2020). I also could not include interaction terms between environmental variables, because of the limited number of recaptures. Many studies have found that a combination of variables drives movement (Falke et al. 2010; Shaw

2016; Miller et al. 2019a). Modeled movements were relatively short distances (i.e., -455 to 205 m) during the winter. I had no way to determine which individuals would later make long-distance movements (i.e., in the spring). Residents and migrants may differ physiologically causing them to have different short-distance movement cues (Dermond et al. 2019). For example, differential movement of juvenile Brown Trout, *Salmo trutta,* between river and reservoir was governed by fish size and growth, hypothesized to be related to predation risk (Dermond et al. 2019). The modeled relationships I found were not species specific, indicating that winter movement patterns are similar between ARS and Emerald Shiner.

Different management strategies could be attempted based on my movement results. My findings suggest that management strategies focused on improving river connectivity in late winter through the spawning season would benefit individuals making upstream movements and propagule drift. However, different strategies may be possible that benefit spawning and recruitment by resident members of the population. For example, access to the floodplain has been hypothesized to produce spawning habitat (Medley and Shirey 2013; Hutson et al. 2018; Valdez et al. 2019); therefore, restoration allowing floodplain access may be beneficial to resident populations. Although I have limited data, it seems floodplain restoration locations near rkm 348 would be most beneficial. I did not capture fish upstream of rkm 379 until late February/March and previous sampling suggests downriver locations near rkm 250 rarely produce adult captures. Determining flows that maintain longitudinal and lateral river connectivity could benefit recruitment for both migratory and resident populations (Perkin et al. 2019; Valdez et al. 2019), but may vary in differing climate conditions (Garbin et al. 2019). Developing experimental flows based on my findings in this and Chapter 2 combined may present a useful area of applied future research.

TABLES

Table 1. Ranks of candidate linear mixed models evaluating mean daily displacement related to fish total length and environmental variables. My complete data set included 60 recaptured individuals: 30 Arkansas River Shiner *Notropis girardi* and 30 Emerald Shiner *Notropis atherinoides*. For each model, Y is displacement at observation i for fish j, β_0 is the grand intercept, γ is the random fish effect, Species is the species of fish j where Arkansas River Shiner is the reference (two levels), TL is fish total length (mm), discharge is the 10 day average discharge (m^3/s), temperature is the 10 day average water temperature (°C), and photoperiod is the number of minutes of daylight on the day of recapture. K is the number of model parameters and comprises both fixed and random effects and interaction parameters. Log likelihood and Akaike's information criterion adjusted for small sample size (AICc) is reported. ΔAICc is calculated as the difference in AICc score between each model and the top model. The Akaike weight (w_i) represents the relative support for each model. Marginal R^2 (R_m^2) represents the amount of variance explained by the fixed effects, whereas the conditional R^2 (R_c^2) represents the variance explained by both fixed and random effects.

Rank	**Model**	***K***	**AICc**	**ΔAICc**	**Log likelihood**	**w_i**	**R_m^2**	**R_c^2**
1	**$Y_{ijk} = \beta_0 + \gamma_j + \beta 3 Temperature_{ik}$**	**2**	**673.15**	**0.00**	**-333.4**	**0.147**	**0.0051**	**0.7878**
2	$Y_{ijk} = \beta_0 + \gamma_j + \beta_2 Discharge_{ik}$	2	673.44	0.29	-333.5	0.127	0.0042	0.7893
3	$Y_{ijk} = \beta_0 + \gamma_j + \beta_4 Photoperiod_{ik}$	2	673.55	0.40	-333.6	0.120	0.0000	0.7849
4	$Y_{ijk} = \beta_0 + \gamma_j + \beta_1 TL_j$	2	674.12	0.97	-333.8	0.091	0.0008	0.7874

5	$Y_{ijk} = \beta_0 + \gamma_j + \beta_1\text{TL}_j + \beta_4\text{Photoperiod}_{ik}$	3	674.55	1.40	-332.9	0.073	0.0242	0.7860
6	$Y_{ijk} = \beta_0 + \gamma_j + \beta_1\text{TL}_j + \beta3\text{Temperature}_{ik}$	3	675.24	2.09	-333.3	0.052	0.0159	0.7915
7	$Y_{ijk} = \beta_0 + \gamma_j + \beta_2\text{Discharge}_{ik} + \beta3\text{Temperature}_{ik}$	3	675.39	2.24	-333.3	0.048	0.0026	0.7873
8	$Y_{ijk} = \beta_0 + \gamma_j + \beta3\text{Temperature}_{ik} + \beta_4\text{Photoperiod}_{ik}$	3	675.43	2.28	-333.4	0.047	0.0075	0.7890
9	$Y_{ijk} = \beta_0 + \gamma_j + \beta_1\text{TL}_j + \beta_2\text{Discharge}_{ik}$	3	675.57	2.42	-333.4	0.044	0.0127	0.7926
10	$Y_{ijk} = \beta_0 + \gamma_j + \alpha_1\text{Species}_j$	2	675.66	2.51	-333.5	0.042	0.0028	0.7849
11	$Y_{ijk} = \beta_0 + \gamma_j$	1	675.67	2.52	-335.7	0.042	0.0000	0.8046
12	$Y_{ijk} = \beta_0 + \gamma_j + \beta_2\text{Discharge}_{ik} + \beta_4\text{Photoperiod}_{ik}$	3	675.72	2.57	-333.5	0.041	0.0024	0.7880
13	$Y_{ijk} = \beta_0 + \gamma_j + \beta_1\text{TL}_j + \beta_2\text{Discharge}_{ik} + \beta_4\text{Photoperiod}_{ik}$	4	676.78	3.63	-332.8	0.024	0.0268	0.7893
14	$Y_{ijk} = \beta_0 + \gamma_j + \beta_1\text{TL}_j + \beta3\text{Temperature}_{ik} + \beta_4\text{Photoperiod}_{ik}$	4	676.88	3.73	-332.9	0.023	0.0249	0.7875
15	$Y_{ijk} = \beta_0 + \gamma_j + \beta_1\text{TL}_j + \beta_2\text{Discharge}_{ik} + \beta3\text{Temperature}_{ik}$	4	676.93	3.78	-332.9	0.022	0.0250	0.7945
16	$Y_{ijk} = \beta_0 + \gamma_j + \beta_2\text{Discharge}_{ik} + \beta3\text{Temperature}_{ik} + \beta_4\text{Photoperiod}_{ik}$	4	677.50	4.35	-333.2	0.017	0.0124	0.7929
17	$Y_{ijk} = \beta_0 + \gamma_j + \alpha_1\text{Species}_j * \beta_1\text{TL}_j$	4	677.97	4.82	-332.2	0.013	0.0459	0.7844
18	$Y_{ijk} = \beta_0 + \gamma_j + \beta_1\text{TL}_j + \beta_2\text{Discharge}_{ik} + \beta3\text{Temperature}_{ik} + \beta_4\text{Photoperiod}_{ik}$	5	679.13	5.98	-332.8	0.007	0.0285	0.7918
19	$Y_{ijk} = \beta_0 + \gamma_j + \alpha_1\text{Species}_j * \beta_4\text{Photoperiod}_{ik}$	4	679.55	6.40	-333.0	0.006	0.0190	0.7934
20	$Y_{ijk} = \beta_0 + \gamma_j + \alpha_1\text{Species}_j * \beta3\text{Temperature}_{ik}$	4	679.83	6.68	-333.1	0.005	0.0131	0.7909

21	$Y_{ijk} = \beta_0 + \gamma_j + \alpha_1\text{Species}_j * \beta_2\text{Discharge}_{ik}$	4	680.27	7.12	-333.3	0.004	0.0068	0.7801
22	$Y_{ijk} = \beta_0 + \gamma_j + \alpha_1\text{Species}_j * (\beta_1\text{TL}_j + \beta_4\text{Photoperiod}_{ik})$	6	682.88	9.73	-332.0	0.001	0.0513	0.7891
23	$Y_{ijk} = \beta_0 + \gamma_j + \alpha_1\text{Species}_j * (\beta_1\text{TL}_j + \beta 3\text{Temperature}_{ik})$	6	682.91	9.76	-332.0	0.001	0.0498	0.7897
24	$Y_{ijk} = \beta_0 + \gamma_j + \alpha_1\text{Species}_j * (\beta_1\text{TL}_j + \beta_2\text{Discharge}_{ik})$	6	683.17	10.02	-332.2	0.001	0.0466	0.7881
25	$Y_{ijk} = \beta_0 + \gamma_j + \alpha_1\text{Species}_j * (\beta 3\text{Temperature}_{ik} + \beta_4\text{Photoperiod}_{ik})$	6	684.22	11.07	-332.7	0.001	0.0280	0.8019
26	$Y_{ijk} = \beta_0 + \gamma_j + \alpha_1\text{Species}_j * (\beta_2\text{Discharge}_{ik} + \beta 3\text{Temperature}_{ik})$	6	684.49	11.34	-332.8	0.001	0.0200	0.8279
27	$Y_{ijk} = \beta_0 + \gamma_j + \alpha_1\text{Species}_j * (\beta_2\text{Discharge}_{ik} + \beta_4\text{Photoperiod}_{ik})$	6	684.55	11.40	-332.9	0.000	0.0236	0.8006

Table 2. Summary of tagged Arkansas River Shiner *Notropis girardi* and Emerald Shiner *Notropis atherinoides* at each site over the duration of the first field season (November 2018-June 2019): number of tagged fish (N), average size (Mean and standard deviation), and minimum and maximum total length. All fish were tagged with Visible Implant Elastomer. ARS ≥ 50 mm total length were also tagged with 8-mm PIT tags in the peritoneum (in parentheses). The original tagging sites (bold) and sites that were added to increase recapture rates (not in bold) are listed from upstream to downstream. See Appendix Table C.1 for sampling event dates at each site.

	N	**Mean ± SD**	**Minimum**	**Maximum**
Arkansas River Shiner				
Taloga	57 (29)	49.34 ± 2.78	44	57
Thomas	137 (68)	48.58 ± 3.41	31	57
Fire Canyon	514 (166)	47.35 ± 3.87	32	60
Caddo	588 (152)	46.30 ± 4.66	33	63
Braum’s	285 (78)	45.88 ± 4.00	35	56
Mustang	436 (78)	44.51 ± 4.99	30	59
Norman	194 (49)	45.95 ± 4.71	32	58
Total	2211(620	46.36 ± 4.49	30	63
Emerald Shiner				
Taloga	27	61.64 ± 5.80	55	75
Thomas	63	62.83 ± 4.60	45	76
Fire Canyon	99	59.38 ± 5.73	42	75
Caddo	272	58.59 ± 6.02	40	76
Braum’s	281	59.12 ± 7.18	40	88
Mustang	144	54.20 ± 7.61	37	78
Norman	25	55.72 ± 7.09	42	68

Total	911	58.47 ± 6.92	37	88

Table 3. Summary of tagged Arkansas River Shiner *Notropis girardi* and Emerald Shiner *Notropis atherinoides* at each site over the duration of the second field season (November 2019-June 2020): number of tagged fish (N), average size (Mean and standard deviation), and minimum and maximum total length. All fish were tagged with p-Chips. Tagging sites are listed from upstream to downstream. See Appendix Table B.1 for sampling event dates at each site.

	N	**Mean ± SD**	**Minimum**	**Maximum**
Arkansas River Shiner				
Fire Canyon	528	47.12 ± 3.70	37	58
Caddo	691	45.04 ± 4.34	35	58
Braum’s	642	45.09 ± 4.80	36	59
Mustang	161	45.90 ± 4.41	37	59
Total	2022	45.67 ± 4.43	35	59
Emerald Shiner				
Fire Canyon	343	53.83 ± 5.06	43	83
Caddo	303	52.35 ± 6.00	39	69
Braum’s	171	55.75 ± 5.51	43	73
Mustang	59	53.23 ± 5.23	35	63
Total	876	53.67 ± 5.62	43	63

Table 4. The number of tagged Arkansas River Shiner *Notropis girardi* and Emerald Shiner *Notropis atherinoides* recaptured within their tagging site (i.e., movements between sites are not included) and the direction and average distance (m) moved in the first field season (November 2018-June 2019). Average movement distances are reported for upstream and downstream movement (mean ± standard deviation).

	Arkansas River Shiner	**Emerald Shiner**
Number recaptured	21	4
Moved downstream	15	3
No movement	4	1
Moved upstream	2	0
Mean ± SD downstream	-286.67 ± 168.47	-300.00 ± 200.00
Mean ± SD upstream	700.00 ± 141.42	-

Table 5. The number of tagged Arkansas River Shiner *Notropis girardi* and Emerald Shiner *Notropis atherinoides* recaptured within their tagging site (i.e., movements between sites are not included) and the direction and average distance (m) moved in the second field season (November 2019-June 2020). Average movement distances are reported for upstream and downstream movement (mean ± standard deviation).

	Arkansas River Shiner	**Emerald Shiner**
Number recaptured	69	36
Moved downstream	43	13
No movement	6	19
Moved upstream	20	4
Mean ± SD downstream	-192.26 ± 179.64	-219.69 ± 159.53
Mean ± SD upstream	255.35 ± 204.21	164.56 ± 68.56

Table 6. Recaptured Arkansas River Shiner *Notropis girardi* and Emerald Shiner *Notropis atherinoides* that moved out of their tagging site. The tag and recapture sites, tag and recapture dates, distance moved (km), and days between tagging and recapture events are reported.

Species	**Tag site**	**Recapture site**	**Tag date**	**Recapture date**	**Distance**	**Days**
Arkansas River Shiner	Caddo	Fire Canyon	1/6/2020	5/21/2020	30.23	136
Arkansas River Shiner	Mustang	Fire Canyon	11/18/2019	6/16/2020	91.94	211
Arkansas River Shiner	Fire Canyon	Thomas	12/2/2019	6/18/2020	49.33	190
Emerald Shiner	Fire Canyon	Taloga	12/10/2019	6/17/2020	100.77	199

Table 7. Model averaged results for the top 5 models (ΔAICc < 2). Unconditional and conditional coefficients and standard error (SE) and relative importance values are reported. Models are listed in Table 1.

	Unconditional coefficients	**Unconditional SE**	**Conditional coefficients**	**Conditional SE**	**Relative importance value**
Temperature	-0.73429	1.38563	-2.78645	1.25204	0.147
Discharge	-0.56976	1.18584	-2.49892	1.16046	0.127
Photoperiod	-0.01843	0.04634	-0.05318	0.06596	0.193
Total length	0.18282	0.8789	0.62403	1.53668	0.164

FIGURES

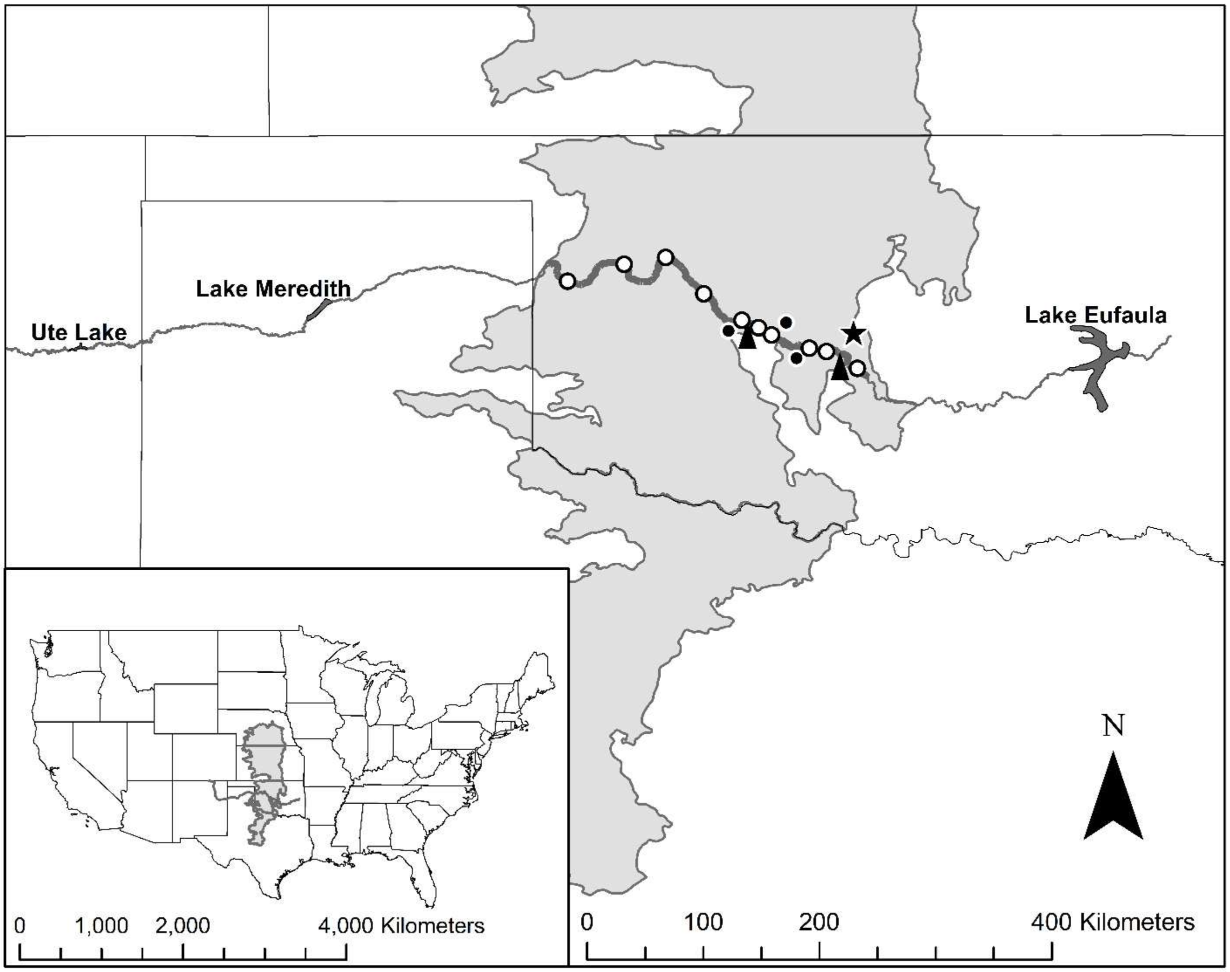

Figure 1. Map of the Canadian River (line) in the Central Great Plains ecoregion (EPA level 3, Woods et al. 2005; shaded area) of Oklahoma. All movement sampling occurred between Roll, OK and Purcell, OK (bold line). Sites (white dots) from left to right were located near the highway 283 bridge crossing north of Roll, the Broadway street bridge crossing south of Camargo, the highway 183 bridge crossing in Taloga, the highway 33 bridge crossing northeast of Thomas, Fire Canyon north of Bridgeport, private property off of Cherokee drive northeast of Hinton, the bridge crossing at south Ranch Road between Union City and Bridgeport, the South Banner Road bridge crossing southeast of Union City, and the

highway 4 bridge crossing south of Mustang, and the highway 35 bridge crossing southwest of Norman. The Mesonet stations (black dots) where I retrieved sunrise and sunset values, US Geological Service stream gages (black arrows) where I retrieved discharge data, and Oklahoma City (star) are shown.

Figure 2. Photos of an Arkansas River Shiner *Notropis girardi* with the Visible Implant Elastomer (VIE) tag locations represented. The top panel is a photo looking down on a fish from above and the bottom panel is a photo looking at the right side of a fish. VIE tags locations are 1) nape/anterior to the dorsal fin, 2) dorsal/laterally adjacent to the dorsal fin, 3) rear dorsal/posterior to the dorsal fin, 4) caudal/on the caudal peduncle.

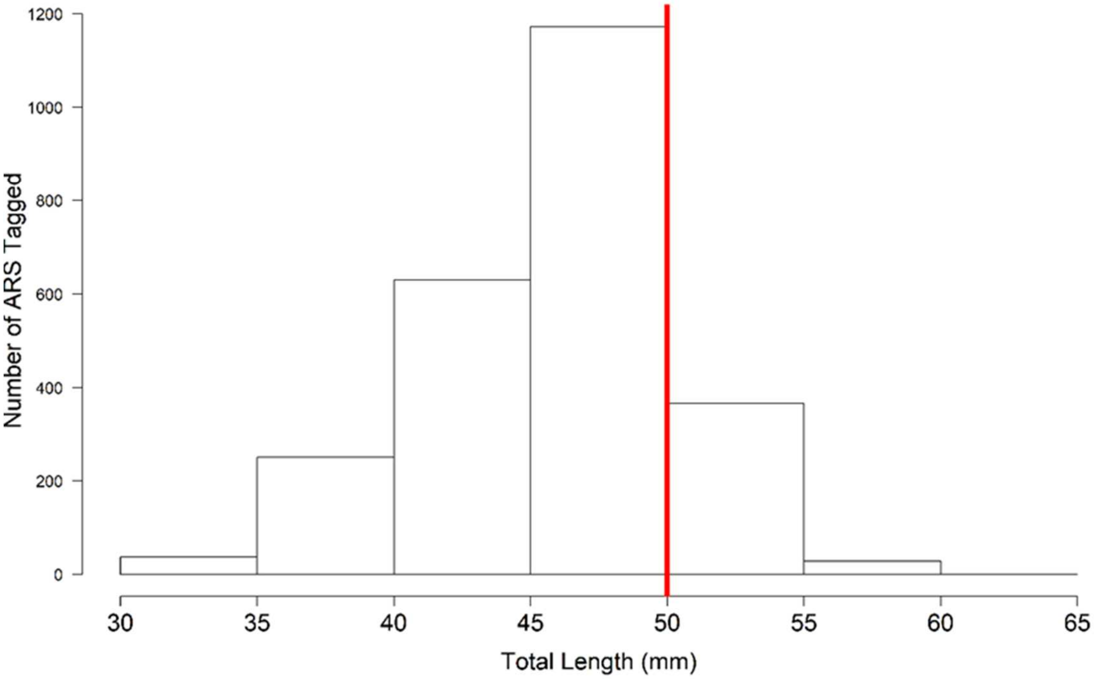

Figure 3. The number of Arkansas River Shiner *Notropis girardi* (ARS) in 5-mm bins of total lengths. Left of the line are individuals tagged with Visible Implant Elastomer (VIE) tags only and right of the line are individuals tagged with VIE and 8-mm Passive Integrated Transponder (PIT) tags that were inserted into the peritoneum.

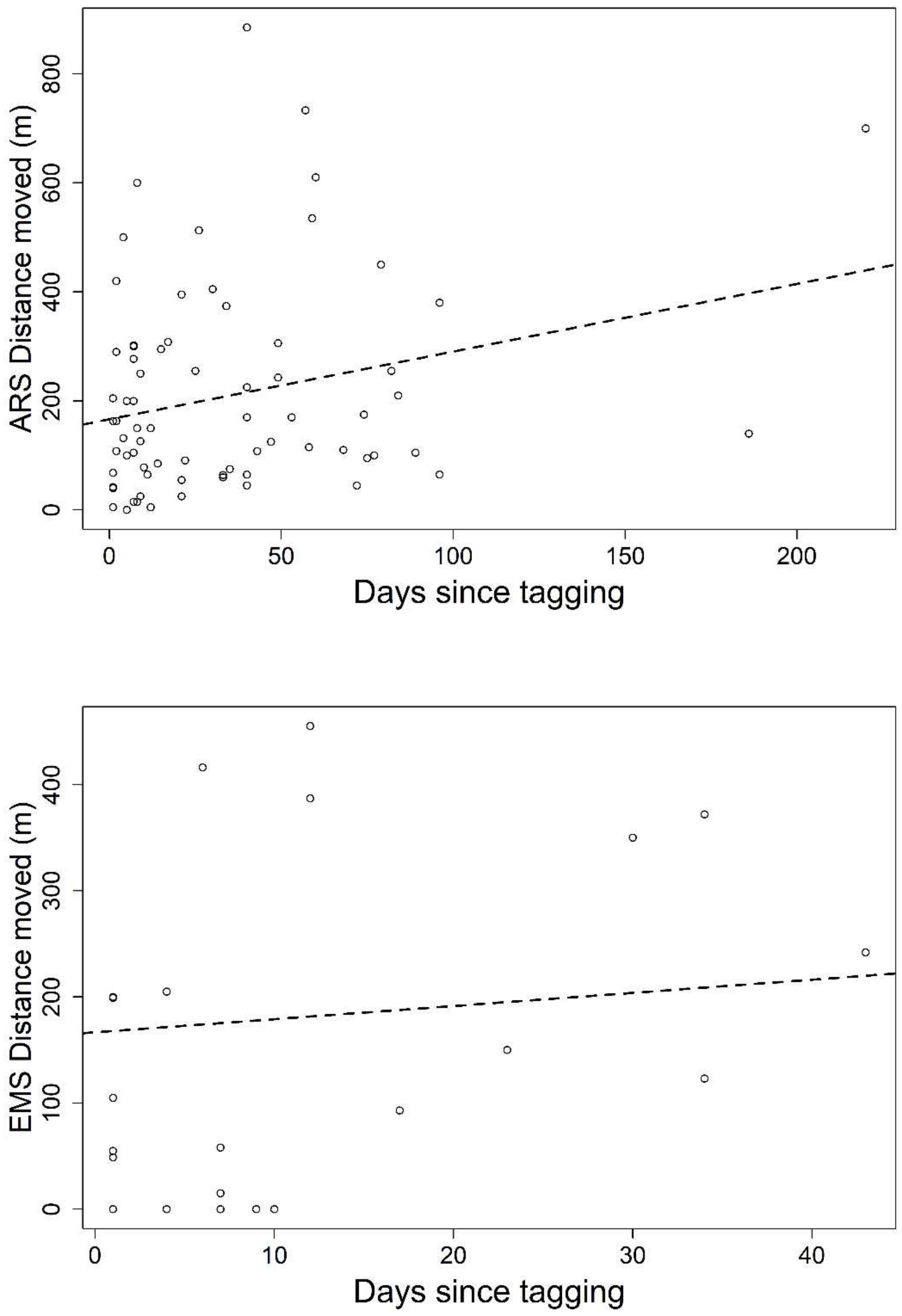

Figure 4. Relationship between days since tagging and non-directional (i.e., absolute value) distance moved (m) by tagged Arkansas River Shiner *Notropis girardi* (ARS) and Emerald Shiner *Notropis atherinoides* (EMS) in the Canadian River, Oklahoma. Only individuals that were recaptured within their tagging site were included in these data.

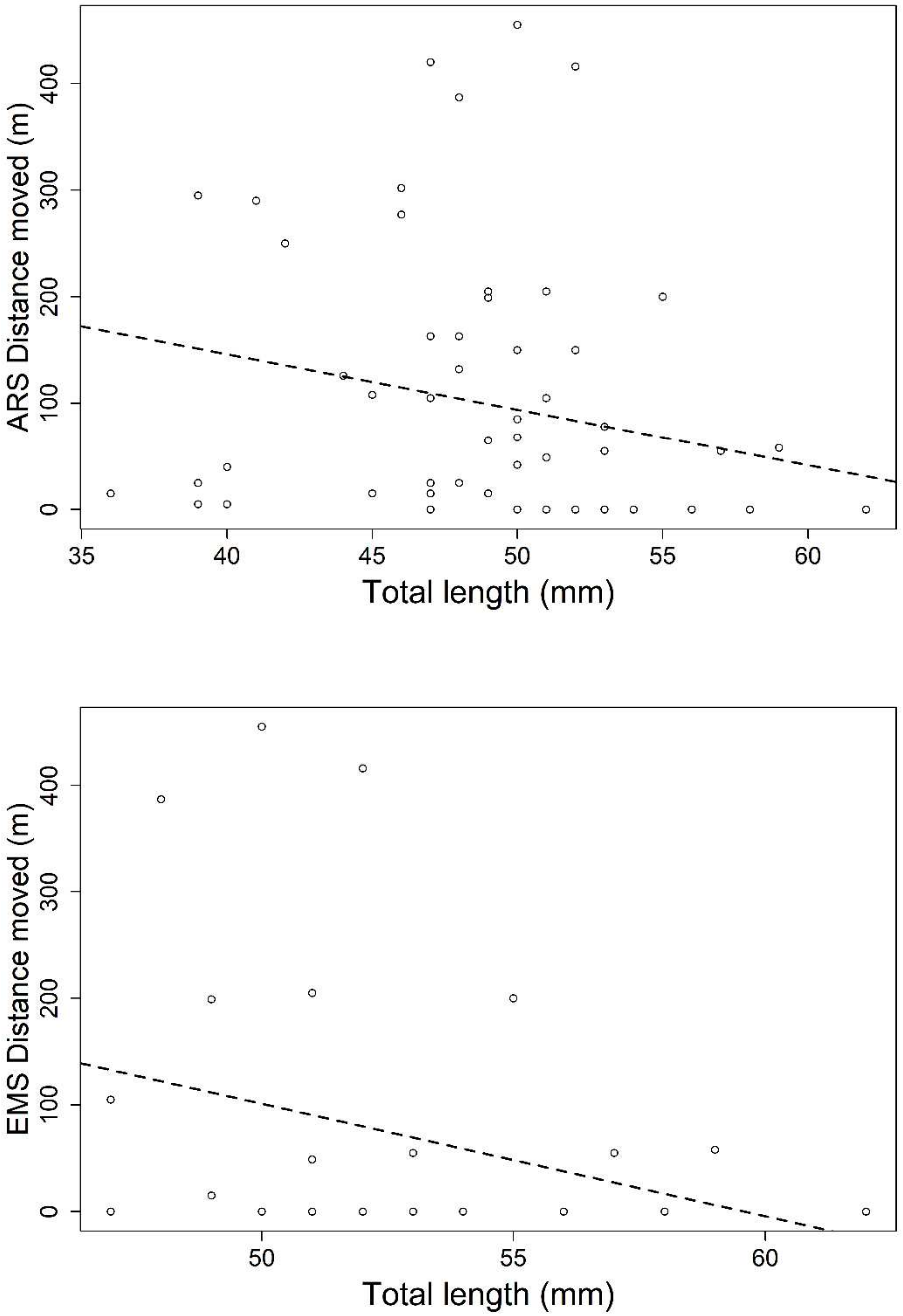

Figure 5. Relationship between total length (mm) and non-directional (i.e., absolute value) distance moved (m) by tagged Arkansas River Shiner *Notropis girardi* (ARS) and Emerald Shiner *Notropis atherinoides* (EMS) in the Canadian River, Oklahoma. Only individuals that were recaptured within their tagging site were included.

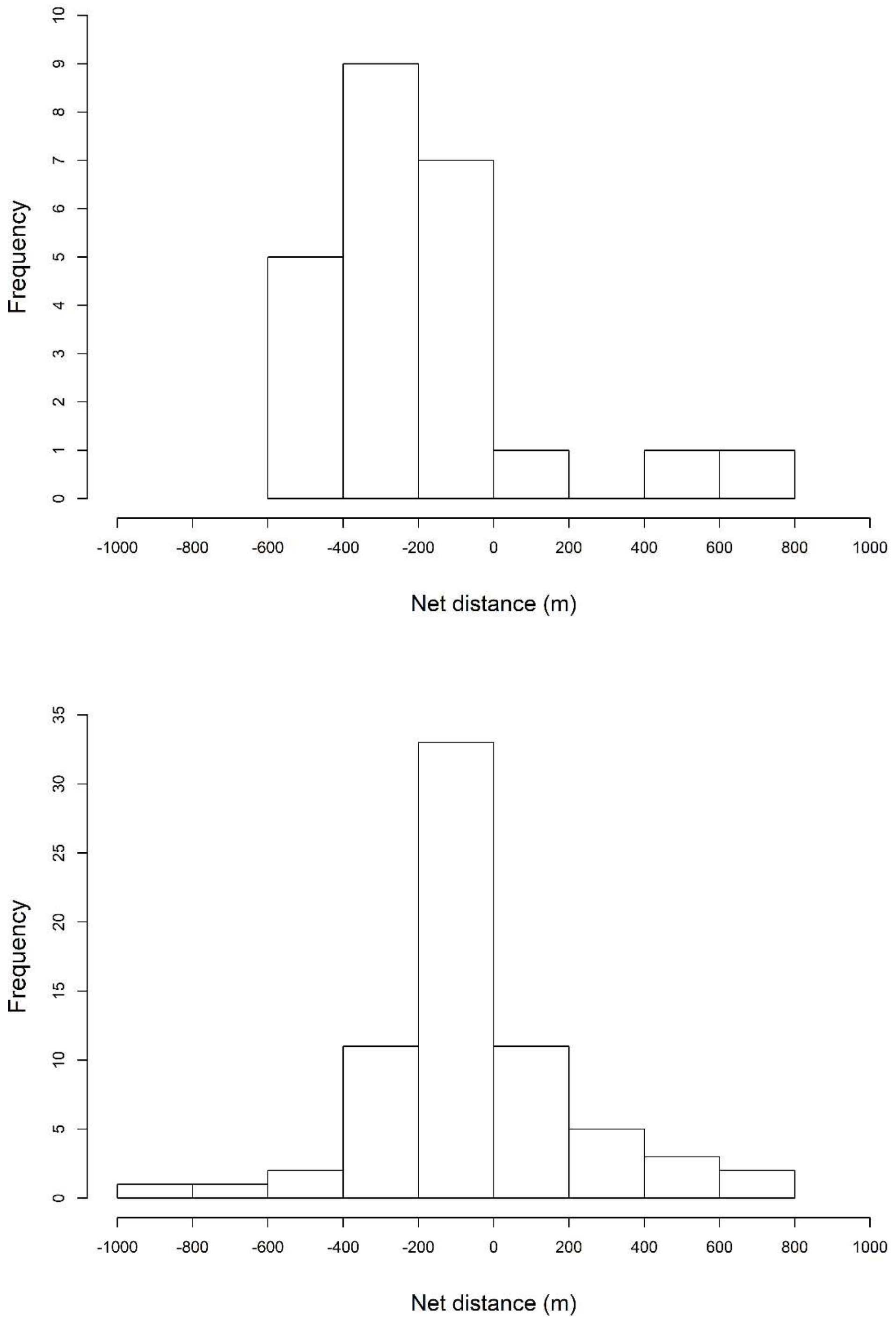

Figure 6. Net distances moved by Arkansas River Shiner *Notropis girardi* recaptured within their tagging site (i.e., movements among sites are not included). The top panel are fish recaptured in the first field season (November 2018-June 2019), and the bottom panel are fish recaptured in the second field season (November 2019-June 2020). Negative values indicate downstream movement, positive values indicate

upstream movement, and zero indicates the individual was recaptured in the same area (i.e., 100-m section for season 1 and within 20 m of tagging in season 2) of the river where it was tagged.

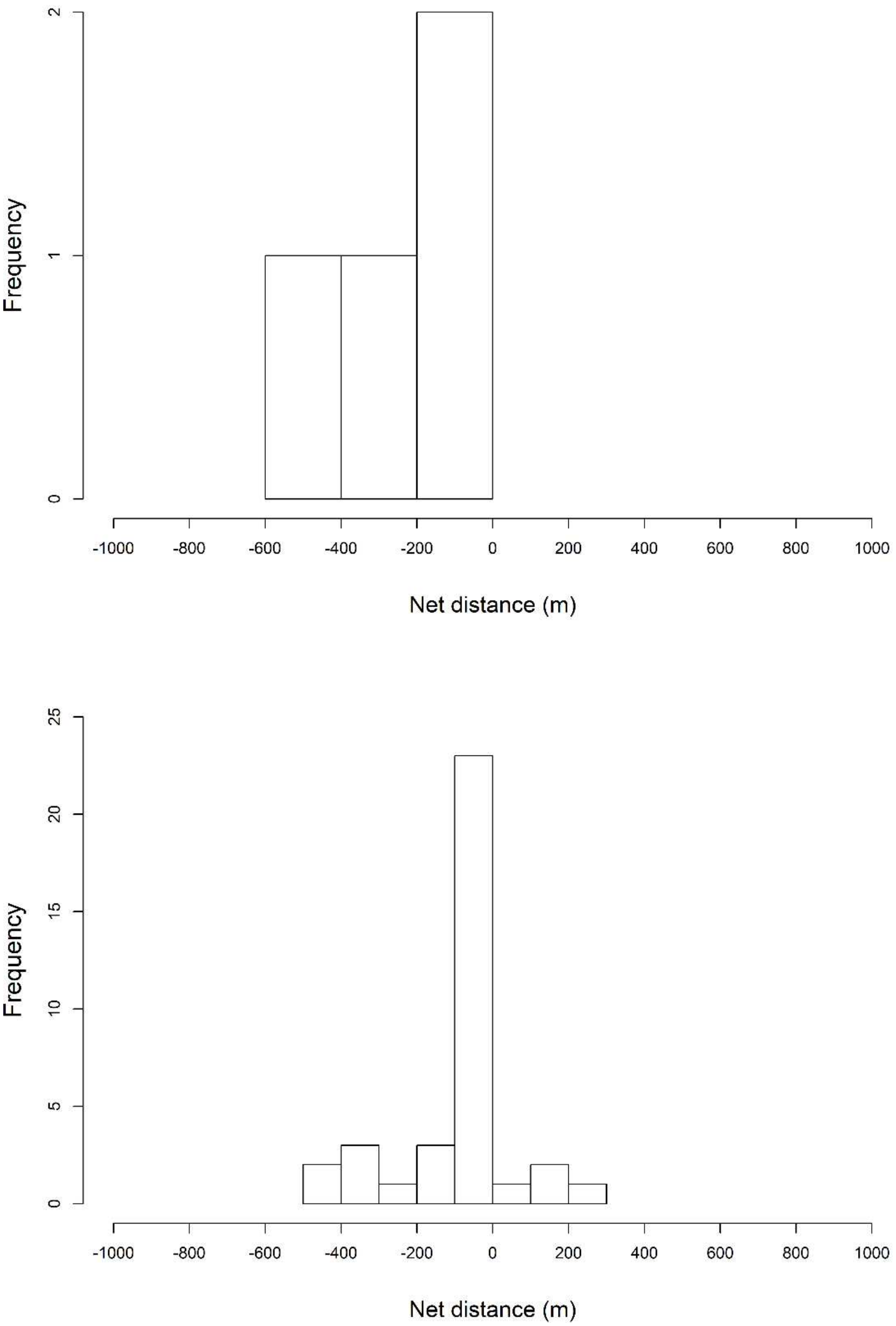

Figure 7. Net distances moved by Emerald Shiner *Notropis atherinoides* recaptured within their tagging site (i.e., movements among sites were not included). The top panel are fish recaptured in the first field season (November 2019-June 2020), and the bottom panel are fish recaptured in the second field season (November 2019-June 2020). Negative values indicate downstream movement, positive values indicate

upstream movement, and zero indicates the individual was recaptured in the same section (i.e., 100-m section for season 1 and within 20 m of tagging in season 2) of the river where it was tagged.

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APPENDICES

APPENDIX A: SUPPLEMENTAL TABLES FOR CHAPTER II

Table A.1. Flow metrics retained in the flow-ecology occupancy model selection for Arkansas River Shiner *Notropis girardi* and Plains Minnow *Hybognathus placitus*. Flow metric definitions are given (see Kennen et al. 2007 for calculations). I hypothesized these flow metrics were related to the ecology of pelagophils. The direction of each hypothesized relationship is represented as positive (+) or negative (-).

Metric	Metric definition	Ecological importance	Hypothesized relationship	Source
MA7	range in daily flows	Pelagophils benefit from a variety of magnitudes in discharge. Habitat complexity is maintained by low, average, and high magnitudes of flow over time. Pelagophils rely on a variety of habitats for their life cycle needs (e.g., foraging, spawning, refuge). Pelagophil propagule drift is also slowed by habitat complexity allowing them to develop in a shorter drift distance.	+	(Worthington et al. 2018)
MA36	variability across monthly flows	Pelagophils benefit from a variety of flow conditions throughout the year. For example, spring floods facilitate synchronous spawning indicating there is a benefit to spawning at higher discharge.	+	(Dudley 2004; Worthington et al. 2018)

		Alternatively, flooding in the winter may decrease overwinter survival because winter refuge habitats are needed.		
ML19	baseflow conditions using median flow	Higher base flows increase river connectivity. Pelagophils rely on connectivity to reach critical habitats and migrate upstream for spawning. Because much of Southern Great Plains rivers have decreased flows due to anthropogenic stressors (e.g., dewatering, damming), base flows are lower than the natural flow regime to which pelagophils are adapted.	+	(Durham and Wilde 2008a, 2009)
MH20	specific median annual maximum flow using median flow	High flow events facilitate synchronous spawning, river connectivity, energy flow from the floodplain to the river, and habitat complexity.	+	(Dudley 2004; Worthington et al. 2016, 2018)
FL3	frequency of low pulse spells using median flow	More low flow events may cause habitat homogenization and cause drifting propagules to settle to the bed and suffocate. River connectivity may also be reduced in one or more dimensions.	-	(Chase et al. 2015; Worthington et al. 2018)

FH2	**variability in high pulse count**	**Pelagophils benefit from a less variable high pulse count. There is an optimal number or range of high flow pulses to maintain habitat complexity, sediment transportation, and energy flow. A year where there are very many or very few high flow pulses would not provide optimal conditions for pelagophils.**	**-**	(Durham and Wilde 2008b; Worthington et al. 2018)
FH11	flood frequency using median flow	High flow events facilitate synchronous spawning, river connectivity, energy flow from the floodplain to the river, and habitat complexity.	+	(Durham and Wilde 2008b; Worthington et al. 2018)
DH15	high flow pulse duration	Longer high flow pulse durations may cause drifting propagules to wash into reservoirs before they are free-swimming individuals. High flow pulses facilitates synchronous spawning and increases the drift distance of propagules (i.e., they drift farther over time).	-	(Worthington et al. 2014a)
DH16	variability in high flow pulse duration	Pelagophils benefit from a less variable high pulse duration. There is an optimal range of high flow pulse duration for propagule drift distance. High flow pulses that are shorter than average may not provide enough drift distance for individuals to become free swimming, but higher than	-	(Durham and Wilde 2008b; Worthington et al. 2018)

		average flow pulse durations would wash propagules into reservoirs before they are free swimming.		
TA2	predictability	Predictability is important for cues that fishes have developed. Fishes may rely on cues for things like migration and spawning. Cues are less reliable with decreasing predictability. More predictable flows may also contribute to providing habitats that fishes are adapted to throughout the year (e.g., foraging habitat).	+	(Worthington et al. 2018)
TA3	**seasonal predictability of flooding**	**Seasonal predictability is important for cues triggering appropriate responses. Fishes may rely on seasonal cues for things like migration and spawning. Cues are less reliable with decreasing seasonal predictability.**	**+**	**(Worthington et al. 2018)**
RA4	variability in fall rate	Pelagophils benefit from a less variable fall rate. There is an optimal fall rate for the success of drifting propagules after a synchronous spawning event (e.g., a short dramatic fall might make propagules settle to the river bed and suffocate before they have time to develop).	-	(Dudley 2004; Durham and Wilde 2008a; Worthington et al. 2014a)

RA5	number of day rises	A higher number of day rises indicates more days that may cue synchronous spawning. There also may be a benefit from the change in flow leading to habitat complexity.	+	(Durham and Wilde 2008b; Worthington et al. 2018)

Table A.2. Pearson's product-moment coefficient (r) for predictor variables used in the occupancy model with selected flow metrics (bold) defined in Appendix Table A.1, season is the climatic season (cool or warm), spawning is the spawning period (spawning or non-spawning), precipitation gradient is the average annual precipitation for the site (cm), and river-fragment length is the distance (rkm) that the river is unimpeded by a major dam according to the dam metric database (Cooper et al. 2017).

	MA7	**MA36**	**ML19**	**MH20**	**FL3**	**FH2**	**FH11**	**DH15**	**DH16**	**TA2**	**TA3**
MA7	1.00										
MA36	0.34	1.00									
ML19	-0.39	0.29	1.00								
MH20	-0.16	-0.10	-0.06	1.00							
FL3	0.40	0.04	-0.35	-0.04	1.00						
FH2	0.19	0.35	0.08	-0.15	0.18	1.00					
FH11	0.17	0.30	-0.13	-0.15	0.08	0.36	1.00				
DH15	0.09	-0.09	-0.12	-0.08	-0.02	0.22	0.01	1.00			
DH16	0.08	0.18	0.00	-0.22	-0.09	0.45	0.26	-0.08	1.00		
TA2	-0.49	-0.10	0.47	-0.10	-0.31	-0.02	-0.15	0.06	-0.13	1.00	
TA3	-0.02	-0.06	0.17	-0.05	-0.18	-0.09	-0.39	0.08	-0.06	0.22	1.00
RA4	0.15	0.33	0.02	-0.05	-0.12	-0.11	0.10	-0.13	0.18	-0.40	-0.17
RA5	0.07	0.04	0.01	-0.18	0.47	0.36	0.24	-0.05	-0.06	0.05	0.00

Season	0.10	0.09	-0.01	-0.13	0.08	0.19	-0.03	-0.14	0.33	-0.22	-0.18
Spawning	0.06	0.05	-0.03	-0.12	0.08	0.13	-0.01	0.14	0.02	-0.07	0.06
Precipitation gradient	-0.11	-0.15	0.27	0.12	-0.09	0.00	-0.33	-0.03	-0.07	0.47	0.19
Disturbance index	-0.09	-0.10	0.03	0.33	0.16	0.08	-0.26	0.02	-0.18	0.04	0.13
River-fragment length	-0.13	-0.13	0.03	-0.01	-0.16	-0.15	-0.13	-0.03	-0.03	0.01	0.07

Table A.2. Cont'd. Pearson's product-moment coefficient (r) for predictor variables used in the occupancy model with selected flow metrics (bold) defined in Appendix Table A.1, season is the climatic season (cool or warm), spawning is the spawning period (spawning or non-spawning), precipitation gradient is the average annual precipitation for the site (cm), and river-fragment length is the distance (rkm) that the river is unimpeded by a major dam according to the dam metric database (Cooper et al. 2017).

	RA4	**RA5**	Season	Spawning	Precipitation gradient	Disturbance index	River-fragment length
MA7							
MA36							
ML19							
MH20							
FL3							
FH2							
FH11							
DH15							
DH16							
TA2							
TA3							
RA4	1.00						
RA5	-0.21	1.00					

Season	0.07	0.11	1.00				
Spawning	0.03	0.01	0.01	1.00			
Precipitation gradient	-0.48	0.13	0.11	-0.05	1.00		
Disturbance index	-0.25	0.05	0.07	-0.09	0.25	1.00	
River-fragment length	0.07	-0.05	0.11	-0.02	0.10	0.07	1.00

Table A.3. U.S. Geological Survey (USGS) stream gages within the study area with the dates each station is actively recording discharge data, ‘Dates active’. Stations are identified by station number, station name, and coordinate location given by USGS. I used 80 gages (bold) for the analysis. Gages were selected based on the ranges of my seven study species, historical fish records, and 20+ years of discharge data. See Figure 1 for a map of gage locations.

Station number	Station name	Longitude	Latitude	Dates active
7148140	Arkansas River near Ponca City, OK	-96.9303139	36.69337167	04/22/1976-10/12/2000
7148350	Salt Fork Arkansas River nr Winchester, OK	-98.7823117	36.9616958	06/01/1960-09/25/1961
7148400	**Salt Fork Arkansas River nr Alva, OK**	**-98.6481394**	**36.81503056**	**04/01/1938-Current**
7148450	Salt Fork Arkansas River nr Ingersoll, OK	-98.3600733	36.8216958	09/01/1961-09/29/1979
7149500	Salt Fork Arkansas River nr Cherokee, OK	-98.3192386	36.8183625	10/01/1940-09/29/1950
7150500	**Salt Fork Arkansas River nr Jet, OK**	**-98.1289547**	**36.75252944**	**10/01/1937-09/30/1993**
7151000	**Salt Fork Arkansas River at Tonkawa, OK**	**-97.3094886**	**36.67197917**	**10/01/1935-Current**
7152000	**Chikaskia River near Blackwell, OK**	**-97.277265**	**36.8114211**	**04/01/1936-Current**
7152290	Greasy Creek near Watchorn, OK	-96.9942053	36.45615667	07/18/1974-06/30/1976
7152500	Arkansas River at Ralston, OK	-96.7283672	36.50421667	10/01/1925-Current

7153000	Black Bear Creek at Pawnee, OK	-96.7994789	36.34366528	07/29/1944-Current
7153100	Ranch Creek at Clev Dam nr Cleveland, OK	-96.576693	36.2833983	12/01/1944-09/29/1963
7153410	BENNETT SPRING NR CAPULIN, NM	-103.9174675	36.76780247	07/12/1977-10/14/1981
7153500	DRY CIMARRON R NR GUY, NM	-103.4241098	36.9875204	10/01/1942-12/31/1973
7154000	CIMARRON R NR FOLSOM, NM	-103.0991	36.93474298	10/01/1927-09/29/1933
7154500	Cimarron River near Kenton, OK	-102.9590956	36.92668694	10/01/1950-Current
7155000	Cimarron River ab Ute Creek nr Boise City, OK	-102.6193594	36.9127986	10/01/1905-09/29/1954
7155500	Cimarron River near Boise City, OK	-102.5196333	36.9189097	10/01/1938-09/29/1942
7157000	Cimarron River near Mocane, OK	-100.3143114	36.9758611	10/01/1942-09/29/1965
7157580	Cimarron River near Englewood, KS	-99.9759683	36.97725194	03/11/1982-09/29/1987
7157950	**Cimarron River near Buffalo, OK**	**-99.3153869**	**36.8519747**	**05/01/1990-Current**
7157960	Buffalo Creek near Lovedale, OK	-99.3670536	36.7705872	08/01/1966-09/30/1993
7157980	Cimarron River at Freedom, OK	-99.11649	36.75503167	10/01/1973-09/29/1980
7158000	**Cimarron River near Waynoka, OK**	**-98.8795367**	**36.51725639**	**10/01/1937-Current**
7158150	Salt Creek near Hitchcock, OK	-98.3703519	36.01559417	08/01/1968-09/29/1970
7158400	Salt Creek near OKeene, OK	-98.193678	36.1030922	07/01/1961-09/29/1979
7158500	Preacher Creek near Dover, OK 18N-08W-13 BBB	-98.0136703	36.041705	10/01/1951-04/30/1957

7159000	Turkey Creek near Drummond, OK	-98.0011719	36.31809	10/01/1947-09/29/1970
7159100	**Cimarron River near Dover, OK**	**-97.9144994**	**35.9517075**	**10/01/1973-Current**
7159200	Kingfisher Creek near Kingfisher, OK	-98.0661686	35.8342097	10/01/1966-09/29/1970
7159500	Bluff Creek ab Lake Hefner nr Oklahoma City, OK	-97.5964303	35.54255639	03/01/1950-09/29/1958
7159720	Cottonwood Creek near Navina, OK	-97.5461539	35.7767158	10/01/1977-09/29/1989
7159750	Cottonwood Creek near Seward, OK	-97.4780969	35.81366	03/01/1973-Current
7160000	**Cimarron River near Guthrie, OK**	**-97.4258744**	**35.9206022**	**10/01/1937-Current**
7160350	Skeleton Creek at Enid, OK	-97.8003333	36.376145	02/17/1996-Current
7160500	Skeleton Creek near Lovell, OK	-97.5850472	36.06004056	10/01/1949-Current
7161000	Cimarron River at Perkins, OK	-97.031975	35.95755639	10/01/1939-09/30/1991
7161450	**Cimarron River near Ripley, OK**	**-96.9122506**	**35.98589278**	**10/01/1987-Current**
7162000	Stillwater Creek at Stillwater, OK	-97.0469783	36.09810778	10/01/1934-09/29/1937
7162500	West Fork Brush Creek near Stillwater, OK	-97.0053114	36.1167197	10/01/1934-09/29/1937
7163000	Council Creek near Stillwater, OK	-96.8678089	36.11616917	04/01/1934-09/30/1993
7163500	Cimarron River at Oilton, OK	-96.5814097	36.09395639	10/01/1934-09/30/1945
7164000	Cimarron River at Mannford, OK	-96.3986264	36.1589636	10/01/1938-06/30/1963
7164500	**Arkansas River at Tulsa, OK**	**-96.0063867**	**36.14064806**	**10/01/1925-Current**

7164600	Joe Creek at 61st St at Tulsa, OK	-95.96055	36.07565056	03/11/1988-Current
7164650	Fred Creek at Evanston Avenue, at Tulsa, OK	-95.946938	36.05231778	10/02/1991-09/29/1992
7165500	Polecat Creek blw Heyburn Res near Heyburn, OK	-96.29445	35.94508139	10/01/1943-09/29/1979
7165550	Snake Creek near Bixby, OK	-95.8886011	35.81898917	07/01/1961-09/29/1970
7165562	Haikey Creek at 101st St South at Tulsa, OK	-95.8488778	36.01704306	01/20/1988-Current
7165564	Little Haikey Tributary at S. Memorial, Tulsa, OK	-95.8861017	36.0362083	04/01/1991-09/29/1992
7165565	Little Haikey Creek at 101st St South at Tulsa, OK	-95.8608228	36.0175983	10/01/1987-Current
7165570	**Arkansas River near Haskell, OK**	**-95.6388697**	**35.82093389**	**06/01/1972-Current**
7165600	Arkansas River near Tullahassee, OK	-95.4030256	35.8042661	10/01/1969-05/31/1972
7171000	Verdigris River near Lenapah, OK	-95.586088	36.8511961	10/01/1938-Current
7171400	Verdigris River near Oologah, OK	-95.6844256	36.42065028	06/01/1961-09/29/1992
7171500	Verdigris River near Sageeyah, OK	-95.6710917	36.39176194	01/01/1939-09/29/1945
7173000	Caney River near Hulah, OK	-96.0852639	36.92701389	10/01/1937-09/29/1993
7174000	Little Caney River near Copan, OK	-95.9349808	36.97090778	10/01/1943-09/29/1958
7174200	Little Caney River blw Cotton Cr, nr Copan, OK	-95.9694267	36.8950786	10/01/1958-02/10/1981
7174310	Little Caney River below Copan Lake nr Copan, OK	-95.9747047	36.88480139	04/30/1991-05/30/1991
7174400	Caney River above Coon Creek at Bartlesville, OK	-95.9722064	36.75564417	10/01/1985-Current

7174500	Caney River at Bartlesville, OK	-95.9602619	36.74508917	10/01/1949-10/06/1993
7174600	Sand Creek at Okesa, OK	-96.1324936	36.7195233	10/01/1959-09/29/1993
7174700	Caney River near Ochelata, OK	-95.934153	36.6406458	04/01/1956-09/29/1976
7175000	Double Creek Sws 5 near Ramona, OK	-95.9405453	36.5139808	12/01/1954-09/29/1969
7175500	Caney River near Ramona, OK	-95.8419306	36.50898167	10/01/1945-Current
7175550	Caney River near Collinsville, OK	-95.8102639	36.39509417	10/01/1935-09/29/1938
7176000	Verdigris River near Claremore, OK	-95.6980372	36.3073186	10/01/1935-Current
7176465	Birch Creek blw Birch Lake nr Barnsdall, OK	-96.1622239	36.53341417	02/17/1977-11/30/1992
7176500	Bird Creek at Avant, OK	-96.0641633	36.48675444	10/01/1945-Current
7176800	Candy Creek near Wolco, OK	-96.0486056	36.53508806	10/01/1969-05/18/1981
7176950	Hominy Creek near Hominy, OK	-96.3789064	36.47367944	10/01/2003-Current
7177000	Hominy Creek near Skiatook, OK	-96.1100025	36.34869694	04/01/1944-09/29/1981
7177410	Hominy Creek below Skiatook Lake nr Skiatook, OK	-96.0886122	36.35258694	10/01/1984-09/30/1993
7177500	Bird Creek near Sperry, OK	-95.9541622	36.27842528	10/01/1938-Current
7177600	Bird Creek at 66th Street near Tulsa, OK	-95.9433286	36.24925889	06/17/1987-06/30/1990
7177650	Flat Rock Creek at Cincinnati Ave at Tulsa, OK	-95.9952758	36.21536944	12/01/1987-Current
7177700	Flat Rock Creek at Us Hwy 75 at Tulsa, OK	-95.9447175	36.2256483	04/23/1987-06/30/1991

7177800	Coal Creek at Tulsa, OK	-95.9141597	36.1945383	01/30/1988-Current
7178000	Bird Creek near Owasso, OK	-95.8686019	36.24870528	10/01/1935-Current
7178035	Mingo Creek at 36th Street North at Tulsa, OK	-95.8591567	36.20620639	09/14/1987-06/02/1991
7178040	Mingo Creek at 46th Street North at Tulsa, OK	-95.8586011	36.22065056	04/24/1987-06/30/1998
7178200	Bird Ck at State Highway 266 near Catoosa, OK	-95.8194322	36.22315139	08/01/1988-Current
7178520	Dog Creek South of Claremore, OK	-95.6113678	36.27870806	08/29/1997-09/29/2004
7178600	Verdigris River near Inola, OK	-95.6199792	36.16426528	10/01/1944-09/29/1970
7185000	Neosho River near Commerce, OK	-94.9574575	36.92868139	10/01/1939-Current
7185095	Tar Creek at 22nd Street Bridge, Miami, OK	-94.8682878	36.90006944	01/11/1984-Current
7185100	Tar Creek at Miami, OK	-94.8624547	36.8747922	08/14/1980-01/10/1984
7190600	Big Cabin Creek near Pyramid Corners, OK	-95.163578	36.8017447	10/01/1963-09/29/1972
7191000	Big Cabin Creek near Big Cabin, OK	-95.1521892	36.56841778	10/01/1947-Current
7192000	Pryor Creek near Pryor, OK	-95.3258025	36.2812058	10/01/1947-09/29/1963
7192500	Neosho River near Wagoner, OK	-95.2691283	35.92898806	04/01/1924-09/29/1949
7194500	Arkansas River near Muskogee, OK	-95.2971867	35.7695433	07/25/2003-Current
7198500	Dirty Creek near Warner, OK	-95.3080233	35.55509806	10/01/1939-09/29/1946
7199000	CANADIAN RIVER NEAR HEBRON, NM	-104.4622099	36.78724913	10/02/1946-09/30/1986

7200000	CHICORICA BL EF N RATON, NM	-104.3783181	36.90141365	10/01/1945-06/30/1951
7200500	CHICORICA CREEK NEAR RATON, NM	-104.3935966	36.80891515	10/01/1983-09/29/1987
7201500	UNA DE GATO C NR HEBRON, NM	-104.3908189	36.77224905	10/01/1946-06/30/1950
7202000	CHICORICA CREEK NEAR HEBRON, NM	-104.3963746	36.77030466	02/01/1945-09/29/1987
7202500	EAGLE TAIL DITCH NR MAXWELL, NM	-104.5591568	36.64864146	01/01/1945-Current
7203505	VERMEJO DITCH NEAR COLFAX, NM	-104.6930493	36.57836704	12/20/1980-09/29/1996
7203525	VERMEJO RIVER NEAR MAXWELL, NM	-104.5713792	36.49669977	11/25/1983-09/29/1994
7208500	RAYADO CREEK NEAR CIMARRON, NM	-104.9700026	36.37225823	10/01/1911-Current
7211000	**CIMARRON RIVER AT SPRINGER, NM**	**-104.5986002**	**36.3603112**	**10/01/1907-09/29/2004**
7211500	**CANADIAN RIVER NEAR TAYLOR SPRINGS, NM**	**-104.493874**	**36.29697658**	**10/01/1939-Current**
7214000	CANADIAN RIVER NEAR ROY, NM	-104.3533148	35.9194805	04/01/1936-09/29/1965
7216500	MORA RIVER NEAR GOLONDRINAS, NM	-105.1636194	35.89087299	04/01/1915-Current
7218000	COYOTE CREEK NEAR GOLONDRINAS, NM	-105.1641751	35.91670581	10/01/1929-Current
7218100	MORA R NR WATROUS, NM	-105.0400041	35.83476305	10/01/1956-09/29/1963
7220000	SAPELLO RIVER AT SAPELLO, NM	-105.2519548	35.7697644	01/01/1917-12/31/1973
7220100	LK ISABEL CA NR SAPELLO, NM	-105.1575072	35.74504247	10/01/1964-06/30/1975
7220600	SAPELLO R NR WATROUS, NM	-105.0416704	35.76809741	10/01/1956-09/29/1963

7221000	MORA RIVER NR SHOEMAKER, NM	-104.7833275	35.80031784	10/01/1919-09/29/1996
7221500	**CANADIAN RIVER NEAR SANCHEZ, NM**	**-104.378037**	**35.65226419**	**10/01/1912-Current**
7222000	CANADIAN RIVER NEAR BELL RANCH, NM	-104.2505335	35.5000457	10/01/1929-06/30/1939
7222500	CONCHAS RIVER AT VARIADERO, NM	-104.4435938	35.40282567	10/01/1936-09/29/1996
7223000	BELL RANCH CANAL BELOW CONCHAS DAM, NM	-104.1858095	35.40282609	10/01/1970-10/11/1984
7223300	CONCHAS CANAL BELOW CONCHAS DAM, NM	-104.1680313	35.37643795	10/01/1970-09/29/1992
7224500	CANADIAN RIVER BELOW CONCHAS DAM, NM	-104.1699758	35.40893713	05/01/1936-09/29/1972
7226500	UTE CREEK NEAR LOGAN, NM	-103.5257944	35.43838224	01/01/1942-Current
7227000	**CANADIAN RIVER AT LOGAN, NM**	**-103.4180137**	**35.35699577**	**01/01/1909-Current**
7227100	**REVUELTO CREEK NEAR LOGAN, NM**	**-103.3949576**	**35.34116298**	**08/01/1959-Current**
7227200	TRAMPEROS CREEK NEAR STEAD, NM	-103.2032752	36.07086764	06/17/1966-12/31/1973
7227448	Punta De Agua Ck nr Channing, TX	-102.4804791	35.66754301	10/01/1967-09/29/1973
7227470	Canadian Rv at Tascosa, TX	-102.2601957	35.51893636	10/01/1968-10/19/1977
7227500	**Canadian Rv nr Amarillo, TX**	**-101.8796281**	**35.47032612**	**04/01/1938-Current**
7227920	Dixon Ck nr Borger, TX	-101.3509971	35.66476491	03/01/1974-09/29/1989
7228000	**Canadian Rv nr Canadian, TX**	**-100.3706884**	**35.935042**	**04/01/1983-Current**
7228400	Deer Creek at Hydro, OK	-98.5781258	35.5411625	10/01/1960-09/29/1980

7228500	**Canadian River at Bridgeport, OK**	**-98.3178389**	**35.54366167**	**10/01/1944-Current**
7229000	Canadian River near Newcastle, OK	-97.5986494	35.30089667	10/01/1938-09/29/1945
7229050	Canadian River at Norman, OK	-97.4850339	35.1945111	02/01/1996-Current
7229100	Canadian River near Noble, OK	-97.3814186	35.0820147	10/01/1959-09/29/1975
7229200	**Canadian River at Purcell, OK**	**-97.347528**	**35.0139611**	**10/01/1959-08/20/2018**
7229300	Walnut Creek at Purcell, OK	-97.3669728	34.9989611	10/01/1965-Current
7229427	Canadian Sandy Creek near Ada, OK	-96.7139017	34.78453028	10/01/1986-09/29/1988
7229500	Little River near Norman, OK	-97.3058633	35.2278461	10/01/1951-09/29/1955
7230000	Little River blw Lk Thunderbird nr Norman, OK	-97.2139164	35.22173639	10/01/1952-Current
7230500	Little River near Tecumseh, OK	-96.9319661	35.17257417	10/01/1943-Current
7230597	Little River near Bowlegs, OK	-96.668628	35.10535667	01/26/1983-09/29/1988
7230800	Salt Creek near Dewright, OK	-96.6669611	35.05008028	10/01/1959-09/29/1967
7231000	Little River near Sasakwa, OK	-96.5125114	34.96536139	10/01/1942-Current
7232250	Beaver River near Felt, OK	-102.6815814	36.62974667	10/01/1980-06/30/2002
7232470	Beaver River near Goodwell, OK	-101.6368272	36.7058583	10/01/2001-06/30/2003
7232500	Beaver River near Guymon, OK	-101.4896047	36.721415	10/01/1937-09/30/1993
7232900	Coldwater Creek near Guymon, OK	-101.3815489	36.57197139	10/01/1980-09/30/2010

7233000	Coldwater Creek near Hardesty, OK	-101.210993	36.64391556	10/01/1939-09/29/1964
7233210	Beaver River near Hardesty, OK	-101.1354375	36.65641556	10/02/1977-09/29/1986
7233500	Palo Duro Ck nr Spearman, TX	-101.305993	36.20225497	06/17/1999-Current
7233650	Palo Duro Creek at Range, OK	-101.0809906	36.54391667	10/01/1991-09/29/2010
7234000	**Beaver River at Beaver, OK**	**-100.5193158**	**36.82224806**	**10/01/1990-Current**
7234100	Clear Creek near Elmwood, OK	-100.5023689	36.64502778	10/01/1965-09/30/1993
7234500	Beaver River near Fort Supply, OK	-99.5920597	36.59170139	10/01/1937-04/30/1951
7235000	Wolf Ck at Lipscomb, TX	-100.2756889	36.23864885	10/01/1940-Current
7235500	Wolf Creek near Shattuck, OK	-99.9129	36.28615056	10/01/1937-09/29/1946
7236000	Wolf Creek near Fargo, OK	-99.6231706	36.39920444	10/01/1942-09/29/1976
7237000	Wolf Creek near Fort Supply, OK	-99.5517806	36.56670167	10/01/1937-10/03/1993
7237500	**North Canadian River at Woodward, OK**	**-99.2784383**	**36.4367033**	**01/01/1905-Current**
7237800	Bent Creek near Seiling, OK	-99.0103753	36.19059528	10/01/1966-09/29/1970
7238000	**North Canadian River near Seiling, OK**	**-98.9212058**	**36.18337278**	**10/01/1946-Current**
7239200	North Canadian River near Watonga, OK	-98.4653525	35.8450436	10/01/1979-09/29/1983
7239300	**North Canadian River blw Weavers Ck nr Watonga, OK**	**-98.4209053**	**35.8119886**	**10/01/1983-Current**
7239450	North Canadian River near Calumet, OK	-98.0653317	35.6169933	10/01/1988-Current

7239500	**North Canadian River near El Reno, OK**	**-97.9575497**	**35.56310667**	**10/01/1902-Current**
7239700	North Canadian River near Yukon, OK	-97.7425442	35.5394986	10/01/1999-Current
7241000	North Canadian River blw Lk Overholser nr OKC, OK	-97.6633753	35.4786686	10/01/1952-Current
7241500	North Canadian River near Oklahoma City, OK	-97.4280922	35.49450528	11/01/1938-06/30/1961
7241520	North Canadian River at Britton Rd at OKC, OK	-97.367258	35.5656147	10/01/1988-Current
7241550	**North Canadian River near Harrah, OK**	**-97.1939194**	**35.5003411**	**10/01/1968-Current**
7241750	Tecumseh Creek at Tecumseh, OK	-96.9708561	35.28146	07/17/1991-09/30/1992
7241800	North Canadian River at Shawnee, OK	-96.8694661	35.33312667	02/07/2001-Current
7242000	**North Canadian River near Wetumka, OK**	**-96.2061167**	**35.26564556**	**10/01/1937-Current**
7242100	Wewoka Creek near Wetumka, OK	-96.2197283	35.2209225	10/01/1959-09/29/1967
7242247	Deep Fork at Hefner Rd at Oklahoma City, Ok	-97.42726	35.58005778	10/01/1995-06/30/1998
7242350	Deep Fork near Arcadia, OK	-97.3600364	35.64727889	10/01/1969-05/31/1994
7242380	Deep Fork at Warwick, OK	-97.0083614	35.6808958	10/01/1983-Current
7242500	Bellcow Creek at Chandler, OK	-96.8891917	35.70228667	10/01/1948-09/29/1955
7243000	Dry Creek near Kendrick, OK	-96.8541914	35.7820086	10/01/1955-09/29/1994
7243500	Deep Fork near Beggs, OK	-96.0686078	35.6739875	09/02/1938-Current
7244000	Deep Fork near Dewar, OK	-95.882773	35.478715	10/01/1937-09/29/1950

7244100	Coal Creek near Henryetta, OK	-95.9558308	35.45288167	03/20/1996-09/29/2004
7295500	Tierra Blanca Ck abv Buffalo Lk nr Umbarger, TX	-102.1760295	34.84867146	10/19/2002-Current
7296100	Tierra Blanca Ck bl Buffalo Lk nr Umbarger, TX	-102.0996401	34.92422518	04/01/1967-09/29/1973
7297500	Pr Dog Twn Fk Red Rv nr Canyon, TX	-101.8918571	35.01061119	01/15/1924-09/29/1949
7297910	**Pr Dog Twn Fk Red Rv nr Wayside, TX**	**-101.4140578**	**34.83755505**	**10/01/1967-Current**
7298200	Tule Ck nr Silverton, TX	-101.42989	34.54339522	08/02/1964-09/29/1986
7298500	Pr Dog Twn Fk Red Rv nr Brice, TX	-100.9406994	34.62783574	09/21/2002-Current
7299000	Mulberry Ck nr Brice, TX	-100.9170876	34.67505649	10/01/1949-07/31/1951
7299200	Pr Dog Twn Fk Red Rv nr Lakeview, TX	-100.7456898	34.57311318	06/01/1963-09/29/1980
7299300	Little Red Rv nr Turkey, TX	-100.7706906	34.54089189	08/01/1968-09/29/1981
7299500	Pr Dog Twn Fk Red Rv nr Estelline, TX	-100.4365099	34.57228021	04/01/1938-06/30/1947
7299512	Jonah Creek near Estelline, TX	-100.3337294	34.57228125	05/22/1974-10/04/1982
7299514	Jonah Ck bl Weir nr Estelline, TX	-100.3395628	34.5592259	05/23/1974-10/06/1976
7299530	Salt Ck nr Estelline, TX	-100.2526163	34.59061508	05/25/1974-05/16/1979
7299540	**Pr Dog Twn Fk Red Rv nr Childress, TX**	**-100.1940023**	**34.56922728**	**04/01/1965-Current**
7299570	Red River near Quanah, TX	-99.7356497	34.41340694	12/01/1959-10/05/1982
7299670	Groesbeck Ck at SH 6 nr Quanah, TX	-99.74037187	34.35451979	12/01/1961-Current

7299710	Sandy Creek near Eldorado, OK	-99.6787044	34.47923889	06/01/1960-12/31/1963
7299850	Salt Fk Red Rv nr Clarendon, TX	-100.8920888	35.00282777	06/01/1960-09/29/1964
7300000	**Salt Fk Red Rv nr Wellington, TX**	**-100.2209491**	**34.95755294**	**06/08/1952-Current**
7300500	**Salt Fork Red River at Mangum, OK**	**-99.5087008**	**34.8583936**	**10/01/1937-07/12/2018**
7300530	Bitter Creek near Martha, OK	-99.3695314	34.71673306	05/13/1998-09/29/2005
7300580	Bitter Creek West of Altus, OK	-99.3825875	34.6234033	04/25/1998-09/29/2005
7301100	Turkey Creek at Olustee, OK	-99.4367556	34.5942375	07/01/1960-09/29/1963
7301110	**Salt Fork Red River near Elmer, OK**	**-99.38231**	**34.4789647**	**10/01/1979-Current**
7301200	McClellan Ck nr McLean, TX	-100.6093011	35.32921513	10/01/1967-09/30/1980
7301300	**N Fk Red Rv nr Shamrock, TX**	**-100.2417855**	**35.26421631**	**02/19/1964-Current**
7301410	Sweetwater Ck nr Kelton, TX	-100.1209501	35.47310419	11/16/1961-Current
7301420	Sweetwater Creek near Sweetwater, OK	-99.9692772	35.4222725	04/22/1986-Current
7301481	North Fork Red River near Sayre, OK	-99.6220406	35.2847733	08/29/2015-Current
7301500	**North Fork Red River near Carter, OK**	**-99.5073128**	**35.1681083**	**10/01/1944-Current**
7302000	North Fork Red River near Granite, OK	-99.3336944	34.97339056	10/01/1903-09/29/1944
7303000	North Fork Red River blw Altus Dam nr Lugert, OK	-99.307028	34.8895047	10/01/1977-Current
7303400	**Elm Fork of North Fork Red River nr Carl, OK**	**-99.9037144**	**35.01172**	**10/01/1959-Current**

7303420	Elm Fork of North Fork Red River nr Reed, OK	-99.6948183	34.96116694	05/01/1965-09/29/1967
7303500	Elm Fork of North Fork Red River nr Mangum, OK	-99.5003669	34.9267247	05/01/1905-09/29/1976
7304500	Elk Creek near Hobart, OK	-99.1139667	34.91422667	10/01/1904-09/29/1993
7305000	**North Fork Red River near Headrick, OK**	**-99.0967464**	**34.63451556**	**04/01/1905-Current**
7305500	West Otter Creek at Snyder Lk nr Mt Park, OK	-98.9864656	34.73395556	04/01/1903-06/30/2003
7306500	Otter Creek at Mountain Park, OK	-98.984243	34.6950686	03/21/1946-09/29/1951
7307026	Ozark Canal at Altus, OK, Altus AFB IRP Ch005	-99.265085	34.65701417	04/16/1991-09/29/1991
7307028	**North Fork Red River near Tipton, OK**	**-99.2081389**	**34.5070208**	**06/28/1983-Current**
7307500	Quitaque Ck nr Quitaque, TX	-101.1179285	34.24006966	10/01/1945-09/29/1959
7307600	N Pease Rv nr Childress, TX	-100.2851145	34.27506807	05/01/1973-10/02/1979
7307750	Middle Pease Rv at Hwy 62 and 83 nr Paducah, TX	-100.3012257	34.2086816	05/01/1973-09/29/1979
7307760	Middle Pease Rv nr Paducah, TX	-100.210945	34.19118279	10/01/1979-10/05/1982
7307800	**Pease Rv nr Childress, TX**	**-100.073718**	**34.22757182**	**12/01/1959-10/01/2011**
7308000	Pease Rv nr Crowell, TX	-99.7300939	34.09591516	10/01/1924-06/30/1947
7308200	**Pease Rv nr Vernon, TX**	**-99.27813681**	**34.17925232**	**12/01/1959-Current**
7308500	**Red Rv nr Burkburnett, TX**	**-98.53172335**	**34.11009327**	**07/11/1924-Current**
7309000	East Cache Creek near Elgin, OK	-98.3669978	34.78201306	10/01/1955-09/29/1958

7310000	Little Medicine Bluff Creek nr Lawton, OK	-98.514225	34.72923556	10/01/1912-09/29/1919
7310500	Medicine Bluff Creek near Lawton, OK	-98.5003356	34.72506917	10/01/1912-09/29/1919
7311000	**East Cache Creek near Walters, OK**	**-98.2825469**	**34.3623075**	**04/01/1938-Current**
7311200	Blue Beaver Creek near Cache, OK	-98.5636711	34.6234061	07/01/1964-09/30/2003
7311500	Deep Red Creek near Randlett, OK	-98.4531103	34.22092444	10/01/1949-Current
7311600	N Wichita Rv nr Paducah, TX	-100.0648269	33.9506365	10/09/1994-09/29/2015
7311622	N Wichita Rv nr Crowell, TX	-99.94704366	33.87008366	10/01/1970-09/29/1976
7311630	Middle Wichita Rv nr Guthrie, TX	-100.0751027	33.79591742	10/01/2007-09/29/2015
7311648	Middle Wichita Rv nr Truscott, TX	-99.96259948	33.85341716	10/01/1970-09/29/1976
7311700	**N Wichita Rv nr Truscott, TX**	**-99.78648219**	**33.82064216**	**12/01/1959-Current**
7311780	S Wichita Rv nr Guthrie, TX	-100.2181622	33.62480793	10/01/1970-10/04/1976
7311790	S Wichita Rv at Ross Ranch nr Benjamin, TX	-100.0139878	33.65508741	10/01/1970-09/29/1979
7311800	**S Wichita Rv nr Benjamin, TX**	**-99.80092517**	**33.64425703**	**10/05/1986-Current**
7311900	Wichita Rv nr Seymour, TX	-99.3886905	33.70036966	12/01/1959-Current
7312100	Wichita Rv nr Mabelle, TX	-99.14284945	33.7600934	10/01/1959-Current
7312110	S Side Canal nr Dundee, TX	-98.93284409	33.81398475	10/03/2002-Current
7312130	Wichita Rv at SH 25 nr Kamay, TX	-98.83923096	33.86926267	06/01/1996-09/30/2002

7312200	Beaver Ck nr Electra, TX	-98.90506664	33.90592785	03/01/1960-Current
7312500	Wichita Rv at Wichita Falls, TX	-98.53366629	33.90954178	04/01/1938-Current
7312700	Wichita Rv nr Charlie, TX	-98.29671387	34.05315042	10/01/1967-Current
7313000	Little Beaver Creek near Duncan, OK	-98.1142067	34.49313556	10/01/1948-12/31/1963
7313600	Cow Creek at Waurika, OK	-98.0017028	34.18203667	03/01/1966-09/29/1970
7314500	Little Wichita Rv nr Archer City, TX	-98.61311245	33.66260225	07/12/1996-Current
7314900	Little Wichita Rv abv Henrietta, TX	-98.24004579	33.8267666	10/02/1996-Current
7315200	E Fk Little Wichita Rv nr Henrietta, TX	-98.08504064	33.81287833	12/01/1963-Current
7315400	Little Wichita Rv nr Ringgold, TX	-98.06837293	33.8987099	03/01/1959-09/29/1965
7315500	**Red Rv nr Terral, OK**	**-97.93447979**	**33.87871061**	**04/01/1938-Current**
7315700	Mud Creek near Courtney, OK	-97.566968	34.00426528	10/01/1960-Current
7315900	Walnut Bayou near Burneyville, OK	-97.3058503	33.94176694	10/01/1960-09/29/1971
7316000	**Red River near Gainesville, TX**	**-97.160013**	**33.7278833**	**10/01/1936-Current**
7316200	Mineral Ck nr Sadler, TX	-96.84777929	33.70232817	12/01/1967-12/31/1976
7316500	**Washita River near Cheyenne, OK**	**-99.6684386**	**35.62643694**	**10/01/1937-07/29/2018**
7317500	Sandstone Creek SWS 16A nr Cheyenne, OK	-99.6698242	35.46949389	01/01/1952-09/29/1970
7318000	Sandstone Creek SWS 16 nr Cheyenne, OK	-99.6114892	35.4806047	10/01/1952-09/29/1969

7318500	Sandstone Creek SWS 14 nr Cheyenne, OK	-99.6031556	35.4778272	10/01/1952-09/29/1970
7319000	Sandstone Creek SWS 17 nr Cheyenne, OK	-99.61149	35.5083825	10/01/1952-09/29/1970
7319500	Sandstone Creek near Berlin, OK	-99.5578772	35.50727139	10/01/1952-09/29/1972
7320000	Sandstone Creek SWS 10A nr Elk City, OK	-99.5559317	35.4667161	10/01/1951-09/29/1970
7320500	Sandstone Creek SWS 6 near Elk City, OK	-99.5031528	35.48616056	10/01/1952-09/29/1970
7321000	Sandstone Creek SWS 5 near Elk City, OK	-99.4892639	35.4917158	10/01/1952-09/29/1970
7321500	Sandstone Creek SWS 3 near Elk City, OK	-99.5114872	35.51116028	10/01/1952-09/29/1970
7322000	Sandstone Creek SWS 9 near Elk City, OK	-99.5337094	35.4944936	10/01/1951-09/29/1970
7322500	East Branch Sandstone Creek nr Elk City, OK	-99.5303769	35.52227139	04/01/1951-09/29/1972
7323000	Sandstone Creek near Cheyenne, OK	-99.5309333	35.55282667	10/01/1951-06/30/1974
7324000	Sandstone Creek SWS 1 near Cheyenne, OK	-99.5031553	35.56671556	10/01/1951-09/29/1970
7324200	**Washita River near Hammon, OK**	**-99.3062086**	**35.65643694**	**10/01/1969-Current**
7324400	**Washita River near Foss, OK**	**-99.1698128**	**35.53893778**	**03/01/1956-Current**
7324500	Barnitz Creek near Arapaho, OK	-99.0434225	35.58060444	10/01/1945-12/31/1963
7325000	**Washita River near Clinton, OK**	**-98.9670297**	**35.5308833**	**10/01/1935-Current**
7325500	Washita River at Carnegie, OK	-98.5639503	35.1172811	10/01/1937-09/29/2006
7325800	Cobb Creek near Eakly, OK	-98.5942308	35.2906111	10/01/1968-Current

7325850	Lake Creek near Eakly, OK	-98.5292286	35.29088917	10/01/1969-Current
7325860	Willow Creek near Albert, OK	-98.4661703	35.23339056	10/01/1970-Current
7326000	Cobb Creek near Fort Cobb, OK	-98.4428358	35.14367028	10/01/1939-Current
7326500	**Washita River at Anadarko, OK**	**-98.243385**	**35.0842283**	**01/01/1903-Current**
7327000	Sugar Creek near Gracemont, OK	-98.2558861	35.17506	10/01/1955-09/29/1974
7327050	Spring Creek near Gracemont, OK	-98.1764397	35.18922694	06/25/1991-09/30/1994
7327442	Little Washita River near Cyril, OK	-98.2331053	34.89256639	10/01/1992-Current
7327445	Chetonia Creek Tributary blw Cyril, OK	-98.1922706	34.88312278	10/01/1989-07/31/1991
7327447	Little Washita River near Cement, OK	-98.1244903	34.83784667	02/01/1992-Current
7327483	Boggy Creek near Ninnekah, OK	-97.9953203	34.88395778	04/01/1996-09/29/2004
7327490	Little Washita River near Ninnekah, OK	-97.9525425	34.9447897	10/01/1963-12/31/1985
7327500	Little Washita River at Ninnekah, OK	-97.9264311	34.95673389	10/01/1951-09/29/1963
7327550	Little Washita River East of Ninnekah, OK	-97.8994861	34.9634008	02/01/1992-Current
7328000	Washita River near Tabler, OK	-97.8728192	34.97173417	10/01/1939-09/29/1952
7328070	Winter Creek near Alex, OK	-97.7614283	34.9931236	10/01/1964-05/14/1987
7328100	Washita River at Alex, OK	-97.7739272	34.92590306	10/01/1964-Current
7328180	North Criner Creek near Criner, OK	-97.5847556	34.97145917	10/01/1989-Current

7328500	**Washita River near Pauls Valley, OK**	**-97.2514125**	**34.7548025**	**10/01/1937-Current**
7328550	Washington Creek near Pauls Valley, OK	-97.2022458	34.8259122	06/27/1991-03/31/1994
7329000	Rush Creek at Purdy, OK	-97.5989189	34.6961886	10/01/1939-09/30/1994
7329500	Rush Creek near Maysville, OK	-97.4053039	34.74341194	10/01/1954-09/29/1976
7329700	Wildhorse Creek near Hoover, OK	-97.2472436	34.54147389	10/01/1969-06/30/2002
7329852	Rock Creek at Sulphur, OK	-96.988628	34.49536694	10/01/1989-Current
7329900	Rock Creek at Dougherty, OK	-97.036405	34.39731278	10/01/1956-06/30/1967
7330500	Caddo Creek near Ardmore, OK	-97.1080697	34.24259278	10/01/1936-12/31/1997
7330700	Caddo Creek Site 7cmp near Gene Autry, OK	-97.0516797	34.2403711	03/28/1996-09/29/1998
7331000	**Washita River near Dickson, OK**	**-96.9758447**	**34.2334272**	**10/01/1928-Current**
7331250	Mill Creek near Ravia, OK	-96.8105619	34.25981639	10/01/1968-09/29/1971
7331300	Pennington Creek near Reagan, OK	-96.70806009	34.34759314	10/01/2003-Current
7332390	Blue River near Connerville, OK	-96.6005578	34.3834261	10/01/1976-Current
7332400	Blue River at Milburn, OK	-96.5488878	34.25064917	10/01/1965-06/30/1987
7332600	Bois D Arc Ck nr Randolph, TX	-96.21469996	33.47566345	12/02/1962-09/29/1985
7342465	S Sulphur at Commerce, TX	-95.91413282	33.21177979	10/01/1991-Current
7342470	S Sulphur Rv nr Commerce, TX	-95.86274213	33.21983497	10/01/1979-09/29/1991

7342480	Middle Sulphur Rv at Commerce, TX	-95.91552173	33.26650079	10/02/1991-Current
7342500	S Sulphur Rv nr Cooper, TX	-95.59495624	33.35649902	10/01/2001-Current
7343000	N Sulphur Rv nr Cooper, TX	-95.58773472	33.47483009	10/01/1999-Current
8042700	North Ck nr Jacksboro, TX	-98.2983819	33.28261252	08/01/1956-10/20/1980
8042800	W Fk Trinity Rv nr Jacksboro, TX	-98.08059791	33.29177926	10/01/2001-Current
8042900	Beans Ck at Wizard Wells, TX	-97.96726152	33.1998374	06/10/1992-09/30/1995
8043500	W Fk Trinity Rv at Bridgeport, TX	-97.75614407	33.20150463	10/01/1924-02/28/1930
8043950	Big Sandy Ck nr Chico, TX	-97.67864243	33.27428117	10/01/1995-08/31/2004
8044000	Big Sandy Ck nr Bridgeport, TX	-97.6947538	33.23178218	10/01/1936-Current
8044135	Garrett Ck nr Paradise, TX	-97.65502817	33.10511968	06/09/1992-09/30/1995
8044140	Salt Ck nr Paradise, TX	-97.6500279	33.09845327	08/03/1992-09/30/1995
8044500	W Fk Trinity Rv nr Boyd, TX	-97.55863596	33.08539917	10/01/2001-Current
8044800	Walnut Ck at Reno, TX	-97.58307976	32.94568035	10/01/1995-Current
8045500	W Fk Trinity Rv at Lk Worth Dam abv Ft Worth, TX	-97.41529696	32.79096222	10/01/1924-09/29/1934
8045850	Clear Fk Trinity Rv nr Weatherford, TX	-97.65197113	32.74040699	10/19/1987-Current
8046000	Clear Fk Trinity Rv nr Aledo, TX	-97.56446798	32.64124423	08/01/1947-10/09/1975
8047000	Clear Fk Trinity Rv nr Benbrook, TX	-97.44196423	32.66513275	11/02/1987-Current

8047050	Marys Ck at Benbrook, TX	-97.44724221	32.69513161	05/24/1998-Current
8047500	Clear Fk Trinity Rv at Ft Worth, TX	-97.35890627	32.732353	10/08/1987-Current
8048000	W Fk Trinity Rv at Ft Worth, TX	-97.33251667	32.76096337	10/01/2001-Current
8048520	Sycamore Ck at IH 35W, Ft Worth, TX	-97.321405	32.66541113	10/01/1969-09/29/1976
8048543	W Fk Trinity Rv at Beach St, Ft Worth, TX	-97.28945979	32.75179721	10/01/2001-Current
8048800	Big Fossil Ck at Haltom City, TX	-97.24862558	32.80735146	02/03/2015-Current
8048850	Little Fossil Ck at Mesquite St, Ft Worth, TX	-97.29140459	32.80929568	10/01/1968-09/29/1976
8048970	Village Ck at Everman, TX	-97.26501422	32.60346909	10/01/1991-Current
8048980	Village Ck at Kennedale, TX	-97.24223584	32.64124563	07/11/1986-09/29/1989
8049000	Village Ck nr Handley, TX	-97.21695748	32.70013253	06/01/1925-02/28/1930
8049500	W Fk Trinity Rv at Grand Prairie, TX	-97.02973015	32.7987406	10/14/1987-Current
8049550	Big Bear Ck nr Grapevine, TX	-97.12917803	32.91345932	12/01/1966-10/01/1979
8049553	Big Bear Ck at Euless/Grapevine Rd nr Grapevine,TX	-97.08222222	32.89472222	10/22/2002-05/14/2004
8049569	Big Bear Ck at SH 183 nr Euless, TX	-97.03583333	32.83555556	10/21/2002-05/17/2004
8049580	Mountain Ck nr Venus, TX	-97.12306461	32.49097237	10/01/1985-Current
8049600	Mountain Ck nr Cedar Hill, TX	-97.02333972	32.58430251	10/01/1960-09/29/1984
8049700	Walnut Ck nr Mansfield, TX	-97.10195335	32.5809695	10/01/2000-Current

8050000	Mountain Ck nr Grand Prairie, TX	-96.96695005	32.7056878	10/01/1924-06/30/1933
8050100	Mountain Ck at Grand Prairie, TX	-96.92583818	32.74763085	10/01/2001-Current
8050300	Elm Fk Trinity Rv nr Muenster, TX	-97.38279826	33.61010775	10/01/1956-10/04/1973
8050400	Elm Fk Trinity Rv at Gainesville, TX	-97.15640159	33.62427479	10/01/2001-Current
8050500	Elm Fk Trinity Rv nr Sanger, TX	-97.08501045	33.38650347	05/01/1949-12/05/1984
8050800	Timber Ck nr Collinsville, TX	-96.94722668	33.55455428	10/07/1987-Current
8050840	Range Ck nr Collinsville, TX	-96.80721914	33.52622016	10/21/1992-Current
8051000	Isle Du Bois Ck nr Pilot Point, TX	-97.01278544	33.40650311	05/01/1949-12/31/1984
8051130	Elm Fk Trinity Rv nr Pilot Point, TX	-97.04723141	33.35039343	10/01/1985-06/03/1993
8051500	Clear Ck nr Sanger, TX	-97.1794587	33.3362269	10/01/2001-Current
8052000	Elm Fk Trinity Rv nr Denton, TX	-97.04528731	33.25067392	10/01/1924-11/30/1926
8052650	Little Elm Ck nr Celina, TX	-96.82388762	33.36539123	02/21/1966-09/29/1976
8052700	Little Elm Ck nr Aubrey, TX	-96.8927805	33.28344977	10/02/1995-Current
8052780	Hickory Ck at Denton, TX	-97.14195727	33.15178696	07/03/2008-Current
8053000	Elm Fk Trinity Rv nr Lewisville, TX	-96.96111727	33.04567733	01/28/2004-Current
8053010	Indian Ck at Hebron Pkwy at Carrollton, TX	-96.92444935	33.01845543	10/01/1986-09/15/1989
8053030	Furneaux Ck at Josey Lane at Carrollton, TX	-96.88639255	33.00151114	10/04/1986-09/13/1989

8053090	Hutton Branch at Broadway, Carrollton, TX	-96.91056003	32.9567905	12/07/1986-08/19/2018
8053500	Denton Ck nr Justin, TX	-97.29057323	33.11901003	01/17/2004-Current
8054000	Denton Ck nr Roanoke, TX	-97.20501445	33.0401228	10/01/1924-09/29/1955
8055000	Denton Ck nr Grapevine, TX	-97.01278566	32.98706771	07/05/2003-Current
8055500	Elm Fk Trinity Rv nr Carrollton, TX	-96.94445004	32.96595701	01/16/1988-Current
8055700	Bachman Branch at Dallas, TX	-96.85389163	32.86040457	10/01/1963-10/04/1979
8056500	Turtle Ck at Dallas, TX	-96.80250131	32.80735055	06/12/2010-Current
8057000	Trinity Rv at Dallas, TX	-96.82194639	32.77485172	10/01/2001-Current
8057100	White Rock Ck at Keller Spgs Rd, Dallas, TX	-96.80555658	32.9704007	08/01/1961-09/29/1979
8057200	White Rock Ck at Greenville Ave, Dallas, TX	-96.75666636	32.88929207	10/01/2001-Current
8057300	White Rock Ck at White Rock Lk, Dallas, TX	-96.72583196	32.80873904	10/01/1962-Current
8057410	Trinity Rv bl Dallas, TX	-96.73583187	32.70763139	10/01/1988-Current
8057445	Prairie Ck at US Hwy 175, Dallas, TX	-96.66999551	32.70485346	10/02/2001-09/30/2011
8057448	Trinity Rv nr Wilmer, TX	-96.62221481	32.61763421	10/01/1998-11/14/2002
8057450	Tenmile Ck at SH 342 at Lancaster, TX	-96.75610804	32.57846916	10/01/1969-10/03/1979
8058500	Honey Ck nr McKinney, TX	-96.65777032	33.27844798	08/01/1951-09/29/1973
8058900	E Fk Trinity Rv at McKinney, TX	-96.60887971	33.24400417	12/01/1989-09/29/2010

8059000	E Fk Trinity Rv nr McKinney, TX	-96.595824	33.20372724	12/17/2009-Current
8059400	Sister Grove Ck nr Blue Ridge, TX	-96.48304128	33.29455786	03/04/2008-Current
8059500	Sister Grove Ck nr Princeton, TX	-96.4758193	33.1931713	09/01/1949-01/31/1975
8060000	E Fk Trinity Rv abv Pilot Grove nr Lavon, TX	-96.47582072	33.02317512	03/01/1949-09/29/1953
8061000	E Fk Trinity Rv nr Lavon, TX	-96.47554292	33.02373066	10/01/1953-Current
8061500	E Fk Trinity Rv nr Rockwall, TX	-96.5058219	32.92373399	10/01/1924-09/29/1954
8061540	Rowlett Ck nr Sachse, TX	-96.61443781	32.95984447	10/01/1994-Current
8061700	Duck Ck nr Garland, TX	-96.59554804	32.83290425	01/01/1958-12/31/1992
8061750	E Fk Trinity Rv nr Forney, TX	-96.50359913	32.77429472	10/01/2003-Current
8061950	S Mesquite Ck at Mercury Rd, N Mesquite, TX	-96.57026887	32.72568565	10/01/1968-09/29/1979
8062000	E Fk Trinity Rv nr Crandall, TX	-96.48526511	32.63874423	10/18/1987-Current
8062900	Kings Ck nr Kaufman, TX	-96.3291479	32.5134711	01/02/1963-09/29/1987
8063100	Richland Ck nr Dawson, TX	-96.68137865	31.93849082	10/01/2003-Current
8063200	Pin Oak Ck nr Hubbard, TX	-96.71748992	31.80043987	09/01/1956-09/29/1972
8063500	Richland Ck nr Richland, TX	-96.42136996	31.95071375	04/01/1939-06/30/1989
8063800	Waxahachie Ck nr Bardwell, TX	-96.64026759	32.24348072	10/01/1996-Current
8064100	Chambers Ck nr Rice, TX	-96.52026387	32.19848231	10/01/1999-Current

8064500	Chambers Ck nr Corsicana, TX	-96.37081354	32.10820817	04/01/1939-09/29/1984
8067500	Cedar Bayou nr Crosby, TX	-94.98576019	29.97271914	10/01/1971-Current
8067525	Goose Ck at Baytown, TX	-94.99965032	29.77078197	10/01/2006-Current
8067700	Caney Ck nr Dobbin, TX	-95.80994977	30.35381805	04/01/1963-09/29/1965
8068720	Cypress Ck at Katy-Hockley Rd nr Hockley, TX	-95.80828347	29.95022371	06/01/1975-Current
8068740	Cypress Ck at House-Hahl Rd nr Cypress, TX	-95.71772487	29.959112	06/01/1975-Current
8068780	Little Cypress Ck nr Cypress, TX	-95.6974463	30.01605437	05/01/1982-Current
8068800	Cypress Ck at Grant Rd nr Cypress, TX	-95.59855448	29.97355566	10/01/1982-Current
8068900	Cypress Ck at Stuebner-Airline Rd nr Westfield, TX	-95.51188496	30.00660994	10/01/1987-09/30/1989
8068900	Cypress Ck at Stuebner-Airline Rd nr Westfield, TX	-95.51188496	30.00660994	09/12/2015-02/22/2018
8069000	Cypress Ck nr Westfield, TX	-95.42882703	30.03577532	07/01/1944-Current
8072300	Buffalo Bayou nr Katy, TX	-95.80689511	29.74328664	07/01/1977-Current
8072700	S Mayde Ck nr Addicks, TX	-95.69244708	29.80106159	07/08/1987-07/07/2007
8072730	Bear Ck nr Barker, TX	-95.68689122	29.83078281	07/01/1977-Current
8072760	Langham Ck at W Little York Rd nr Addicks, TX	-95.64661209	29.86717035	07/01/1977-Current
8073500	Buffalo Bayou nr Addicks, TX	-95.60577819	29.76189581	09/01/1945-Current
8073600	Buffalo Bayou at W Belt Dr at Houston, TX	-95.5577213	29.76217336	09/01/1971-Current

8073700	Buffalo Bayou at Piney Point, TX	-95.52355378	29.7468959	10/01/1963-Current
8074000	Buffalo Bayou at Houston, TX	-95.40855048	29.76022829	06/01/1936-07/14/2018
8074020	Whiteoak Bayou at Alabonson Rd at Houston, TX	-95.4804961	29.87078073	10/01/1999-Current
8074150	Cole Ck at Deihl Rd, Houston, TX	-95.48799646	29.85133693	04/14/1964-02/11/2004
8074150	Cole Ck at Deihl Rd, Houston, TX	-95.48799646	29.85133693	10/25/2015-07/06/2018
8074250	Brickhouse Gully at Costa Rica St, Houston, TX	-95.469385	29.82800424	08/14/1964-02/11/2004
8074250	Brickhouse Gully at Costa Rica St, Houston, TX	-95.469385	29.82800424	09/03/2015-Current
8074500	Whiteoak Bayou at Houston, TX	-95.39716115	29.77522777	05/25/1936-Current
8074540	Little Whiteoak Bayou at Trimble St, Houston, TX	-95.36805556	29.79277778	01/01/2017-09/30/2017
8074760	Brays Bayou at Alief, TX	-95.58716682	29.71106411	10/01/2006-Current
8074800	Keegans Bayou at Roark Rd nr Houston, TX	-95.56216641	29.65662136	08/13/1964-02/11/2004
8074800	Keegans Bayou at Roark Rd nr Houston, TX	-95.56216641	29.65662136	09/11/2015-03/29/2018
8074810	Brays Bayou at Gessner Dr, Houston, TX	-95.52827652	29.6727317	10/02/2001-Current
8075000	Brays Bayou at Houston, TX	-95.41216197	29.69717469	05/25/1936-Current
8075400	Sims Bayou at Hiram Clarke St, Houston, TX	-95.44605219	29.61884399	08/18/1964-Current
8075500	Sims Bayou at Houston, TX	-95.28938068	29.67439687	10/01/1952-09/01/2001
8075500	Sims Bayou at Houston, TX	-95.28938068	29.67439687	10/25/2015-08/30/2017

8075730	Vince Bayou at Pasadena, TX	-95.216323	29.69467363	10/01/1971-Current
8075770	Hunting Bayou at IH 610, Houston, TX	-95.2679907	29.79328217	04/14/1964-Current
8075780	Greens Bayou at Cutten Rd nr Houston, TX	-95.51966334	29.94911178	09/03/2015-07/08/2018
8075900	Greens Bayou nr US Hwy 75 nr Houston, TX	-95.41799362	29.95688886	08/03/1965-Current
8076000	Greens Bayou nr Houston, TX	-95.3068796	29.91827842	10/01/1952-Current
8076500	Halls Bayou at Houston, TX	-95.33493645	29.86189143	10/01/1952-Current
8076700	Greens Bayou at Ley Rd, Houston, TX	-95.23326715	29.83716952	12/02/1971-05/12/2018
8077000	Clear Ck nr Pearland, TX	-95.28660292	29.59745458	08/01/1944-09/04/1994
8077600	Clear Ck nr Friendswood, TX	-95.17854405	29.51745517	05/04/1987-02/11/2004
8078000	Chocolate Bayou nr Alvin, TX	-95.32076975	29.36940462	03/01/1959-Current
8079000	Oyster Ck nr Angleton, TX	-95.47577532	29.15857735	10/01/1944-09/29/1980
8079500	N Fk DMF Brazos Rv at Lubbock, TX	-101.8282217	33.58564081	10/01/1939-09/24/1949
8079575	N Fk DMF Brazos Rv nr Post, TX	-101.3384715	33.24870448	10/02/1983-09/30/1993
8079600	**DMF Brazos Rv at Justiceburg, TX**	**-101.1976302**	**33.03843236**	**12/01/1961-Current**
8080000	DMF Brazos Rv nr Rotan, TX	-100.4881658	32.93038043	12/21/1949-09/29/1951
8080500	**DMF Brazos Rv nr Aspermont, TX**	**-100.1806589**	**33.00815771**	**09/09/1995-Current**
8080540	McDonald Ck nr Post, TX	-101.227084	33.35092391	10/01/1965-09/29/1978

8080700	Running Water Draw at Plainview, TX	-101.7026748	34.17896048	10/02/2002-Current
8080950	Duck Ck nr Girard, TX	-100.7051211	33.35620109	10/01/1964-09/29/1989
8081000	Salt Fk Brazos Rv nr Peacock, TX	-100.431778	33.21203853	01/02/1950-09/29/1986
8081200	Croton Ck nr Jayton, TX	-100.4315008	33.28842503	09/02/1959-09/29/1986
8081500	Salt Croton Ck nr Aspermont, TX	-100.4084454	33.40092169	10/01/1956-10/05/1977
8082000	Salt Fk Brazos Rv nr Aspermont, TX	-100.2381622	33.33398012	10/02/2000-Current
8082100	Stinking Ck nr Aspermont, TX	-100.213439	33.23342759	10/01/1965-10/06/1983
8082180	N Croton Ck nr Knox City, TX	-100.0812116	33.38314796	09/26/1965-09/29/1986
8082500	**Brazos Rv at Seymour, TX**	**-99.26757557**	**33.58092766**	**12/01/1923-Current**
8082700	Millers Ck nr Munday, TX	-99.46508251	33.32926607	08/01/1963-Current
8083000	Brazos Rv nr Graham, TX	-98.72700615	33.08205929	12/01/1915-02/29/1920
8083100	Clear Fk Brazos Rv nr Roby, TX	-100.3887183	32.78760777	01/01/1962-Current
8083230	Clear Fk Brazos Rv nr Noodle, TX	-100.0725999	32.67455998	10/01/2001-Current
8083240	Clear Fk Brazos Rv at Hawley, TX	-99.81509185	32.59817587	10/01/1967-Current
8083245	Mulberry Ck nr Hawley, TX	-99.79259093	32.56789913	12/11/1967-09/29/1989
8083300	Elm Ck nr Abilene, TX	-99.80786725	32.35235006	10/01/1963-10/02/1979
8083400	Little Elm Ck nr Abilene, TX	-99.85259106	32.39151513	10/01/1963-09/29/1979

8083420	Cat Claw Ck at Abilene, TX	-99.74925564	32.47540219	10/01/1970-Current
8083430	Elm Ck at Abilene, TX	-99.74120017	32.50817902	10/01/1979-09/29/1983
8083470	Cedar Ck at Abilene, TX	-99.72064348	32.44901423	10/01/1970-10/09/1984
8083480	Cedar Ck at IH 20, Abilene, TX	-99.71619952	32.49956829	06/01/2001-Current
8084000	**Clear Fk Brazos Rv at Nugent, TX**	**-99.66953258**	**32.69011862**	**03/01/1924-Current**
8084800	California Ck nr Stamford, TX	-99.64258805	32.93094435	10/01/1962-Current
8085000	Paint Ck nr Haskell, TX	-99.54369615	33.07760645	01/01/1950-09/29/1951
8085500	**Clear Fk Brazos Rv at Ft Griffin, TX**	**-99.22452145**	**32.93455573**	**02/01/1924-Current**
8086000	Clear Fk Brazos Rv at Crystal Falls, TX	-98.83367648	32.90011812	12/01/1921-02/28/1929
8086015	Hubbard Ck nr Sedwick, TX	-99.23924279	32.60178915	10/01/1963-09/29/1966
8086050	Deep Ck at Moran, TX	-99.17007397	32.55929085	08/10/2002-Current
8086100	Hubbard Ck nr Albany, TX	-99.16479694	32.68928712	02/01/1962-10/09/1975
8086120	Salt Prong Hubbard Ck at US Hwy 380 nr Albany, TX	-99.26841077	32.68373104	10/01/1963-09/29/1968
8086150	N Fk Hubbard Ck nr Albany, TX	-99.27507775	32.7076192	11/02/1962-09/29/1990
8086200	Salt Prong Hubbard Ck nr Albany, TX	-99.21202053	32.70067531	02/01/1962-09/24/1963
8086210	Snailum Ck nr Albany, TX	-99.175353	32.72095279	10/01/1963-09/29/1966
8086212	Hubbard Ck bl Albany, TX	-99.14062992	32.73289717	10/01/1966-Current

8086235	Battle Ck nr Moran, TX	-99.10923897	32.55290253	10/01/1966-09/29/1968
8086260	Pecan Ck nr Eolian, TX	-99.03284824	32.58373549	10/01/1966-10/09/1975
8086290	Big Sandy Ck abv Breckenridge, TX	-99.00451456	32.64845613	02/01/1962-Current
8086500	Hubbard Ck nr Breckenridge, TX	-98.94812459	32.83706252	05/02/1955-09/29/1986
8087300	Clear Fk Brazos Rv at Eliasville, TX	-98.76672978	32.96011772	12/01/1915-10/04/1982
8088000	**Brazos Rv nr South Bend, TX**	**-98.64394811**	**33.02428377**	**10/01/1938-Current**
8088100	Salt Ck at Olney, TX	-98.74478368	33.37038526	05/01/1958-10/05/1977
8088200	Salt Ck nr Newcastle, TX	-98.64894817	33.21677844	05/01/1958-09/29/1960
8088300	Briar Ck nr Graham, TX	-98.6186695	33.21205664	05/02/1958-09/29/1989
8088450	Big Cedar Ck nr Ivan, TX	-98.72395091	32.82762161	12/02/1964-09/29/1989
8088600	Brazos Rv at Morris Sheppard Dam nr Graford, TX	-98.42588607	32.87206697	10/01/1976-09/29/1995
8088610	Brazos Rv nr Graford, TX	-98.411719	32.85817854	10/01/2002-Current
8089000	**Brazos Rv nr Palo Pinto, TX**	**-98.30254916**	**32.86262361**	**06/09/1987-Current**
8090500	Palo Pinto Ck nr Santo, TX	-98.18087814	32.63096402	10/01/1924-09/29/1976
8090800	Brazos Rv nr Dennis, TX	-97.92587026	32.61568907	10/01/1999-Current
8091000	Brazos Rv nr Glen Rose, TX	-97.70252676	32.25903188	10/01/2003-Current
8091500	Paluxy Rv at Glen Rose, TX	-97.77725121	32.23153166	10/01/2001-Current

8091750	Squaw Ck nr Glen Rose, TX	-97.73252777	32.2701423	10/01/1994-09/30/2006
8092000	Nolan Rv at Blum, TX	-97.40279404	32.15070534	08/01/1924-Current
8092600	Brazos Rv at Whitney Dam nr Whitney, TX	-97.36695761	31.86682458	10/01/1987-09/29/1991
8093100	**Brazos Rv nr Aquilla, TX**	**-97.29778823**	**31.8123822**	**10/09/1987-Current**
8093250	Hackberry Ck at Hillsboro, TX	-97.15000688	32.00571108	10/01/1979-09/30/1992
8093360	Aquilla Ck abv Aquilla, TX	-97.20306337	31.8954362	04/23/1987-Current
8093400	Cobb Ck nr Abbott, TX	-97.09944852	31.91987996	12/01/1966-09/29/1979
8093500	Aquilla Ck nr Aquilla, TX	-97.20139613	31.84460414	01/01/1939-05/24/2001
8093700	N Bosque Rv at Stephenville, TX	-98.19892983	32.21569591	03/01/1958-09/29/1979
8094800	N Bosque Rv at Hico, TX	-98.03475656	31.97820321	04/15/2014-Current
8095000	N Bosque Rv nr Clifton, TX	-97.56807481	31.785991	10/01/2007-Current
8095200	N Bosque Rv at Valley Mills, TX	-97.46946018	31.66960575	08/19/1995-Current
8095300	Middle Bosque Rv nr McGregor, TX	-97.36584508	31.50933179	10/01/2007-Current
8095400	Hog Ck nr Crawford, TX	-97.35640042	31.55571992	10/02/2007-Current
8095500	S Bosque Rv nr Speegleville, TX	-97.25028556	31.51683253	04/01/1924-04/30/1930
8095600	Bosque Rv nr Waco, TX	-97.19361646	31.60127597	10/01/1959-06/23/1982
8096500	Brazos Rv at Waco, TX	-97.07333254	31.53600056	10/01/2007-Current

8097500	Brazos Rv nr Marlin, TX	-96.969713	31.28850993	10/01/1938-09/29/1951
8098000	Deer Ck at Chilton, TX	-97.0586055	31.26628755	04/01/1934-09/29/1936
8098290	Brazos Rv nr Highbank, TX	-96.82498262	31.13407098	10/01/2002-Current
8098300	Little Pond Ck nr Burlington, TX	-96.98832139	31.02657314	08/23/2002-Current
8099100	Leon Rv nr De Leon, TX	-98.53310583	32.17375022	10/03/2007-Current
8099300	Sabana Rv nr De Leon, TX	-98.60560671	32.11402957	09/01/1960-Current
8099500	Leon Rv nr Hasse, TX	-98.4592116	31.95792351	01/02/1939-Current
8100000	Leon Rv nr Hamilton, TX	-98.12142376	31.78876325	10/02/2007-Current
8100500	Leon Rv at Gatesville, TX	-97.76196701	31.43294223	10/01/2007-Current
8101000	Cowhouse Ck at Pidcoke, TX	-97.88502445	31.28489022	10/01/2007-Current
8101500	Cowhouse Ck nr Killeen, TX	-97.71557458	31.20600559	09/25/1924-07/31/1942
8102500	Leon Rv nr Belton, TX	-97.44139674	31.07017947	10/04/2007-Current
8102600	Nolan Ck at Belton, TX	-97.45723041	31.05184675	01/31/1974-11/03/1982
8103800	Lampasas Rv nr Kempner, TX	-98.01669192	31.08184021	10/01/1962-Current
8103900	S Fk Rocky Ck nr Briggs, TX	-98.0369676	30.91156805	12/30/1989-Current
8104000	Lampasas Rv at Youngsport, TX	-97.70862624	30.95740298	03/01/1924-09/29/1980
8104100	Lampasas Rv nr Belton, TX	-97.49250876	31.0018485	02/01/1963-Current

8104310	Salado Ck bl Salado Spgs at Salado, TX	-97.52417571	30.95212782	10/01/1984-04/02/1987
8104500	Little Rv nr Little River, TX	-97.34611308	30.96657232	10/02/2007-Current
8104700	N Fk San Gabriel Rv nr Georgetown, TX	-97.71139933	30.66185923	10/01/2001-Current
8104900	S Fk San Gabriel Rv at Georgetown, TX	-97.69112037	30.6257499	10/01/2007-Current
8105000	San Gabriel Rv at Georgetown, TX	-97.65528632	30.65408242	03/01/1924-04/01/1987
8105095	Berry Ck at Airport Rd nr Georgetown, TX	-97.66639851	30.70324685	10/01/2005-Current
8105100	Berry Ck nr Georgetown, TX	-97.65612023	30.69130304	07/20/1967-09/29/2003
8105200	Berry Ck at SH 971 nr Georgetown, TX	-97.61445201	30.67602639	10/01/1984-04/01/1987
8105400	San Gabriel Rv nr Circleville, TX	-97.47333565	30.62880763	02/01/1924-11/30/1976
8105700	San Gabriel Rv at Laneport, TX	-97.27888488	30.69436146	10/01/2007-Current
8106300	Brushy Ck nr Rockdale, TX	-97.07859965	30.69408463	08/01/1967-10/01/1980
8106310	San Gabriel Rv nr Rockdale, TX	-97.03887613	30.72769422	09/17/1980-10/29/1992
8106350	Little Rv nr Rockdale, TX	-97.01387539	30.76074836	02/12/1981-Current
8106500	Little Rv at Cameron, TX	-96.94665119	30.83519047	04/16/2000-Current
8107000	Big Elm Ck nr Temple, TX	-97.23583201	31.04962529	04/01/1934-06/30/1936
8107500	Big Elm Ck nr Buckholts, TX	-97.10415855	30.94740794	04/01/1934-09/29/1936
8108000	N Elm Ck nr Ben Arnold, TX	-97.05026755	30.95018612	10/01/1934-09/29/1936

8108200	N Elm Ck nr Cameron, TX	-97.02054396	30.93129814	10/01/1962-09/24/1973
8110200	Brazos Rv at Washington, TX	-96.15523665	30.36131908	11/01/1965-03/15/1987
8110325	Navasota Rv abv Groesbeck, TX	-96.52081399	31.57433657	11/06/2007-Current
8113500	Richmond Irrigation Co S Canal nr Richmond, TX	-95.78356164	29.56690419	10/01/1931-10/03/1978
8114000	Brazos Rv at Richmond, TX	-95.75772753	29.5824589	10/02/1999-Current
8114500	Brazos Rv nr Juliff, TX	-95.53299956	29.45551617	06/01/1949-09/29/1969
8115000	Big Ck nr Needville, TX	-95.81272922	29.47662987	06/01/1947-Current
8115500	Fairchild Ck nr Needville, TX	-95.76161663	29.44607466	06/01/1947-10/31/1954
8116000	Big Ck nr Guy, TX	-95.71022626	29.4127416	07/01/1947-06/30/1950
8116400	Dry Ck nr Rosenberg, TX	-95.74689417	29.51190593	12/13/2007-Current
8116500	Dry Ck nr Richmond, TX	-95.7116155	29.50551692	06/01/1947-06/30/1950
8116650	**Brazos Rv nr Rosharon, TX**	**-95.58244482**	**29.34968582**	**04/01/1967-Current**
8117290	Brazos River at Freeport, TX	-95.38716325	28.92136042	02/14/2002-09/05/2002
8117500	San Bernard Rv nr Boling, TX	-95.8938421	29.31357967	10/01/2008-Current
8117900	Big Boggy Ck nr Wadsworth, TX	-95.95079382	28.80748426	06/18/1970-10/11/1977
8117995	Colorado Rv nr Gail, TX	-101.2854044	32.62872081	03/01/1988-Current
8118500	Bull Ck nr Ira, TX	-101.0942879	32.60011109	10/01/1947-09/29/1962

8119000	Bluff Ck nr Ira, TX	-101.0509533	32.59150035	10/01/1947-09/29/1965
8119500	Colorado Rv nr Ira, TX	-101.0537304	32.53844624	10/02/1947-09/29/1989
8120500	Deep Ck nr Dunn, TX	-100.9078941	32.57372368	07/25/2001-Current
8120700	**Colorado Rv nr Cuthbert, TX**	**-100.9498382**	**32.47733745**	**03/01/1965-10/06/2002**
8121000	**Colorado Rv at Colorado City, TX**	**-100.8787246**	**32.39261865**	**10/01/1997-Current**
8121500	Morgan Ck nr Westbrook, TX	-101.02595	32.39511765	06/01/1954-09/29/1963
8122500	Morgan Ck nr Colorado City, TX	-100.9501149	32.38817389	05/01/1947-04/30/1949
8123500	Champion Ck nr Colorado City, TX	-100.8248337	32.31706613	10/01/1947-09/29/1959
8123650	Beals Ck abv Big Spring, TX	-101.4909581	32.25039797	01/01/1959-09/29/1979
8123700	Beals Ck at Big Spring, TX	-101.4420685	32.26262005	03/01/1957-12/31/1958
8123720	Beals Ck nr Coahoma, TX	-101.3620666	32.24900973	07/01/1983-02/29/1988
8123800	Beals Ck nr Westbrook, TX	-101.0140029	32.19929132	10/01/1958-Current
8123850	**Colorado Rv abv Silver, TX**	**-100.762052**	**32.05374399**	**08/29/1967-Current**
8123900	Colorado Rv nr Silver, TX	-100.7359399	32.01957904	10/01/1956-09/29/1970
8124000	**Colorado Rv at Robert Lee, TX**	**-100.4806543**	**31.88542168**	**11/01/1923-Current**
8126380	**Colorado Rv nr Ballinger, TX**	**-100.0264755**	**31.71542973**	**06/01/1907-Current**
8126500	Colorado Rv at Ballinger, TX	-99.95397442	31.73292864	06/01/1907-10/01/1979

8127000	**Elm Ck at Ballinger, TX**	**-99.94786362**	**31.74931697**	**04/01/1932-Current**
8127500	S Concho Irrigation Co Canal at Christoval, TX	-100.5000943	31.18822294	12/01/1939-10/17/1983
8128000	S Concho Rv at Christoval, TX	-100.5020388	31.18711186	03/01/1930-Current
8128400	Middle Concho Rv abv Tankersley, TX	-100.7112125	31.42738264	04/01/1961-Current
8128500	Middle Concho Rv nr Tankersley, TX	-100.614265	31.37655145	03/01/1930-03/31/1961
8129300	Spring Ck abv Tankersley, TX	-100.6403767	31.33016396	10/01/1960-09/30/1995
8130500	Dove Ck at Knickerbocker, TX	-100.6309318	31.27405465	10/01/1960-05/27/2014
8130700	Spring Ck abv Twin Buttes Res nr San Angelo, TX	-100.600931	31.33099733	10/01/2001-Current
8131000	Spring Ck nr Tankersley, TX	-100.5350958	31.35849668	03/01/1930-09/29/1960
8131190	S Concho Rv abv Gardner Dam nr San Angelo, TX	-100.5078722	31.2829432	09/26/2001-06/27/2004
8131400	Pecan Ck nr San Angelo, TX	-100.4459254	31.30905356	07/01/1961-Current
8132500	S Concho Rv at San Angelo, TX	-100.4253701	31.44599499	10/01/1931-09/29/1953
8133250	N Concho Rv abv Sterling City, TX	-101.1051131	31.89735742	02/17/2000-Current
8133500	N Concho Rv at Sterling City, TX	-100.9937215	31.83013918	09/01/1939-10/23/2015
8133900	Chalk Ck nr Water Valley, TX	-100.6906582	31.64654097	10/01/2001-09/29/2011
8134000	**N Concho Rv nr Carlsbad, TX**	**-100.6370449**	**31.5926549**	**04/01/1924-Current**
8134230	Grape Ck nr Grape Creek, TX	-100.5856544	31.57515616	10/01/2001-10/23/2011

8134250	North Concho Rv nr Grape Creek, TX	-100.5550977	31.54265769	02/14/2000-Current
8135000	N Concho Rv at San Angelo, TX	-100.4478712	31.46599438	10/28/1915-12/31/1990
8136000	Concho Rv at San Angelo, TX	-100.4106474	31.45460602	10/01/1915-Current
8136150	Concho Rv nr Veribest, TX	-100.2195303	31.5376603	06/27/1998-09/29/2000
8136500	Concho Rv at Paint Rock, TX	-99.91952257	31.51599082	10/01/1915-Current
8136700	**Colorado Rv nr Stacy, TX**	**-99.57395318**	**31.49376898**	**10/01/1996-Current**
8138000	**Colorado Rv at Winchell, TX**	**-99.16227261**	**31.46793807**	**12/11/1923-06/29/2011**
8139500	Deep Ck nr Mercury, TX	-99.12171611	31.40238421	10/01/1953-10/02/1973
8140500	Dry Prong Deep Ck nr Mercury, TX	-99.13727189	31.40266186	07/01/1951-09/29/1971
8140700	Pecan Bayou nr Cross Cut, TX	-99.13033762	31.97264442	04/16/1968-06/04/2017
8140800	Jim Ned Ck nr Coleman, TX	-99.41478889	31.98319733	03/01/1965-10/02/1980
8141500	Hords Ck nr Valera, TX	-99.53479113	31.83431205	05/01/1947-12/31/1990
8142000	Hords Ck nr Coleman, TX	-99.42395349	31.84736747	10/01/1940-Current
8143500	Pecan Bayou at Brownwood, TX	-98.97393881	31.73181874	11/01/1923-10/04/1983
8143600	Pecan Bayou nr Mullin, TX	-98.74060307	31.51738612	10/01/1967-Current
8144500	San Saba Rv at Menard, TX	-99.78562537	30.91906245	10/01/2007-Current
8144600	San Saba Rv nr Brady, TX	-99.26894375	31.00405951	07/01/1979-09/29/2012

8144800	Brady Ck nr Eden, TX	-99.84118306	31.18433242	05/01/1962-10/09/1985
8145000	Brady Ck at Brady, TX	-99.33505525	31.13822366	06/01/1939-Current
8146000	San Saba Rv at San Saba, TX	-98.71948712	31.21322316	10/01/1915-Current
8147000	**Colorado Rv nr San Saba, TX**	**-98.56448402**	**31.21794515**	**11/01/1915-Current**
8148500	N Llano Rv nr Junction, TX	-99.80617897	30.5174101	06/15/2001-Current
8150000	Llano Rv nr Junction, TX	-99.73450968	30.50435487	10/01/1915-Current
8150700	Llano Rv nr Mason, TX	-99.10921855	30.66073685	03/07/1968-Current
8150800	Beaver Ck nr Mason, TX	-99.09588478	30.64351548	08/01/1963-Current
8151000	Llano Rv nr Castell, TX	-98.88365663	30.71684571	12/01/1923-09/29/1939
8151500	Llano Rv at Llano, TX	-98.6697599	30.75128991	11/20/1996-Current
8152000	Sandy Ck nr Kingsland, TX	-98.47225114	30.55768901	10/01/1998-Current
8152900	Pedernales Rv nr Fredericksburg, TX	-98.86976085	30.22048147	03/16/1998-Current
8153000	Pedernales Rv at Stonewall, TX	-98.66697728	30.25020201	08/01/1924-09/29/1934
8153500	Pedernales Rv nr Johnson City, TX	-98.3994674	30.29186695	10/01/1996-Current
8154000	Pedernales Rv nr Spicewood, TX	-98.08084871	30.42103112	12/01/1923-07/31/1939
8154510	Colorado Rv bl Mansfield Dam, Austin, TX	-97.90806655	30.39186795	10/01/1974-09/29/1990
8154700	Bull Ck at Loop 360 nr Austin, TX	-97.78473012	30.37214825	05/07/2001-Current

8155200	Barton Ck at SH 71 nr Oak Hill, TX	-97.92556496	30.29631587	10/01/2001-Current
8155240	Barton Ck at Lost Ck Blvd nr Austin, TX	-97.84472961	30.27409573	10/01/2007-Current
8155260	Barton Ck nr Camp Craft Rd nr Austin, TX	-97.8288959	30.27020723	09/01/1982-10/11/1988
8155300	Barton Ck at Loop 360, Austin, TX	-97.80222831	30.24465306	10/01/2007-Current
8155400	Barton Ck abv Barton Spgs at Austin, TX	-97.77222804	30.26354168	09/24/1998-Current
8156700	Shoal Ck at NW Pk at Austin, TX	-97.74500659	30.34742756	03/28/1975-09/29/1984
8156800	Shoal Ck at W 12th St, Austin, TX	-97.75028338	30.27659704	10/02/2001-Current
8157000	Waller Ck at 38th St, Austin, TX	-97.72694972	30.29715196	04/01/1955-10/23/1980
8157500	Waller Ck at 23rd St, Austin, TX	-97.73389418	30.28576347	01/01/1955-10/23/1980
8157600	E Bouldin Ck at S 1st St, Austin, TX	-97.75417195	30.25215352	04/03/1997-01/31/2001
8157700	Blunn Ck nr Little Stacy Pk, Austin, TX	-97.74389386	30.24743158	04/17/1997-01/31/2001
8158000	**Colorado Rv at Austin, TX**	**-97.69444803**	**30.24465429**	**02/26/1997-Current**
8158050	Boggy Ck at US Hwy 183, Austin, TX	-97.67250324	30.26326487	03/02/1976-09/29/2001
8158100	Walnut Ck at FM 1325 nr Austin, TX	-97.71167327	30.40992537	10/19/1984-09/07/1986
8158380	Little Walnut Ck at Georgian Dr, Austin, TX	-97.6980609	30.35437215	02/16/2008-Current
8158600	Walnut Ck at Webberville Rd, Austin, TX	-97.65500305	30.28326423	10/01/2001-Current
8158700	Onion Ck nr Driftwood, TX	-98.00778589	30.08298924	07/01/1979-Current

8158800	Onion Ck at Buda, TX	-97.8480606	30.0860467	02/01/1992-06/28/1995
8158810	Bear Ck bl FM 1826 nr Driftwood, TX	-97.94000762	30.15548749	10/01/2007-Current
8158819	Bear Ck nr Brodie Lane nr Manchaca, TX	-97.86111111	30.14416667	10/01/2003-09/29/2010
8158827	Onion Ck at Twin Creeks Rd nr Manchaca, TX	-97.82111613	30.12632345	04/03/2003-Current
8158840	Slaughter Ck at FM 1826 nr Austin, TX	-97.90334089	30.20909723	01/16/1978-Current
8158860	Slaughter Ck at FM 2304 nr Austin, TX	-97.83222788	30.16215542	05/31/2003-Current
8158880	Boggy Ck (S) at Circle S Rd, Austin, TX	-97.78222716	30.1807666	06/05/1985-05/15/1986
8158920	Williamson Ck at Oak Hill, TX	-97.86028483	30.23520806	10/24/2007-Current
8158922	Williamson Ck at Brush Country Blvd, Oak Hill, TX	-97.84139549	30.22631976	03/11/1993-09/29/2003
8158930	Williamson Ck at Manchaca Rd, Austin, TX	-97.79361675	30.22132064	01/25/2000-Current
8158970	Williamson Ck at Jimmy Clay Rd, Austin, TX	-97.73250397	30.18937803	08/25/2007-09/28/2012
8159000	Onion Ck at US Hwy 183, Austin, TX	-97.68861378	30.17798987	10/01/2001-Current
8159150	Wilbarger Ck nr Pflugerville, TX	-97.60083729	30.45464692	09/01/1963-10/02/1980
8162000	Colorado Rv at Wharton, TX	-96.10384821	29.30913668	03/01/2008-Current
8162500	Colorado Rv nr Bay City, TX	-96.01245877	28.97414621	05/01/1948-06/22/2018
8162600	Tres Palacios Rv nr Midfield, TX	-96.17107472	28.92803769	06/17/1970-Current
8164000	Lavaca Rv nr Edna, TX	-96.6863668	28.95998449	08/13/1938-Current

8164390	Navidad Rv at Strane Pk nr Edna, TX	-96.6741434	29.06553568	10/01/1996-Current
8164450	Sandy Ck nr Ganado, TX	-96.54636084	29.16025391	10/02/1997-Current
8164500	Navidad Rv nr Ganado, TX	-96.55247332	29.02581472	06/01/1939-07/05/1999
8164503	W Mustang Ck nr Ganado, TX	-96.46719265	29.07164592	10/01/1977-Current
8164504	E Mustang Ck nr Louise, TX	-96.41719132	29.07081227	10/01/1996-Current
8164600	Garcitas Ck nr Inez, TX	-96.81914857	28.89137591	08/22/2002-Current
8164800	Placedo Ck nr Placedo, TX	-96.76887018	28.72527137	06/16/1970-Current
8165300	N Fk Guadalupe Rv nr Hunt, TX	-99.38699162	30.06409747	08/01/1967-Current
8165500	Guadalupe Rv at Hunt, TX	-99.32171219	30.06993074	10/17/1941-Current
8166000	Johnson Ck nr Ingram, TX	-99.28310056	30.10020707	05/22/1987-Current
8166140	Guadalupe Rv abv Bear Ck at Kerrville, TX	-99.19532034	30.06965378	04/16/1999-Current
8166200	Guadalupe Rv at Kerrville, TX	-99.16337499	30.05326601	10/01/1986-Current
8166500	Guadalupe Rv nr Comfort, TX	-98.89253308	29.94938219	01/01/1918-09/29/1932
8167000	Guadalupe Rv at Comfort, TX	-98.8928112	29.96965926	05/31/1939-Current
8167500	Guadalupe Rv nr Spring Branch, TX	-98.38362748	29.8604957	12/23/1986-Current
8167600	Rebecca Ck nr Spring Branch, TX	-98.36973866	29.91854906	02/01/1960-02/26/1979
8167800	Guadalupe Rv at Sattler, TX	-98.18001056	29.85910758	05/01/2002-Current

8168500	Guadalupe Rv abv Comal Rv at New Braunfels, TX	-98.11000826	29.7149465	11/13/1990-Current
8169000	Comal Rv at New Braunfels, TX	-98.12250854	29.70605788	12/19/1927-Current
8169500	Guadalupe Rv at New Braunfels, TX	-98.10667482	29.69800265	01/27/1915-08/25/2011
8170500	San Marcos Rv at San Marcos, TX	-97.93417178	29.88910804	10/01/1995-Current
8171000	Blanco Rv at Wimberley, TX	-98.08889798	29.99438081	08/06/1924-Current
8171300	Blanco Rv nr Kyle, TX	-97.91000511	29.97938297	10/01/2001-Current
8172400	Plum Ck at Lockhart, TX	-97.67916669	29.92299833	04/30/1959-Current
8172500	Plum Ck nr Lockhart, TX	-97.58416394	29.82161359	02/01/1925-03/31/1930
8175000	Sandies Ck nr Westhoff, TX	-97.44943529	29.2152475	03/10/1930-Current
8175800	Guadalupe Rv at Cuero, TX	-97.3297129	29.09053147	10/01/2001-Current
8176500	Guadalupe Rv at Victoria, TX	-97.01304291	28.79304563	11/04/1934-Current
8177520	Guadalupe Rv nr Bloomington, TX	-96.96527778	28.66194444	10/01/2011-Current
8177700	Olmos Ck at Dresden Dr, San Antonio, TX	-98.51029579	29.49911923	06/07/1968-Current
8177860	San Antonio Rv at Woodlawn Ave, San Antonio, TX	-98.47862802	29.4513432	10/01/1990-09/29/1995
8178000	San Antonio Rv at San Antonio, TX	-98.49501705	29.40967796	02/01/1915-Current
8178050	San Antonio Rv at Mitchell St, San Antonio, TX	-98.49473915	29.39301189	10/01/1992-Current
8178500	San Pedro Ck at Furnish St, San Antonio, TX (CSG)	-98.51085076	29.40634472	08/01/1916-Current

8178505	San Antonio Rv at Theo Ave, San Antonio, TX	-98.4986281	29.38801205	05/20/1999-09/02/1999
8178565	San Antonio Rv at Loop 410 at San Antonio, TX	-98.45029322	29.32218127	10/01/1986-Current
8178585	Salado Ck at Wilderness Rd at San Antonio, TX	-98.56557553	29.63078135	12/06/1997-09/30/2012
8178700	Salado Ck at Loop 410 at San Antonio, TX	-98.43112714	29.5160633	02/19/2011-Current
8178800	Salado Ck at Loop 13 at San Antonio, TX	-98.41279255	29.35718017	09/01/1960-Current
8178880	Medina Rv at Bandera, TX	-99.07003496	29.72383537	10/01/1982-Current
8179000	Medina Rv nr Pipe Creek, TX	-98.97614292	29.67550339	12/01/1922-10/19/1982
8179100	Red Bluff Ck nr Pipe Ck, TX	-98.9555868	29.68105863	04/01/1956-11/27/1981
8179520	Medina Rv bl Medina Lk nr San Antonio, TX	-98.93530749	29.53411873	04/20/2001-07/01/2002
8180000	Medina Canal nr Riomedina, TX	-98.90336189	29.50550851	10/04/2004-09/29/2007
8180500	USGS Medina Rv nr Riomedina, TX	-98.90475079	29.49828651	01/23/2001-Current
8180640	Medina Rv at La Coste, TX	-98.81308102	29.32412462	12/19/1986-09/29/2000
8180700	Medina Rv nr Macdona, TX	-98.68974407	29.33495798	05/29/1987-Current
8180720	Medina Rv nr Von Ormy, TX	-98.64224261	29.29523712	05/08/2003-03/28/2016
8180800	Medina Rv nr Somerset, TX	-98.5814074	29.26218286	10/02/1997-Current
8181400	Helotes Ck at Helotes, TX	-98.69168963	29.57856108	10/01/1986-Current
8181480	Leon Ck at IH 35 at San Antonio, TX	-98.58418561	29.32995837	09/13/1984-Current

8181500	Medina Rv at San Antonio, TX	-98.49084933	29.26412761	10/01/2004-Current
8183850	Cibolo Ck at IH 10 abv Boerne, TX	-98.7536379	29.81466345	05/24/1996-05/09/2007
8183900	Cibolo Ck nr Boerne, TX	-98.69752458	29.77410932	10/01/1997-Current
8184000	Cibolo Ck nr Bulverde, TX	-98.42723877	29.72605615	05/01/1946-11/30/1965
8184500	Cibolo Ck abv Bracken, TX	-98.38362631	29.67522471	04/01/1946-09/29/1951
8185000	Cibolo Ck at Selma, TX	-98.31112416	29.5941166	10/01/2001-Current
8188750	GBRA Calhoun Canal-Flume No. 2 nr Long Mott, TX	-96.7613716	28.50277789	07/01/1972-03/31/1986
8188800	Guadalupe Rv nr Tivoli, TX	-96.88470836	28.5058337	08/04/2000-Current
8189200	Copano Ck nr Refugio, TX	-97.11249127	28.30361693	06/17/1970-Current
8189500	Mission Rv at Refugio, TX	-97.27915932	28.29195088	07/01/1939-Current
8189700	Aransas Rv nr Skidmore, TX	-97.62082909	28.28250426	03/27/1964-Current
8189800	Chiltipin Ck at Sinton, TX	-97.5038832	28.04695919	07/23/1970-09/29/1991
8190000	Nueces Rv at Laguna, TX	-99.9972871	29.42856679	10/01/1923-Current
8190500	W Nueces Rv nr Brackettville, TX	-100.2364633	29.47273118	11/03/1996-Current
8191500	Nueces Rv nr Uvalde, TX	-99.89617132	29.19579516	10/01/1927-09/29/1939
8192000	Nueces Rv bl Uvalde, TX	-99.89478224	29.12385307	04/05/1939-Current
8192500	Nueces Rv nr Cinonia, TX	-99.83366684	28.78358701	08/01/1915-09/29/1925

8193000	Nueces Rv nr Asherton, TX	-99.68199346	28.50026313	10/01/1997-Current
8194000	Nueces Rv at Cotulla, TX	-99.24003224	28.42637889	10/01/1926-Current
8194200	San Casimiro Ck nr Freer, TX	-98.96696778	27.96501329	01/01/1962-Current
8194500	Nueces Rv nr Tilden, TX	-98.5572384	28.30888906	12/01/1942-Current
8194600	Nueces Rv at Simmons, TX	-98.284454	28.42138408	04/01/1965-10/17/1977
8195000	Frio Rv at Concan, TX	-99.70477562	29.48856496	09/30/1924-Current
8196000	Dry Frio Rv nr Reagan Wells, TX	-99.78144517	29.50467576	02/27/1997-Current
8196500	Dry Frio Rv at Knippa, TX	-99.65866382	29.29190325	09/01/1952-09/29/1953
8197500	Frio Rv bl Dry Frio Rv nr Uvalde, TX	-99.67449782	29.24579341	10/01/2001-Current
8198000	Sabinal Rv nr Sabinal, TX	-99.49282413	29.49106441	10/01/1942-Current
8198500	Sabinal Rv at Sabinal, TX	-99.47976922	29.30162473	09/01/1952-Current
8199700	Frio Rv nr Frio Twn, TX	-99.40865542	29.08579856	05/01/1924-09/29/1927
8200000	Hondo Ck nr Tarpley, TX	-99.24670636	29.56967272	02/24/1997-Current
8200500	Hondo Ck nr Hondo, TX	-99.18559299	29.45162049	08/01/1952-10/31/1964
8200700	Hondo Ck at King Waterhole nr Hondo, TX	-99.15142524	29.39078876	10/01/2001-07/23/2006
8201500	Seco Ck at Miller Ranch nr Utopia, TX	-99.40309974	29.573284	05/01/1961-Current
8202000	Seco Ck nr Utopia, TX	-99.40643304	29.5505069	08/01/1952-09/29/1961

8202500	Seco Ck nr D'hanis, TX	-99.38809895	29.48911973	08/01/1952-10/31/1964
8202700	Seco Ck at Rowe Ranch nr D'Hanis, TX	-99.28781853	29.37078917	11/01/1960-Current
8204005	Leona Rv nr Uvalde, TX	-99.74338884	29.15440772	03/01/2003-Current
8204500	Leona Rv nr Divot, TX	-99.2411468	28.7930303	05/01/1924-09/29/1929
8205500	Frio Rv nr Derby, TX	-99.14475565	28.73664429	08/01/1915-Current
8206600	Frio Rv at Tilden, TX	-98.54751737	28.46749279	07/14/1978-Current
8206700	San Miguel Ck nr Tilden, TX	-98.54585129	28.58748787	01/25/1964-Current
8207000	Frio Rv at Calliham, TX	-98.34667835	28.49221399	10/01/1924-03/23/1981
8207500	Atascosa Rv nr McCoy, TX	-98.33834537	28.86497693	08/31/2002-Current
8208000	Atascosa Rv at Whitsett, TX	-98.28139882	28.62220899	05/22/1932-Current
8210000	Nueces Rv nr Three Rivers, TX	-98.17806252	28.42749545	07/01/1915-Current
8210300	Ramirena Ck nr George West, TX	-98.10333768	28.14195362	03/01/1968-03/31/1972
8210400	Lagarto Ck nr George West, TX	-98.09694821	28.05973533	10/02/2002-Current
8211000	Nueces Rv nr Mathis, TX	-97.86027692	28.03834719	08/05/1939-Current
8211100	Nueces Rv bl Mathis, TX	-97.80083137	27.98557161	01/27/1966-02/28/1967
8211200	Nueces Rv at Bluntzer, TX	-97.7758308	27.93779594	01/26/1966-Current
8211500	Nueces Rv at Calallen, TX	-97.62527306	27.88307697	10/01/1989-Current

8211503	Rincon Bayou Channel nr Calallen, TX	-97.6255509	27.89696521	01/27/1998-Current
8211520	Oso Ck at Corpus Christi, TX	-97.50193772	27.71141879	09/20/1995-Current
8211800	San Diego Ck at Alice, TX	-98.07555841	27.76669257	10/01/1963-09/29/1989
8211900	San Fernando Ck at Alice, TX	-98.03361304	27.77252575	04/03/1999-Current
8212000	San Fernando Ck nr Alice, TX	-97.9877787	27.72530545	04/01/1962-09/29/1963
8212400	Los Olmos Ck nr Falfurrias, TX	-98.13583782	27.26448618	04/01/1999-Current
8379500	**PECOS RIVER NEAR ANTON CHICO, NM**	**-105.1088898**	**35.17894022**	**10/01/1910-Current**
8382000	GALLINAS RIVER NEAR LOURDES, NM	-105.1602824	35.47087824	07/01/1951-Current
8382500	GALLINAS R NR COLONIAS, NM	-104.9002727	35.18199671	01/01/1951-Current
8382600	**PECOS R ABV CANON DEL UTA NR COLONIAS, NM**	**-104.8005477**	**35.09144382**	**01/01/1975-Current**
8382650	**PECOS RIVER ABOVE SANTA ROSA LAKE, NM**	**-104.7619356**	**35.05977809**	**02/28/1976-Current**
8382730	LOS ESTEROS CREEK ABOVE SANTA ROSA LAKE, NM	-104.6641554	35.09505534	07/26/1973-09/30/1997
8382760	LOS ESTEROS CR TRIB ABOVE SANTA ROSA LAKE, NM	-104.6727667	35.09311091	07/25/1973-01/31/1991

8382800	PECOS R AB LOS ESTE DAMSITE NR SANTA ROSA, NM	-104.6816559	35.04061221	10/01/1965-02/28/1977
8382830	**PECOS RIVER BELOW SANTA ROSA DAM, NM**	**-104.6863782**	**35.03116799**	**01/17/1980-Current**
8383000	**PECOS RIVER AT SANTA ROSA, NM**	**-104.6991562**	**34.94339212**	**10/01/1912-09/29/1992**
8383500	**PECOS RIVER NEAR PUERTO DE LUNA, NM**	**-104.5249857**	**34.73006242**	**05/01/1938-Current**
8384500	**PECOS RIVER BELOW SUMNER DAM, NM**	**-104.3877577**	**34.60423138**	**10/01/1912-Current**
8385000	FORT SUMNER MAIN CANAL NEAR FORT SUMNER, NM	-104.2783094	34.50839905	03/29/1939-Current
8385500	PECOS RIVER NEAR FORT SUMNER, NM	-104.2724756	34.4786776	10/01/1994-10/07/2003
8385520	PECOS RIVER BELOW FORT SUMNER, NM	-104.1730273	34.34812654	08/22/1957-09/15/1970
8385522	PECOS RIVER BELOW TAIBAN CREEK NEAR FORT SUMNER,NM	-104.1808052	34.33229367	08/12/1992-Current
8385620	PECOS RIVER BL. YESO ARROYO NR. FT. SUMNER, NM	-104.2296942	34.22785233	11/11/1964-09/29/1968
8385630	**PECOS RIVER NEAR DUNLAP, NM**	**-104.307194**	**34.06480192**	**08/20/1993-Current**
8385640	PECOS RIVER AB. HUGGINS CR. NR. ROSWELL, NM	-104.2783031	33.91536137	10/27/1964-09/29/1968

8385643	PECOS RIVER BELOW SIXMILE DRAW NEAR ROSWELL, NM	-104.2919136	33.85314051	09/30/2001-05/31/2003
8388000	RIO RUIDOSO AT HONDO, NM	-105.27554	33.38342216	10/01/1930-09/29/1955
8389500	RIO BONITO AT HONDO, NM	-105.27554	33.38897752	10/01/1930-09/29/1955
8390100	RIO HONDO AT PICACHO, NM	-105.1574818	33.35703652	12/04/1956-06/30/1962
8390500	RIO HONDO AT DIAMOND A RANCH NR ROSWELL, NM	-104.8519196	33.349264	10/01/1939-Current
8393200	ROCKY ARROYO AB TWO RIVERS RES NR ROSWELL, NM	-104.7969187	33.28537753	05/01/1963-09/29/1980
8397600	RIO PENASCO NEAR DUNKEN, NM	-105.178312	32.88204526	10/01/1956-Current
8449000	Devils Rv nr Juno, TX	-101.1453849	29.96353604	06/01/1925-09/29/1973
8459200	Rio Grande at Pipeline Crossing bl Laredo, TX	-99.48865287	27.4005766	09/27/1997-10/21/2007
73274406	Little Washita River ab SCS Pond No 26 nr Cyril,OK	-98.2508839	34.91478778	02/24/1995-09/29/2013
73274408	Little Washita River Tributary near Cyril, OK	-98.2336611	34.92589889	02/24/1995-09/29/2004
73274458	Little Washita River Tributary near Cement, OK	-98.1411578	34.86256806	06/01/1995-09/29/2004

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APPENDIX B: SUPPLEMENTAL TABLES FOR CHAPTER III

Table B.1. Sites listed from upstream to downstream with the dates of each tagging and recapture sampling event over the duration of the first field season (November 2018-June 2019) and second field season (November 2019-June 2020). The location of each site can be found on Figure 8.

Site	**Season 1**	**Season 2**
Roll	1/8/2019	1/22/2020
	3/18/2019	4/11/2020
Camargo	1/8/2019	3/3/2020
	3/18/2019	6/17/2020
Taloga	1/6/2019	1/20/2020
	3/9/2019	2/17/2020
	3/19/2019	3/3/2020
	4/18/2019	4/11/2020
		5/8/2020
		5/20/2020
		6/17/2020
Thomas	1/7/2019	1/3/2020
	2/3/2019	1/8/2020
	3/10/2019	2/3/2020
	3/20/2019	2/17/2020
	4/17/2019	3/3/2020
		4/11/2020
		5/8/2020

		5/19/2020
		5/20/2020
		6/17/2020
		6/18/2020
Fire Canyon	11/6/2018	11/9/2019
	11/11/2018	11/17/2019
	11/20/2018	11/24/2019
	11/21/2018	12/2/2019
	11/25/2018	12/10/2019
	11/26/2018	12/17/2019
	12/4/2018	12/10/2019
	12/12/2018	12/17/2019
	12/21/2018	12/27/2019
	1/17/2019	1/3/2020
	1/26/2019	1/4/2020
	2/8/2019	1/5/2020
	3/11/2019	1/9/2020
	3/21/2019	1/13/2020
	4/15/2019	1/23/2020
		1/25/2020
		2/4/2020
		2/8/2020
		2/15/2020
		2/22/2020

		3/1/2020
		3/2/2020
		4/12/2020
		5/7/2020
		5/18/2020
		5/21/2020
		6/16/2020
Little		1/26/2020
		2/16/2020
		3/3/2020
		5/9/2020
		5/22/2020
		6/16/2020
Caddo Jake	11/1/2018	11/8/2019
	11/7/2018	11/16/2019
	11/12/2018	11/25/2019
	11/19/2018	12/1/2019
	11/20/2018	12/11/2019
	11/27/2018	12/18/2019
	12/3/2018	1/6/2020
	12/10/2018	1/15/2020
	12/20/2018	1/21/2020
	1/18/2019	1/27/2020

	2/2/2019	2/8/2020
	3/8/2019	2/18/2020
	4/16/2019	3/4/2020
		5/9/2020
		6/15/2020
Braum’s	11/2/2018	11/15/2019
	11/13/2018	11/19/2019
	11/19/2018	11/22/2019
	11/25/2018	11/24/2019
	12/2/2018	11/30/2019
	12/13/2018	12/7/2019
	12/19/2018	12/8/2019
	1/25/2019	12/9/2019
	2/4/2019	12/19/2019
	4/22/2019	12/28/2019
		1/7/2020
		1/14/2020
		1/28/2020
		2/19/2020
		2/20/2020
		2/23/2020
		3/5/2020
		5/10/2020
Mustang	11/3/2018	11/10/2019

	11/8/2018	11/18/2019
	11/13/2018	12/16/2019
	11/14/2018	1/8/2020
	11/18/2018	1/16/2020
	11/28/2018	5/9/2020
	12/5/2018	
	12/11/2018	
	12/18/2018	
Norman	2/1/2019	11/19/2019
	6/4/2019	12/16/2019
	6/28/2019	5/10/2020
		6/18/2020

Table B.2. Pearson's product moment coefficient (r) for predictor variables used to develop mean daily displacement candidate models. Discharge (m^3/s) and water temperature (°C) were represented as the 10 day average for the 10 days prior to each recapture event. Only fishes tagged with p-Chips and recaptured within 14 days of tagging or previous recaptures were included. Photoperiod was minutes of daylight on the day at recapture. Total length was fish total length (mm).

	Discharge	**Photoperiod**	**Total length**	**Temperature**
Discharge	1.00			
Photoperiod	0.32	1.00		
Total length	0.07	-0.07	1.00	
Temperature	-0.22	0.42	-0.31	1.00

APPENDIX C: LABORATORY TAGGING EFFORTS

Methods

Experiment 1

I first completed a tagging study in a controlled lab setting to determine the size of fish appropriate for tagging with Passive Integrated Transponder (PIT) tags. I used Emerald Shiner as a surrogate for ARS to avoid unnecessary handling of a federally threatened species. PIT tags are electronic tags encased in glass and are magnetically charged rather than having an internal power source, resulting in very small tags with decades of operational life (Oregon, RFID). I used two size groups of fish: small (<0.9 g) and large (≥0.9 g). I also included control fish in each size group that were handled the same as experimental fish except with no tag inserted. To reduce stress while tagging and handling, I anesthetized individuals using tricaine methanesulfonate (MS-222) at 100 mg/L buffered with sodium bicarbonate to match the pH of the holding tank water (≈200 mg/L). Fish were anesthetized one at a time by submerging them in the anesthesia solution for 1-3 minutes (Topic Popovic et al. 2012; Wells et al. 2017), until fish lost equilibrium and operculum movements slowed. Once anesthetized, I measured fish total length (TL; 1.0 mm) and weight (0.01 g). Following Musselman et al. (2017), I injected a small (8 X 1.4 mm) full-duplex (FDX) PIT tag (Oregon RFID, Portland, OR, USA) into the peritoneum of each fish using a 1.6 mm diameter injection needle. Full-duplex PIT tags are smaller than half duplex tags but have shorter read ranges and are more susceptible to environmental noise. After tagging, fish were placed in an aerated recovery cooler until they resumed normal swimming behavior. Control fish were anesthetized and treated the same as treatment fish but without inserting a tag (i.e., just injecting the needle and then removing).

Fish were randomly assigned to experimental tanks for holding to examine mortality, tag retention, and growth for 90 days post-tagging. Up to 8 fish were randomly assigned to each of 8 20-g tanks. Control fish were evenly dispersed among tanks (i.e., 3 control fish were randomly assigned to each tank) to ensure that any tank effects would be evident in the control group. Fish were fed ≈0.5 tsp/tank twice daily a diet of frozen bloodworms (Fish Gum Drops Bloodworms, San Francisco Bay Brand, Newark, CA) and fish flakes (Advanced Nutrition Perfect Protein Tropical Fish Flake Food, Wardley, Secaucus, NJ). Water quality (i.e., pH, ammonia NH_3/NH_4^+, nitrite NO_2^-, and nitrate NO_3^-) and mortality were recorded daily, and tag loss was recorded either at the death of an individual or at the 30, 60, or 90-day marks by scanning all fish. In the event of a mortality the tank number, experimental group, TL, weight, and date were recorded. Dead specimens were measured, weighed, and then frozen for later examination.

Experiment 2

Due to hypothesized higher mortality during field tagging (based on low recaptures), I tagged and held ARS for 120 days to investigate the survival, retention, and growth of ARS. A very low recapture rate (<1%) over my first field season suggested Emerald Shiner may not respond to tagging in the same way. For this series of trials, I tagged ARS using either Visible Implant Elastomer (VIE; Northwest Marine Technology, Shaw Island, WA, https://www.nmt.us/visible-implant-elastomer/) or PIT tags. VIE tags are lightweight, cost effective, quick and relatively easy to use, and have high retention rates, making them useful for recapture of large numbers of fish where individual identification is not necessary (Walker and Adams 2016; Branco et al. 2017). Elastomer is a liquid that is injected into translucent tissue and solidifies but remains flexible and visible. Combining the two tag types provided a useful comparison.

I formed 12 experimental groups to examine the effects of VIE and PIT tags and the use of anesthesia on survival, retention, and growth of ARS. I had four VIE tag groups (described below), a PIT tag group, and a control group. Each of the six tagging groups contained individuals that were both anesthetized and not anesthetized for a total of 12 experimental groups (Appendix Table B.1). Fish were examined for injury or infection before they were allowed to be assigned to an experimental group. All fish with a defect were removed from the study and euthanized by overdose of tricaine methanesulfonate (MS-222). Fish of all sizes (36 – 56 mm TL) were allowed to be included in the control and VIE experimental groups. However, because PIT tags are larger and heavier than VIE tags, I only allowed fish ≥ 50 mm TL to be assigned to the PIT tag experimental groups. This length requirement was developed as a conservative cutoff to ensure lower mortality of a federally listed species in the field (see Experiment 1 results). Small-bodied fishes as small as 40-mm TL have high retention and survival associated with PIT tags (Musselman et al. 2017). Fish without visible defects were randomly assigned to an experimental group given the size restrictions. I anesthetized half of all individuals within each experimental group (PIT, VIE, and control) to determine the effects of anesthesia on mortality related to handling stress. Fish in an anesthetized group were anesthetized in the same manner as Experiment 1. Fish that were in non-anesthetized treatments skipped these steps and went directly to measurement and tagging. Similarly, fish in the PIT tag groups were tagged as described in Experiment 1.

Using VIE, I tagged ARS in four locations on the body to assess the mortality, tag retention, and growth differences among tag locations. The elastomer was injected subcutaneously into each fish according to manufacturer guidelines (Northwest Marine Technology Inc. 2017a). Elastomer injection should cease before the needle is fully removed to reduce the possibility of a ‘trail’ of elastomer from the wound. I used four tag locations commonly used to VIE tag cyprinids: the nape, parallel to the base of the dorsal fin, posterior to the dorsal fin, and the caudal peduncle (Figure 8; Northwest Marine Technology Inc. 2017b, 2017a). After tagging, fish were placed in a recovery cooler until they resumed normal swimming behavior.

I included a control group to examine mortality and growth effects between handling stress and tag stress. Half of the control group was anesthetized to examine the effects of anesthesia apart from tagging stress. Fish were measured and punctured with a needle, but not injected with a tag. The control group allowed me to determine how much mortality was due to tag stress and if the tags affect the growth of fish.

Fish were randomly assigned to experimental holding tanks to examine mortality, tag retention, and growth for 120 days post-tagging. Up to 10 fish were allowed to be randomly assigned to each of 12 20-g tanks with two control fish randomly assigned to each tank. Minnows typically school together and thus, were assigned to groups rather than individual tanks. Fish were fed and held under the same conditions as in Experiment 1, except the trial duration was increased to 120 days. Individuals that died within 2 days post-tagging were replaced with a new individual if specimens were available.

Experiment 3

Due to high initial mortality (≥30%) across all experimental groups in Experiment 2, I conducted additional trials to examine mortality and retention of PIT tags with shorter tagging time. Increased handling time increases fish stress and possibly mortality (Bolland et al. 2009; Ramsay et al. 2009). This trial differed from the initial experiment by retaining only the anesthetized control (n = 24) and PIT tag (n = 24) experimental groups with fish ≥ 50 mm TL and omitting the growth portion of the study. I anesthetized all individuals in this trial because Experiment 2 shows that anesthetized fish had higher survival. I omitted the growth portion of the study because weighing individuals added the most time to the tagging process and would not typically be conducted in the field. Additionally, all individuals in Experiment 2 gained weight, which indicated growth was not inhibited by PIT tagging. All other factors were treated the same as with Experiment 2.

Experiment 4

I examined ARS mortality and tag retention related to p-Chips, a relatively new type of tag. P-Chips are microtransponder tags (500 µm x 500 µm x 100 µm) with photocells that are powered by a handheld laser wand to emit a unique 9-digit signal (PharmaSeq Inc., Princeton, NJ). Little research has been published on the survival and tag retention associated with p-Chips in fishes, but Zebrafish (*Danio rerio*) have a 96% retention rate and no noticeable effects on fish health (Chen et al. 2013). I included a p-Chip experimental group (n = 62) and a control group (n = 40) in this study where control fish were treated the same (i.e., anesthetized, handled, and punctured with a needle) except without tag placement. All sizes of fish were considered in the p-Chip tag study (30–56 mm TL), because this protocol was developed for Zebrafish ≥ 20 mm. Additionally, a study examining the mortality and retention associated with p-Chips in larval European sea bass (*Dicentrarchus labrax L.*) found very small individuals (10 mm standard length) had high mortality (38%), but individuals ≥ 20-mm standard length had relatively low mortality (≤16%; Faggion et al. 2020). I injected each fish with a p-Chip subcutaneously left of the base of the dorsal fin using a 0.8 mm diameter injection needle according to manufacturer guidelines (Pharmaseq Inc 2020). During tagging, fish were held in a shallow dish to allow water to the gills and reduce handling stress. The placement and nature of tagging with p-Chips made it possible to keep fish submerged throughout the tagging process, unlike VIE and PIT tags. Each p-Chip was scanned with the handheld laser reader after tagging to ensure it was functioning and record tag numbers. I used the same 120 day period and holding conditions as described in Experiment 2.

Results

Experiment 1

Total length of Emerald Shiner ranged 35-61 for experimental fish (n = 36) and 34-55 for control fish (n = 24) (Appendix Table B.2). Survival (i.e., percentage alive) was 100% for all groups 24 h post tagging. After 30 days, all mortalities occurred in the small PIT tag group (50%; Appendix Table B.3).

Retention (i.e., percentage tags retained by living fish) was slightly higher for the small PIT tag group (86%) than the large PIT tag group (82%) at 30 days post tagging. At 60 days post tagging, a single mortality occurred in each the small control and large PIT groups (Appendix Table B.3). After 60 days, 5 large fish were missing their tags. A single small PIT-tagged fish died and one additional tag was lost in the large PIT group between 60 and 90 days post tagging. Both size groups of PIT-tagged fish grew more than control fish (Appendix Table B.3). These results indicate that PIT tag survival and retention are relatively high for small-bodied fishes. However, field tagging resulted in a very low recapture rate (<1%), showing that these results may be dependent on ambient conditions (e.g., temperature). Emerald Shiner also may not be an appropriate surrogate for ARS tagging.

Experiment 2

Total length for ARS ranged 50-56 for PIT-tagged fish (n = 30), 36-51 for VIE tagged fish (n = 75), and 39-53 for control fish (n = 30) (Appendix Table B.4). Dead VIE and control fish were replaced on the first (n = 9) and second (n = 7) days post tagging (Appendix Table B.5). PIT-tagged fish were not replaced because I did not have specimens large enough (i.e., ≥50 mm). All mortality occurred within 16 days post tagging. Survival was higher for the anesthetized group of almost every category (Appendix Table B.6), indicating anesthesia reduced handling stress for ARS. VIE tags had higher survival and retention than PIT tags, but the control group also had low survival (Appendix Table B.6). All tag loss was recorded at the end of the trial (i.e., 120-days post tagging). Survival was low across treatments, which may show that handling stress is the primary cause of death rather than actual tagging. Interestingly, the nape VIE location had the highest mortality and should not be used for future ARS tagging. Several nape tags showed signs of infection at death.

Experiment 3

ARS total length ranged from 45-49 (45.38 ± 2.10; mean ± standard deviation (SD)) for control fish and 50-56 (50.92 ± 1.56; mean ± SD). Survival was relatively high for the control (88%; n = 24) and PIT tag (83%; n = 24) groups. PIT tag retention was 45%, with most tag loss occurring in the first two months (9 of 11). Although PIT tag mortality was minimized with shorter handling times, tag retention was still low. More individuals would need to be tagged in field studies to increase recapture rates.

Experiment 4

ARS total length ranged from 34-55 (44.05 ± 4.72; mean ± SD) for control fish and 33-56 (44.34 ± 5.25; mean ± SD) for p-Chip fish. Survival was relatively high for the control (85%; n = 40) and p-Chip (87%; n = 62) groups. P-Chip tag retention was much higher than PIT tags (72%), with most tag loss occurring in the first month (13 of 15). These results indicate that p-Chips are a more useful tag type for small-bodied, sensitive, or federally listed fishes. High survival and retention rates result in higher recapture rates than other tag types (e.g., VIE and PIT tags).

Table C.1. Arkansas River Shiner *Notropis girardi* lab Passive Integrated Transponder (PIT) and Visible Implant Elastomer (VIE) tag study. Multiple groups were included in the study to evaluate the effects of anesthesia (MS-222) on the fish and tag retention of the tag in different locations. VIE tagging locations were anterior to the dorsal fin (nape), laterally adjacent to the dorsal fin (dorsal), posterior to the dorsal fin (rear dorsal), and on the caudal peduncle (caudal). See Figure 7 for a visual representation of VIE tag locations.

Experimental group	**Anesthesia**	**No anesthesia**
Control	12	12
PIT	15	16
VIE nape	8	8
VIE dorsal	8	8
VIE rear dorsal	8	9
VIE caudal	8	8
VIE total	32	33
Total	58	60

Table C.2. Summary of total lengths and weights of Emerald Shiner *Notropis atherinoides* in laboratory passive integrated transponder (PIT) tagging study (experiment 1). The mean, standard deviation (SD), minimum, and maximum values are reported.

Treatment parameter	**Mean ± SD**	**Minimum**	**Maximum**
Large Control			
Total length	53.20 ± 1.08	52	55
Weight	1.02 ± 0.08	0.93	1.12
Small Control			
Total length	42.22 ± 5.28	34	51
Weight	0.56 ± 0.16	0.28	0.79
Large PIT tag			
Total length	53.66 ± 2.73	48	61
Weight	1.05 ± 0.12	0.9	1.35
Small PIT tag			
Total length	43.18 ± 5.19	35	50
Weight	0.62 ±0.18	0.36	0.83

Table C.3. Survival, retention, and growth of Emerald Shiner *Notropis atherinoides* laboratory passive integrated transponder (PIT) tagging study (experiment 1). Emerald Shiner were tagged using 8-mm X 1.4-mm full-duplex tags. Fish were tagged in groups based on weight where 'large' were 0.9-1.35 g and 'small' were < 0.9 g. Control fish (large control and small control) were treated exactly the same as the tagged fish (i.e., anesthetized and punctured with 1.6 mm diameter needle), but no tag was inserted. Fish were held in 20 gallon tanks and fed two times daily. Tag retention ('retention') was assessed by scanning all fish 30, 60, and 90-d post tagging. Fish growth ('growth') was assessed at the conclusion of the study (90-d post tagging) and is the percentage of individuals in that group that gained weight over the study period.

Treatment	Sample size	30-day retention	30-day survival	60-day retention	60-day survival	90-day retention	90-day survival	Growth
Large control	5	NA	100%	NA	100%	NA	100%	80%
Small control	19	NA	100%	NA	95%	NA	95%	53%
Large PIT tag	22	82%	100%	77%	95%	73%	95%	91%
Small PIT tag	14	86%	50%	86%	50%	86%	43%	71%

Table C.4. Summary of total lengths and weights of Arkansas River Shiner *Notropis girardi* in the laboratory passive integrated transponder (PIT) and Visible Implant Elastomer (VIE) tagging study (experiment 2). The mean, standard deviation (SD), minimum, and maximum values are reported. Fish in the anesthetized group were anesthetized with a 100 g/L solution of Tricaine Methanesulfonate (i.e., MS-222) buffered with sodium bicarbonate until a pH level of 7.2 was reached to match the holding tank. Fish in the PIT tag group were a minimum of 50-mm in total length and were tagged with 8-mm PIT tags in the peritoneum. Fish in the VIE tag group had no length limits and were tagged with VIE tags at a randomly selected location. Dead fish were replaced at one and two days post-tagging, except for PIT-tagged individuals because there were too few ≥ 50-mm fish. VIE tags locations are nape/anterior to the dorsal fin, dorsal/laterally adjacent to the dorsal fin, rear dorsal/posterior to the dorsal fin, caudal/on the caudal peduncle. Fish tag locations are shown in Figure 8. Control fish were treated exactly the same as the tagged fish (i.e., anesthetized and punctured with 1.6 mm diameter needle), but no tag was inserted.

Treatment	**Sample size**	**TL mean ± SD**	**TL minimum**	**TL maximum**	**Weight mean ± SD**	**Weight minimum**	**Weight maximum**
No anesthesia							
Total	70	46.54 ± 3.55	39	55	0.65 ± 0.15	0.43	0.96
Control	16	45.25 ± 3.53	39	53	0.59 ± 0.14	0.43	0.94
PIT	15	51.47 ± 1.36	50	55	0.84 ± 0.09	0.61	0.96
VIE total	39	45.17 ± 2.27	40	51	0.60 ± 0.10	0.46	0.89
Caudal	10	44.70 ± 2.06	40	47	0.59 ± 0.09	0.47	0.72
Dorsal	11	45.14 ± 2.67	41	49	0.60 ± 0.08	0.49	0.75

Nape	9	45.11 ± 1.62	43	47	0.60 ± 0.10	0.46	0.81
Rear dorsal	9	45.78 ± 2.73	42	51	0.61 ± 0.12	0.46	0.89
Anesthesia							
Total	65	45.72 ± 3.68	36	56	0.63 ± 0.13	0.41	0.97
Control	14	45.29 ± 2.16	41	51	0.62 ± 0.09	0.48	0.84
PIT	15	50.87 ± 1.51	50	56	0.83 ± 0.07	0.72	0.97
VIE	36	43.75 ± 2.60	36	49	0.56 ± 0.08	0.41	0.72
Caudal	8	43.50 ± 2.56	39	48	0.54 ± 0.06	0.46	0.64
Dorsal	8	44.75 ± 4.33	36	49	0.57 ± 0.09	0.41	0.67
Nape	10	42.90 ± 1.73	40	46	0.53 ± 0.08	0.45	0.7
Rear dorsal	10	44.00 ± 1.33	42	46	0.59 ± 0.08	0.46	0.72

Table C.5. The number of Arkansas River Shiner *Notropis girardi* dead and replaced from the first and second days after tagging. Fish in the anesthetized group were anesthetized with a 100 g/L solution of Tricaine Methanesulfonate (i.e., MS-222) buffered with sodium bicarbonate until a pH level of 7.2 was reached to match the holding tank. Fish in the PIT tag group were a minimum of 50-mm in total length and were tagged with 8-mm Passive Integrated Transponder (PIT) tags in the peritoneum. Fish in the VIE tag group had no length limits and were tagged with Visible Implant Elastomer (VIE) tags at a randomly selected location. Dead fish were replaced at one and two days post-tagging, except for PIT-tagged individuals because there were too few ≥ 50-mm fish. VIE tags locations are nape/anterior to the dorsal fin, dorsal/laterally adjacent to the dorsal fin, rear dorsal/posterior to the dorsal fin, caudal/on the caudal peduncle. Fish tag locations are shown in Figure 8. Control fish were treated exactly the same as the tagged fish (i.e., anesthetized and punctured with 1.6 mm diameter needle), but no tag was inserted.

Treatment	**Day 0 sample size**	**Dead day 1**	**Added day 1**	**Dead day 2**	**Added day 2**
Total	119	11	9	9	7
Anesthetized	59	4	3	4	3
Not anesthetized	60	7	6	5	4
Control	24	2	3	3	3
PIT tag	30	3	0	2	0
VIE total	65	6	6	4	4
VIE nape	16	2	2	1	1
VIE dorsal	16	1	1	2	2

VIE rear dorsal	17	1	1	1	1
VIE caudal	16	2	2	0	0

Table C.6. Survival, retention, and growth of Arkansas River Shiner *Notropis girardi* laboratory passive integrated transponder (PIT) and Visible Implant Elastomer (VIE) tagging study (experiment 2). Fish in the anesthetized group were anesthetized with a 100 g/L solution of Tricaine Methanesulfonate (i.e., MS-222) buffered with sodium bicarbonate until a pH level of 7.2 was reached to match the holding tank. Fish in the PIT tag group were a minimum of 50-mm in total length and were tagged with 8-mm PIT tags in the peritoneum. Fish in the VIE tag group had no length limits and were tagged with VIE tags at a randomly selected location. VIE tags locations are nape/anterior to the dorsal fin, dorsal/laterally adjacent to the dorsal fin, rear dorsal/posterior to the dorsal fin, caudal/on the caudal peduncle. Fish tag locations are shown in Figure 8. Control fish were treated exactly the same as the tagged fish (i.e., anesthetized and punctured with 1.6 mm diameter needle), but no tag was inserted.

	Total	**Survival**	**Retention**	**Growth**
Anesthetized				
Total	65	58%	82%	94%
Control	14	43%		83%
PIT tag	15	53%	50%	100%
VIE tag	36	67%	88%	95%
Nape	10	50%	100%	100%
Dorsal	8	75%	83%	80%
Rear dorsal	10	40%	67%	100%
Caudal	8	88%	100%	100%
Not anesthetized				
Total	70	54%	87%	100%
Control	16	31%		100%
PIT tag	15	47%	57%	100%

VIE tag	39	67%	92%	100%
Nape	9	44%	100%	100%
Dorsal	11	73%	88%	100%
Rear dorsal	9	78%	100%	100%
Caudal	10	70%	86%	100%

Appendix C: References

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