12-2017

# Habitat and Population Assessments of the Lake Sturgeon *Acipenser fulvescens* Reintroduced to the Upper Tennessee River

Daniel James Walker

*University of Tennessee*

To the Graduate Council:

I am submitting herewith a dissertation written by Daniel James Walker entitled "Habitat and Population Assessments of the Lake Sturgeon *Acipenser fulvescens* Reintroduced to the Upper Tennessee River." I have examined the final electronic copy of this dissertation for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Doctor of Philosophy, with a major in Natural Resources.

J. Brian Alford, Major Professor

We have read this dissertation and recommend its acceptance:

Mark Bevelhimer, Vasileios Maroulas, Liem Tran

Accepted for the Council:
Carolyn R. Hodges

Vice Provost and Dean of the Graduate School

(Original signatures are on file with official student records.)

# Habitat and Population Assessments of the Lake Sturgeon *Acipenser fulvescens* Reintroduced to the Upper Tennessee River

A Dissertation Presented for the

Doctor of Philosophy

Degree

The University of Tennessee, Knoxville

Daniel James Walker

December 2017

## DEDICATION

I dedicate this work with tremendous appreciation and abiding love to my wife, Rebecca Lindsey Waddell Walker. I would never have had the stamina and fortitude necessary to pursue this degree and complete the work here without her unending support, patience, and compassion.

# ACKNOWLEDGEMENTS

I gratefully acknowledge first and foremost my support system comprised of my immediate family: my parents Jim and Kay Walker, my brother Evan Walker, and my wife, Becca. My family has always supported my pursuit of knowledge and education, and I am deeply grateful for that. I first learned to love and be inquisitive about things that live in the water from my Mom, who taught me to fish.

I next would like to acknowledge the many incredible instructors I have encountered during my scientific training, beginning with Mr. Ned Granville, who during my high school Honors Biology course first introduced me to the study of living things. I am deeply grateful to Drs. Michael Gangloff, Robert Creed, and Leslie Sargent Jones for building the foundations for my scientific career, and providing me with my first opportunities to conduct real research during my time at Appalachian State University. I am next indebted to Dr. J. Larry Wilson, who took a chance on me to let me pursue my graduate career at the University of Tennessee. I thank my primary doctoral advisor, Dr. J. Brian Alford, under whose guidance I feel that I was able to develop the skills and pursue the research that ultimately allowed me to consider myself a ‘real’ scientist. I am also thankful for the invaluable instruction and material contributions I have received for this research from my doctoral committee: Dr. Mark Bevelhimer, ORNL, Dr. Liem Tran, UTK, and Dr. Vasileios Maroulas, UTK.

Finally, I would like to acknowledge all of the help I received while developing and conducting these research projects from the people of the Fisheries Lab at the University of Tennessee. Without their many hours of hard work in the field, I would never have reached this point.

# ABSTRACT

The Lake Sturgeon *Acipenser fulvescens* historically occurred throughout the United States and Canada. However, due to widespread overfishing and habitat loss it was extirpated from much of its range, especially in the lower latitudes. Since the year 2000, fisheries managers have been working to restore this species to the Tennessee and Cumberland Rivers where it has been extirpated since c. 1961. This reintroduction is comprised of annual releases of young-of-the-year Lake Sturgeon reared in head-start aquaculture facilities around the Southeastern U.S., and annual monitoring efforts that track the spread and growth of reintroduced individuals. In 2015, a management plan guiding this reintroduction effort was drafted which included a variety of research needs to assist with and improve the ongoing restoration of this species. Two of these research needs are an assessment of habitats available to and occupied by Lake Sturgeon in the Upper Tennessee River, and a quantitative assessment of population size.

In this dissertation, I explain how I addressed these two research needs, and based on the results, I offer management recommendations for the continued success of Lake Sturgeon recovery in the Southeastern U.S. I characterized two important types of habitat relevant to different life stages of the species: spawning habitat and summer holding areas. I also used 5 years of mark-recapture data to generate the first quantitative assessments of population density and size-specific survival. My results indicate that there is ample suitable spawning substrate within the tailwaters I surveyed. I collected detailed measurements of various physical habitat variables from an area suspected to be important summer refugia for this species and describe in detail the physical habitat characteristics of this important area of habitat. I used a population model to evaluate the mark-recapture data, and found that while Lake Sturgeon are persisting in the Upper Tennessee River, many fall into the slowest-growth category. Finally, I used simulations to show that without natural recruitment, current stocking rates are unlikely to reach stated population goals through stocking alone. The information I provide here will be instrumental in aiding the adaptive management of this population.

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# INTRODUCTION

## *Acipenseriformes in North America*

The order Acipenseriformes, the sturgeons and paddlefish, enjoyed a wide Holarctic distribution across much of their 200 million year evolutionary history (Bemis and Kynard 1997). However, due in large part to overharvest and degradation of habitat, many of the 27 extant species are now considered threatened or endangered (Billard and Lecointre 2001). There are ten extant species of Acipenseriformes comprising three genera that inhabit the freshwater and marine systems of North America (Cech and Doroshov 2004). All ten of these species are considered vulnerable, threatened, or endangered by the American Fisheries Society's Endangered Species Committee, and many have been afforded state or federal protections (Jelks et al. 2008). Indeed, the Shortnose Sturgeon (*Acipenser brevirostrum*), native to the Atlantic Slope, was included in the very first list of endangered species produced by the U.S. Fish and Wildlife Service under the Endangered Species Protection Act of 1966 (USFWS 1967).

The alarming conservation statuses of these species in North America is due in large part to a brief period of intense harvesting pressure that peaked in the late 19$^{th}$ and early 20$^{th}$ centuries (Saffron 2004). During this time, there was an increase in global demand for American and Canadian caviar and sturgeon meat. This increase in global demand and the development of suitable processing and canning techniques among sturgeon processors of the east coast led to a boom in harvest and export of some sturgeon species native to North America. At the same time, all species within this order exhibit several life history traits that make them inherently susceptible to overharvest. They are long-lived and grow slowly, taking relatively longer to achieve sexual maturity than other fish species, and they exhibit spawning periodicity, whereby individual fish do not spawn in successive years (Scott and Crossman 1973; Bemis and Kynard 1997; Sadovy 2001).

In addition to being susceptible to overharvest, all North American species of Acipenseriformes exhibit some form of migration over the course of their life history (Bemis and Kynard 1997). Migration patterns vary among species. Within the United States and Canada, five species are fully or semi-anadromous, spawning in freshwater and maturing in estuarine or marine environments. The remaining five species are potamodromous, thus they inhabit freshwater systems only and migrate upstream to spawn (Boreman 1997). Life history differences aside, all Acipenseriformes species require upstream connectivity to suitable spawning habitat for successful reproduction. Therefore, the reduction in population sizes and ranges of Acipenseriformes can be attributed to the loss of essential migratory routes and habitats due to anthropogenic effects, primarily the construction of dams, within watersheds (Wilson and McKinley 2004).

***The Lake Sturgeon***

The Lake Sturgeon (*Acipenser fulvescens*) is an Acipenseriforme species that inhabits freshwater for the duration of its life history, and exhibits potamodromous migrations to spawn (Boreman 1997). The Lake Sturgeon occurred broadly in larger rivers and lakes of the Mississippi River drainage as well as the Great Lakes and Hudson Bay drainages (Harkness and Dymond 1961; Scott and Crossman 1973; Becker 1983; Etnier and Starnes 1993). However, demand for the meat, roe, and swim bladder of this long-lived, slow growing fish in the 19th and early to mid-20th centuries drove commercial fishing to overharvest fish stocks to the point of collapse in much of its range within the U.S. (Williamson 2003).

Lake Sturgeon life cycles are defined by a relatively long period of growth prior to maturity and spring spawning migrations. Becker (1983) stated that in Wisconsin, female Lake Sturgeon do not reach sexual maturity until 140 cm total length (TL, the length from the anterior-most part of the fish to the furthest tip of the caudal fin (Anderson and Gutreuter 1983)), or approximately 24-26 years old, while males become sexually mature around 114 cm TL, or approximately 20-21 years old. However, in the warmer waters of the Southeastern U.S., the Lake Sturgeon could have reached sexual maturity at a younger age (Etnier and Starnes 1993). When mature, Lake Sturgeon migrate from their lentic habitats to more lotic systems to spawn, often encountering physical barriers in the form of dams that can curtail preferred spawning substrate and reduce reproductive success of Lake Sturgeon in the wild (Auer 1996).

The low reproductive rates, fragmentation and loss of habitat, and historic overfishing have left the Lake Sturgeon in peril across much of its historic range. The Lake Sturgeon is considered endangered in eight U.S. states and threatened in three U.S. states and six Canadian provinces (Peterson et al. 2006). Populations are believed to be extirpated from the mid-southern to southern reaches of the Mississippi River, where numbers may have been low prior to anthropogenic alterations to the river (Etnier and Starnes 1993; Williamson 2003).

Prior to restoration efforts, the last record of Lake Sturgeon in the Upper Tennessee River was collected c. 1960 (Etnier and Starnes 1993). In 1987, the Tennessee Valley Authority (TVA) began its Reservoir Releases Improvement program at dams it manages along the Tennessee River system in the states of Tennessee and Alabama. In the following nine years, the $50 million improvement program improved the water quality of rivers flowing downstream of 20 TVA-managed hydroelectric dams (Higgins and Brock 1999). The primary goals of the improvement efforts were to ensure that rivers downstream of the hydroelectric dams were meeting minimum dissolved oxygen requirements conducive to aquatic life (daily average dissolved oxygen levels of $\geq$ 5.0 mg/L for tailwaters with warmwater fisheries), and that reaches susceptible to periods of zero flow supported required minimum flow levels (Mansfield 2014). Subsequent analyses of metrics of water quality and benthic macroinvertebrate communities below some TVA dams found significant improvements as a result of the implementation of the program (Bednarek and Hart 2005).

As river conditions began improving across Tennessee, managers recognized that areas of the Holston and French Broad Rivers appeared to meet water quality and habitat requirements that could support Lake Sturgeon reintroduction (Southeastern Lake Sturgeon Working Group (SLSWG) 2013). In 2000, 41 age-2 Lake Sturgeon were implanted with radio telemetry devices and released into the French Broad River to monitor success (Martin 2001). These fish survived and persisted within the system at rates up to 75%, so the Tennessee Aquarium Conservation Institute (TNACI) in conjunction with three USFWS National Fish Hatcheries began head-start aquaculture for Lake Sturgeon reared from Wolf River, WI, brood stock (SLSWG 2013). A biotelemetry study of reintroduced juvenile Lake Sturgeon found that Lake Sturgeon released into the French Broad River displayed a persistence rate of 50%, with individuals dispersing throughout the system (Huddleston 2006). Now, a total of 16 TVA hydroelectric dams in the Tennessee River system employ some form of reservoir release improvement system. Yearly Lake Sturgeon reintroduction and monitoring efforts in the Tennessee and Cumberland Rivers continue under the guidance of the Southeastern Lake Sturgeon Working Group (formerly Tennessee Lake Sturgeon Reintroduction Working Group), a partnership of federal, state, and local agencies, non-governmental organizations, and universities.

### *Aquatic habitat and river ecology*

Aquatic habitat is the summation of the physical, chemical, and biological features comprising the environment that biota interact with for protection, reproduction, rearing, foraging, or resting (Maddock 1999). Characteristics of physical habitat in rivers, for example, include substrate composition, depth, velocity, temperature, dissolved oxygen, conductivity, organic matter, turbidity, and salinity, which are in turn largely influenced by factors such as the climate and geomorphology of the river system at the basin and watershed scale (Thorp et al. 2006). The Lake Sturgeon is native to large rivers and lakes, and as such its physical habitat is defined by the processes governing large river systems (those with mean discharge >350 $m^3$/sec) (Harkness and Dymond 1961; Scott and Crossman 1973; Nilsson et al. 2005). Generally speaking, sturgeons prefer moderately turbid, cool (< 25$^oC$ annual mean), well-oxygenated waters (> 3 mg $O_2$/L) (Cech and Doroshov 2004). Furthermore, Lake Sturgeon require areas of habitat with both clean, rocky substrate and high benthic macroinvertebrate densities during various phases of their life history (Harkness and Dymond 1961; Scott and Crossman 1973). The Upper Tennessee River is a large high-order (Strahler stream order > 6) river system, with annual mean discharge of 906 $m^3$/sec as measured at the outlet of Chickamauga Dam (Strahler 1957; NRC 2015). In the 2013 management and recovery plan for Lake Sturgeon in the

Tennessee and Cumberland Rivers, the SLSWG outlined several management objectives. The first objective listed was to assess the habitat and carrying capacity for Lake Sturgeon in the river systems where they were reintroduced. For proper ecological context, it is necessary, to explore the factors that likely define the available habitat for Lake Sturgeon in the Upper Tennessee River.

Within the field of stream ecology, there have been periodic efforts to develop a generalized unifying theory of stream function that relates abiotic and biotic factors of river ecosystems, which in turn influence the availability of habitat within a system, in a predictable manner across broad spatial scales. The general trend over the course of the development of river ecology theories has been one of increasing complexity and spatial scale, and an emphasis on hierarchical classification of abiotic and biotic processes within watersheds.

The river continuum concept (RCC) (Vannote et al. 1980) was developed as an early attempt at describing a generalized mechanism for the determination of riverine communities via relationships with changing biotic and abiotic factors, and it received widespread testing, criticism, and modification following its publication. The core tenet of the RCC is that river systems exhibit longitudinal, generally one-way transport of organic matter from the headwaters (stream order 1-3) through mid-sized river reaches (stream order 4-6) into large river channels (stream order > 6). The authors postulated that there was a predictable gradient to the changes in source and size of organic matter along the river continuum, from allochtonous coarse particulate organic matter in the headwaters to autochthonous fine particulate and dissolved organic matter downstream. In addition to changes in the morphology of the rivers as they become larger, these changes in the sources of energy within the riverine ecosystem were then linked to changes in the invertebrate and fish communities.

The continual gradient aspect of the RCC theory was subsequently challenged by the resource spiralling concept, which noted that the processing of organic matter and mobilization and immobilization of nutrients are affected by the uneven unidirectional downstream flow inherent to river systems which leads to partially open or spiralling cycles (Webster and Patten 1979; Newbold et al. 1982). Both the RCC and the resource spiraling concept focused more specifically on the resource and nutrient pathways at the base of river trophic systems, which then determine the biotic communities through trophic linkages. Changes in the bioavailability and types of organic matter and energy available to primary consumers (e.g. benthic macroinvertebrates) throughout a river system are critical to determining the development of appropriate forage bases for high-level consumers, such as Lake Sturgeon, in a specific habitat.

The serial discontinuity concept built upon the core tenets of the RCC by describing the influences of dams as major interruptions in the proposed continuous gradient along rivers which can sequester or isolate biotic and abiotic factors for extended periods of time (Ward and Stanford 1983; Ward and Stanford 1995). The serial discontinuity concept states that sequestration of river flow into reservoirs, and anthropogenic alterations to the thermal and flow regimes of the downstream tailwaters via manipulation of releases (such as for hydropower generation) effectively reset the river continuum from a mid- or large-sized river back to characteristics typical of small streams with lower orders. The interruptions in the natural hydrological regime caused by dams can then lead to drastic changes in physical habitat variables, such as year-round cooling of water, fine sediment deposition in reservoirs and scouring in tailwaters, and low dissolved oxygen concentrations in downstream reaches. These changes in the physical composition of the tailwaters and reaches further downstream can lead to the most drastic changes in the biotic community if the dam releases are unmitigated, hypolimnetic releases of cold, hypoxic water.

At large spatial and temporal scales, hierarchical classification of streams (Frissell et al. 1986) is often employed to explain the spatiotemporal causal elements governing habitat variables at a certain river segment, reach, or microhabitat. This theory states that physical habitat at smaller scales, which may vary in small time increments (e.g., days or weeks), is determined by factors operating at the next largest spatial scale in a nested pattern up to basin-level spatial scales that vary in large time increments (e.g., thousands of years). For example, the substrate found at the microhabitat scale (e.g., 1 $m^2$) of a particular habitat unit (e.g. a riffle that is 100 $m^2$) is determined by the gradient and runoff at the reach scale (e.g., 1,000 $m^2$), which are in turn determined by the geology and morphology of the valley (e.g.,10,000 $m^2$), which is subsequently constrained by hydrologic and geologic processes at the watershed or basin scale. This hierarchical formation of the physical habitat at a site then determines which biological traits may persist at the site under the assumption that the biological community found at any particular scale is adapted to the mean state of the river at that scale (physical habitat variables, disturbance regime, etc.) and is filtered from a pool of potential species capable of colonizing the river.

Hierarchical stream theory has since been incorporated into more recent broad-scale concepts. The process domains concept (Montgomery 1999) builds on stream hierarchy by hypothesizing that across multiple spatiotemporal scales, random geomorphic processes underlying a river system are responsible for governing disturbance regimes, which in turn influence the abiotic habitat characteristics of specific reaches and structure the biotic communities. Broad scale geomorphic processes are defined to include climate, geology, and topography. These factors in turn influence runoff, substrate type, and stream gradient, and the effects of these factors then cascade into various smaller scale physical habitat factors in a hierarchical fashion. Thus, mechanistic factors underlying the trends postulated by stream hierarchy theory are developed. The fluvial landscape ecology concept (Poole 2002) builds on both the serial discontinuity concept and stream hierarchy theory by defining river system habitat as a patchy discontinuum determined by the hierarchical factors found at various scales in both the landscape and fluvial morphology. In a more comprehensive effort at explaining how rivers function, the riverine ecosystem synthesis (Thorp et al. 2006) groups areas of patches and their associated underlying hydrogeomorphology into functional process zones, which have been proposed for modeling changes in biotic community structure across river systems. These functional process zones can be a useful binning tool for dividing areas of riverine habitat into holistic management units which incorporate the geomorphology, topography, hydrology, and biota of an area.

### *Aquatic habitat and species decline*

Measures of aquatic environmental health, and subsequently biodiversity, have been found to negatively correlate with increasing human economic development in watersheds, placing increasing numbers of aquatic species at risk (Clausen and York 2008). Species restoration aims to return a particular species to an original state or enhanced condition that existed prior to degradation (Bradshaw 1996). Habitat assessment and management forms a basis for many aquatic species restoration programs (Bain and Stevenson 1999). Fish species that are considered habitat specialists are more likely to become imperiled, while invasive species often benefit by employing generalist approaches to their habitat requirements (Galat and Zweimüller 2001). These trends are likely to place preservation and restoration of aquatic habitat high on the list of priorities for conservationists and fisheries managers.

It appears that two specific life history traits make fish species more susceptible to concurrent population loss with habitat loss and degradation: specific habitat requirements for successful reproduction (e.g. substrate types), and dispersal. The loss of quality spawning habitat and migration routes for anadromous salmonid species has been linked to reductions in their populations in the Pacific Northwest (Gregory and Bisson 1997; Sheer and Steel 2006). Dam construction in the state of Maine from the 17th to the 19th centuries, and the subsequent loss of longitudinal stream connectivity and habitat has been implicated in the decline of River Herring (*Alosa* sp.) populations (Hall et al. 2011). Sturgeons have been negatively impacted globally by both the interruption of migration routes by the construction of dams as well as the degradation of spawning and nursery grounds by the alteration of flows by dam regulation or by remove substrate from river beds (Rochard et al. 1990). It has been argued that sturgeons , given their unique adaptations for life in heterogeneous large river systems, require increased evaluation efforts and more comprehensive designations of critical habitat for restoration and management of declining stocks (Beamesderfer and Farr 1997).

***Lake Sturgeon habitat suitability model***

Beginning in the early 1980's, agencies within the U.S. Department of the Interior began efforts to develop habitat suitability index models (HSM) which intended to predict the relative response of species (e.g. relative abundance) as a function of one or more quantitative habitat metrics (Schramberger et al. 1982). The original purpose of these models was to aid managers in evaluating the potential effects of removing, improving or mitigating habitat critical to species, with the added benefits of providing quantitative habitat descriptions rather than anecdotal or qualitative descriptions of species' habitat requirements. One such

HSM was generated for Lake Sturgeon to be used in the evaluation of new hydroelectric dam construction projects or modifications of existing projects on large, low-gradient rivers in the Canadian province of Ontario (Threader et al. 1998).

The overall HSM is comprised of two submodels which describe foraging habitat and spawning habitat (see Appendix 1). The foraging submodel is further comprised of habitat variables that are specific to either juvenile (i.e. not reproductively mature) or adult (i.e. reproductively mature) Lake Sturgeon. The authors defined eight habitat variables that determine the overall habitat suitability of an area for Lake Sturgeon. The foraging habitat submodel encompasses measurements of adult preferred substrate type, juvenile preferred substrate type, juvenile preferred foraging depth, and juvenile preferred foraging velocity. Good foraging habitat is essentially defined by what the authors believed to be the most productive for benthic macroinvertebrate communities; that is, non-spawning habitat is assumed to be defined by what produces the most ‘groceries’ for Lake Sturgeon to exploit. The reproduction submodel consists of measurements of temperature, velocity, substrate type, and depth of the area considered as potential spawning habitat. The authors established either categorical (in the case of substrate types) or continuous measurements of each of the eight variables, and assigned a score for each measurement level (ranging from a score of 1.0 indicating the highest suitability to a score of 0.0 indicating lowest suitability for a given level).

The Lake Sturgeon HSM was developed by reviewing literature describing Lake Sturgeon habitat use and by communication with managers working within the Moose River basin, Ontario, Canada (Threader et al. 1998). The authors used field data collected from four sites within that basin to validate the predictions of the HSM. They acknowledged that their Lake Sturgeon HSM may not be useful for generating predictions about Lake Sturgeon in rivers in other regions and subsequent efforts at verifying predictions developed with the HSM showed that at a coarse scale, the model provides some insight into where Lake Sturgeon in other systems may be aggregating, but only manages a low predictive power (Haxton et al. 2008). However, given that the development of a HSM for a species was motivated in large part to standardize habitat descriptions for that species, and develop more quantitative methods for describing and managing fish and wildlife habitat, in general the Lake Sturgeon HSM does provide a useful foundation for studying the quantity and quality of physical habitat available to Lake Sturgeon in the Upper Tennessee River system.

***Lake Sturgeon habitat descriptions***

In the Lake Sturgeon HSM, substrate descriptions were divided into seven categories: clay, silt, sand, gravel, rubble/cobble, boulder, and bedrock (Threader et al. 1998). These categories were defined by visual and tactile characteristics (clay, silt, and bedrock) or by measures of diameter (sand, gravel, rubble/cobble, and boulder). In the submodels, differing suitability scores were given to the various substrate types based on whether the habitat was being evaluated for foraging or reproductive suitability. For example, optimal foraging habitat for both adult and juvenile Lake Sturgeon was deemed to consist predominantly of finer substrate particles: silt, sand, and gravel. However, optimal substrate for spawning was said to be comprised of coarser particles (i.e. rubble/cobble and boulder). Descriptions of Lake Sturgeon habitat use have been varied in both their methods and their final conclusions, but there are several themes common to all of them that appear to support at least the broad predictions of the Lake Sturgeon HSM.

The close linkage between non-spawning habitat and foraging was noted in an early description of Lake Sturgeon habitat preferences: the Lake Sturgeon was described as a shallow-water fish, as the shallows of lakes were thought to be the only places with productivity high enough to support the benthic invertebrate communities necessary for fish capable of growing as large as Lake Sturgeon (Harkness and Dymond 1961). The authors noted that a few Lake Sturgeon were collected from samples in deeper water, but it does not appear that they considered deeper habitats very important overall for Lake Sturgeon. Harkness and Dymond (1961) had a similarly restricted description of the physical characteristics of Lake Sturgeon spawning habitat, describing Lake Sturgeon spawning in rapidly moving water at temperatures between 13 – 18$^{o}$C.

Physical habitat characteristics required by Lake Sturgeon vary depending on the life history stage (Kerr et al. 2010). Furthermore, as noted in an early introduction to habitat suitability models, descriptions of habitat use for fish and wildlife species are often generated using novel or varying techniques, depending on the species and system under study (Schramberger et al. 1982). Thus, modern studies providing descriptions of Lake Sturgeon habitat use can be characterized in two ways: first, by the life history stage of Lake Sturgeon under study (young-of-the-year, juvenile/sub-adult, adult, foraging, and/or spawning) and by the method used to generate information about the habitat, particularly when describing the substrate (visual, sieving, sonar).

Visual methods for describing substrate characteristics of Lake Sturgeon habitat often involve predefined substrate classes applied to either samples collected from river beds or by sending a submersible camera to the substrate. Substrate characteristics in habitat of age-0 Lake Sturgeon in the lower Peshtigo River, Wisconsin, were assessed using visual determination of dominant substrate type in samples collected with a petite Ponar dredge from areas where young-of-the-year Lake Sturgeon were collected (Benson et al. 2005). Both foraging and spawning habitat availability for adult Lake Sturgeon were described using the substrate classifications of Threader et al. (1998) in tributaries of Lake Michigan by either sampling with a wading pole or by collecting sediment with a petite Ponar dredge (Daugherty et al. 2008). Substrate descriptions derived from underwater video collected during a riverbed mapping study were used to describe the habitat in areas of core use by Lake Sturgeon sub-adults in the French Broad River, a tributary to the Tennessee River (Huddleston 2006). Underwater video was again used when the viability of the substrate definitions in

Threader et al. (1998) for predicting Lake Sturgeon abundance in another system was tested using substrate classified from video recorded with an Aqua-Vu submersible camera system (Haxton et al. 2008).

Studies that have described the substrate composition of Lake Sturgeon habitat by partitioning substrate samples into size classes using sieves are usually done with predetermined sieve sizes that correspond with substrate classes of silt/clay, sand, gravel, and then larger particle classes of cobble/rubble and boulder. Ponar grab samples of substrate from areas utilized by age-0 Lake Sturgeon in the Portage Lake system, Michigan, were sieved and found to be dominated by small gravel particles (Holtgren and Auer 2004). Sub-adult Lake Sturgeon in the Winnipeg River, Canada, were found to utilize deep (>13.7m) areas of either clay/silt or sand substrate, as determined by samples collected by Ponar grabs and sieved into clay/silt, sand, gravel/cobble, and a large-particle catch-all category for any areas where the Ponar sampler was unable to collect substrate (Barth et al. 2009). Those authors stressed the importance of sampling deep areas to capture juvenile Lake Sturgeon in monitoring efforts, which suggests deep habitat is important for juvenile Lake Sturgeon. Lord (2007) provided additional support for the use of deep (12-18 m) areas by juvenile Lake Sturgeon from telemetry data on nine fish in the St. Clair River at the boundary between Michigan and Ontario, Canada. In that system, areas with near complete coverage of Zebra Mussel (*Dreisenna polymorpha*) appeared to be avoided by Lake Sturgeon, that instead were found to associate with areas of high gravel content in the substrate. The author implied that the apparent preference for gravel habitat is due to increased foraging success in areas dominated by gravel substrate. Juvenile Lake Sturgeon in the Moose River system, Canada, displayed avoidance of impacted habitat, as well. They were found to congregate in areas near clay or sand habitat, which correlated with greater benthic macroinvertebrate abundance, and avoided areas of substrate dominated by wood chips which were a remnant effect of anthropogenic alterations to the system arising from wood-processing activities (Chiasson et al. 1997). The sediment samples collected by Chiasson et al. (1997) were sieved into clay, sand, gravel, and cobble and wood chips size classes. It is apparent that there is not an insignificant level of variety in the habitat preferences of different Lake Sturgeon populations in different systems, and that trophic productivity is often suggested as the underlying factor driving non-spawning habitat preferences. Preliminary results of habitat assessments of Lake Sturgeon in the Upper Tennessee River have found that clay/silt substrate comprised over 70% of sediment collected from areas of sub-adult Lake Sturgeon use during June and July 2014 (D.J. Walker, unpublished data, Appendix I & II). These assessments match the findings of many of the studies cited above, that found fine substrate particles to comprise a large proportion of Lake Sturgeon non-spawning habitat.

The studies of Lake Sturgeon habitat preferences reviewed to this point have focused primarily on their foraging habitat associations, where there is some disagreement in their findings. Assessments of Lake Sturgeon spawning habitat have displayed a much greater level of agreement in their conclusions. Sixteen years of observations of the spawning Lake Sturgeon population in the Lake Winnebago system, Wisconsin, were presented in the comprehensive description of Lake Sturgeon spawning behavior by Bruch and Binkowski (2002). Their findings, such as a spawning temperature range of 8 – 21$^{o}$C, supported some of the earliest conclusions about Lake Sturgeon spawning behavior (i.e. the reported spawning temperature range of Harkness and Dymond [1961] of 13.9 – 16$^{o}$C). Bruch and Binkowski (2002) reported temperature as the key physical habitat variable governing the onset and end of spawning, adding water velocity and substrate as important secondary variables. Lake Sturgeon are considered lithophilic spawners, requiring clean, stable substrate with interstitial spaces to deposit their adhesive fertilized eggs, and flowing disturbed water with its attendant elevated dissolved oxygen content to keep the eggs aerated. Descriptions of spawning habitat utilized by Lake Sturgeon fitting these characteristics come from the Detroit River and the Lake Huron-Lake Erie channel in the U.S., and from the Des Prairies and L'Assomption Rivers in Quebec, Canada (LaHaye et al. 1992; Manny and Kennedy 2002; Caswell et al. 2004). In each of these cases, evidence suggesting spawning activities by Lake Sturgeon came from the collection of fertilized eggs through either drift nets or by egg mat traps.

### *Population estimation and wildlife management*

The management of wildlife populations often includes or even prioritizes maximizing the population size of the specie(s) of interest (Williams et al. 2002). This task necessitates the accurate assessment of current population sizes and the impacts that changing environmental and anthropogenic selectors may have on future populations. Complicating matters is the fact that, in the case of wildlife and fisheries management, the individuals within the population of interest can be mobile, cryptic, or both. Wildlife and fisheries managers must utilize one of many models to provide estimates of population size and/or other parameters of interest (e.g., survival, immigration and emigration, recruitment, mortality) (Williams et al. 2002). Furthermore, sampling and subsequent parameter estimation may be required if managers lack sufficient resources to conduct censuses.

The field of fisheries science has a long history of relying on statistical models to estimate parameters, including population size, due to the cryptic and motile nature of fishes and the inherently reduced visibility in aquatic habitats. Due to the economic importance of commercial recreational fisheries, management decisions are based upon rigorous quantitative findings (Allen and Hightower 2010). In addition, it is necessary for fisheries managers to acknowledge and include in their conclusions the sources of bias that may be encountered when sampling fish populations, such as the size- and species-selection bias inherent in all sampling methods and the sources and extent of variation in estimation methods (Allen and Hightower 2010). However, with careful sampling design, methodology, and model selection, fisheries managers can use population estimation models to inform future management actions and harvest targets.

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# CHAPTER I
# MAPPING LAKE STURGEON SPAWNING HABITAT IN THE UPPER TENNESSEE RIVER USING SIDE SCAN SONAR

A version of this chapter was originally published by Daniel J. Walker and Dr. J. Brian Alford:

Walker, D.J. and J.B. Alford. 2016. Mapping Lake Sturgeon spawning habitat in the Upper Tennessee River using side scan sonar. North American Journal of Fisheries Management 36: 1097-1105.

I submitted the first draft of this manuscript after Dr. Alford's revisions to the journal North American Journal of Fisheries Management on 11 March 2016. The manuscript was accepted with major revision by the North American Journal of Fisheries Management on 10 May 2016. I completed the requested edits and sent in the revised edition on 25 May 2016. The manuscript was fully accepted for publication on 10 August 2016. I was the primary author and developed the objectives and survey methods used, and I completed all data analysis.

## ABSTRACT

The Lake Sturgeon *Acipenser fulvescens* is a fish species that was once dispersed widely throughout the Mississippi River drainage but was largely extirpated from the southern portions of its range by overfishing and habitat degradation. There is an ongoing restoration effort to reestablish the Lake Sturgeon to rivers of the Southeastern United States. Reintroduced juvenile Lake Sturgeon now occupy several reservoirs along the Upper Tennessee River that

are separated from each other by hydroelectric dams. To complete their life history, Lake Sturgeon will migrate upriver from reservoir habitats to more lotic habitats and spawn over coarse rocky substrate, even in the tailwaters of impassable dams. Using low-cost, consumer-grade side scan sonar and a geographic information system, I mapped the substrate of four tailwaters that may be future Lake Sturgeon spawning locations. I used video imagery collected from random locations within the mapped areas to validate my digitization of sonar imagery. I calculated the area of four substrate classes displayed in the maps to evaluate that aspect of the suitability of each of the tailwaters for Lake Sturgeon spawning. The revised maps show that the best spawning substrate (unembedded, coarse, rocky substrate 6 – 25 cm in diameter) comprised 17.0 – 30.5% of the total area mapped at each tailwater, while the least suitable substrate class (fine sediment <0.2 cm in diameter) comprised 6.2 – 30.7% of the mapped areas. My results suggest any future spawning events by Lake Sturgeon below each of these dams are likely to encounter some suitable spawning substrate patches, while management opportunities exist to supplement tailwater areas with suitable spawning substrate.

# INTRODUCTION

The Lake Sturgeon *Acipenser fulvescens* historically occurred in large rivers and lakes of the Mississippi River, Great Lakes, and Hudson Bay drainages (USA) (Harkness and Dymond 1961; Scott and Crossman 1973; Becker 1983; Etnier and Starnes 1993). The Lake Sturgeon is believed to be largely extirpated from the southern reaches of the Mississippi River, where numbers may have been low prior to anthropogenic alterations to the populations (Etnier and Starnes 1993; Williamson 2003). A multi-agency effort, which includes annual releases of age-0 Lake Sturgeon (minimum total length 15.24 cm) sourced from the Wolf River, Wisconsin, is ongoing to restore the Lake Sturgeon to its historic range in the Southeastern United States. Over 150,000 juvenile Lake Sturgeon have been released in rivers across the Southeast since 2000, with the majority of the fish reintroduced to the Upper Tennessee River (M. Cantrell, U.S. Fish and Wildlife Service, unpublished data). A key objective of this reintroduction effort is to facilitate the resurgence of successful natural spawning and recruitment of Lake Sturgeon in the Tennessee River.

Lake Sturgeon spawning migrations are largely triggered by rising springtime water temperatures (Bruch and Binkowski 2002). In many river systems occupied by Lake Sturgeon, the river is fractured by dams that are likely to be impassable by migrating Lake Sturgeon (Auer 1996a). When the reintroduced Tennessee River Lake Sturgeon reach sexual maturity, they will attempt spawning migrations upstream from the reservoirs. When this occurs, many of the fish will encounter small and large dams, including four large hydroelectric dams on the main channel of the Upper Tennessee River (Fort Loudoun, Watts Bar, Chickamauga, and Nickajack Dams). Some of the reproductively mature Lake Sturgeon may attempt to spawn in the tailwaters below these hydroelectric dams in a manner similar to Lake Sturgeon in other river systems (e.g., LaHaye et al. 1992; McKinley et al. 1998; Caswell et al. 2004).

In their habitat suitability model (HSM) for Lake Sturgeon, Threader et al. (1998) identify four habitat variables that contribute to spawning habitat suitability for this species: water temperature, water velocity, substrate, and depth. Of these four variables temperature, velocity, and depth will be governed largely by the hydroelectric management schedules at the large dams and recent river flows and environmental factors at the small dams at the time of the future migrations. Indeed, Lake Sturgeon spawning effectiveness and recruitment has been positively impacted by alterations to flow management regimes in other systems (e.g., Auer 1996b). The remaining variable is substrate. Artificial spawning reefs have been constructed by hydroelectric producers and fisheries managers to augment Lake Sturgeon spawning events below hydroelectric dams in other systems (Johnson et al. 2006; Dumont et al. 2011; Bouckaert et al. 2014). In light of this, I set out to document the type and areas of substrate in the tailwaters directly below the four Upper Tennessee River hydroelectric dams. To assess the suitability of these four tailwaters for Lake Sturgeon spawning, I collected and processed side scan sonar imagery of the riverbed using a consumer-grade fish finder unit (Kaeser and Litts 2010). I used sonar imagery, reference video imagery, and their associated global positioning system (GPS) coordinates in a geographic information system to create maps of the substrate in the tailwaters. My objectives were to 1) classify and score the substrate found in the tailwaters using the Lake Sturgeon HSM (Threader et al. 1998), and 2) to estimate the total area of each substrate class at each dam. This information will serve as a baseline assessment of the suitability of the substrate in these tailwaters for future Lake Sturgeon spawning events.

## METHODS

### *Study sites*

I conducted sonar surveys of the tailwaters immediately downstream of the four upstream-most dams on the mainstem Tennessee River, listed here in order from upstream to downstream: Fort Loudoun Dam, Watts Bar Dam, Chickamauga Dam, and Nickajack Dam (Figure 1). For the purposes of this study, I refer to the tailwater sites by the name of the dam immediately upstream, although the site is actually a part of the next reservoir downstream (e.g., what I refer to as the Fort Loudoun tailwater is a part of Watts Bar reservoir, etc.). Fort Loudoun Dam is located on the Tennessee River in Loudoun County, Tennessee (35.791$^{o}$ N, 84.243$^{o}$ W). The dam was completed in 1943, and contains four hydroelectric generating units with a combined capacity of 162 MW. The dam measures 37 m tall by 1277 m wide. Watts Bar Dam is located at the boundary between Meigs and Rhea Counties, Tennessee (35.621° N, 84.782° W). Watts Bar dam was completed in 1943, and contains 5 hydroelectric generating units with a combined capacity of 182 MW. Watts Bar Dam is 34 m tall and 902 m wide. Chickamauga Dam is located in Hamilton County, Tennessee (35.105° N, 85.229° W). Chickamauga Dam was completed in 1940, and houses four hydroelectric generating units with a combined capacity of 199 MW. The dam is 39 m tall by 1767 m wide. The descriptive information for each of the dams is available at the Tennessee Valley Authority's website (available online at https://www.tva.gov/Energy/Our-Power-System/Hydroelectric/).

Each survey consisted of parallel downstream transects using a total sonar beam width of 76.2 m. Each transect began as close to the dam as conditions allowed, and continued downstream for approximately two river kilometers (RKM).My sonar surveys of each tailwater were completed between 10 May 2015 and 26 May 2015, when flows had subsided from the higher spring releases.

***Sonar imagery collection***

I utilized the sonar imagery collection and geoprocessing procedure developed by Kaeser and Litts (2008; 2010) and Kaeser et al. (2013) with some modification. I used the GPS data from the fish finder unit as this streamlined the data collection process after preliminary tests confirmed its accuracy when compared to GPS data collected at the same test locations with a handheld GPS unit. I conducted all of the surveys in a 4.62 m aluminum johnboat with a 60 hp outboard jet motor. I used a custom-built, adjustable aluminum arm to mount the sonar transducer in the bow of the boat, where the sonar imagery would not be affected by propeller wash (Figure 2). As the GPS data are collected from the sonar screen unit and not the sonar transducer, all of the final sonar imagery products are displayed approximately 4 m upstream of their true physical location. A discrepancy at this small scale is acceptable given the coarse mapping resolution and large areas mapped.

### *Sonar data processing*

To process the individual sonar images into mosaics for each transect, I first batch-clipped the sonar imagery using the program IrfanView (Irfan Skilijan 2015) to remove the extraneous collar saved with the sonar imagery when captured. I then uploaded the waypoints associated with each of the image captures to ArcMap 10.0 (ESRI, Redlands, CA). I used the ‘sonar tools’ toolbox in ArcGIS 10.0 (available for download online at http://www.fws.gov/panamacity/sonartools.html) to process the raw sonar images into georeferenced sonar image mosaics. I processed each transect individually and saved the spatially-explicit georeferenced sonar image mosaics for each transect as individual raster layers for display, adjusting for improved clarity and the digitization process.

***Ground data collection and processing***

I loaded the relevant georeferenced sonar imagery raster layers into ArcMap over a National Agriculture Imagery Program (USDA 2014) orthoimage of each tailwater (natural color, 1 m ground sample distance, 6 m horizontal accuracy). I digitized a polygon that bounded all of the mapped substrate area bounded by the riverbank displayed in the NAIP image at the raster resolution scale (1:939), and then used the random point generator tool in ArcMap to randomly generate 50 points within the polygon outlining the area mapped (Congalton and Plourde 2002). I chose a sample size of 50 reference points in light of the logistical requirements of revisiting the sites and the time required for operating the underwater camera system effectively. I set a buffer of 20 m radius around each point to reduce overlap among the points and ensure I could collect reference data at each point from a boat that was likely to be moving continuously during ground data collection. I converted the location data of each point at each tailwater from UTM to GPS coordinates, and revisited each tailwater to collect reference ground data of the substrate.

***Substrate classification and assessment***

I began with an initial classification scheme that contained 10 classes of substrate (Table 1). I defined the classes such that if I were unable to generate sonar image maps of sufficient resolution to accurately interpret the various classes from the sonar imagery, I could collapse the original substrate classes into fewer more broadly defined classifications. I conducted analog image interpretation and digitization of the various substrate classes listed in Table 1 (Narumalani et al. 2002). I conducted all of the digitization at the raster resolution scale (1:939). I employed a holistic decision-making process to classify the substrate patches based on the intensity of the sonar reflection (brighter images indicating harder substrate) and texture of nearby sonar imagery. While I attempted more rigorous automated classification techniques (unsupervised and supervised), the sonar imagery produced with this method does not contain the necessary data for the automated classification tools to perform.

I assigned the original ten substrate classes used in the digitization and video image classification scores of 1 – 4 based on the scoring in the Lake Sturgeon HSM, to contribute biological relevance to the substrate classes and simplify validation. Once I had completed digitizing patches of substrate following the classification scheme in Table 1, I overlaid the waypoints and associated substrate classifications of the ground data reference points. I calculated an accuracy assessment of the first substrate maps by generating simple error matrices which compared the classification of the substrate below each reference point from the sonar image digitization to the substrate classification assigned from the video reference imagery. In response to low accuracy rates, I created second editions of the substrate maps using four more broadly defined substrate classes and scores (Table 2). I reclassified the substrate observed in the ground data video imagery into the four classes of substrate from the HSM and then overlaid the ground data on the georeferenced sonar image mosaics. I then completed a second analog digitization of the substrate using both the sonar image mosaics and the reclassified video imagery. I used the sonar imagery as a guide to identify boundaries among patches of the four substrate classes. As I used both the reference ground data and the sonar imagery in creating the second edition maps, I did not calculate a second error matrix. All of the ground data points were contained in polygons of their respective substrate type.

## RESULTS

### *First edition substrate maps*

The substrate maps I generated using the first classification scheme are shown in Figure 3. The first edition maps indicated fine substrate particles (< 0.2 mm diameter; shown on each map in beige) were the predominant substrate at each of the dams. I observed that bedrock was present immediately below each of the dams. My overall accuracy ranged from 29% to 33% for the first digitization of the substrate using the initial classifications (Table 3). Given this high rate of error, I do not report areal measurements of the substrate classes used in these maps here.

### *Second edition substrate maps*

The second edition of the substrate maps showed similar patterns to what I observed in the first edition maps: at the base of the dam, there was an area of bedrock, and towards the downstream end of the mapped areas there appeared to be an increase in the finer sediment classes (Figure 4). The total areas ($m^2$) of each of the four substrate types at each of the dams are displayed in Table 4. The area of each substrate type as a percent of the total area mapped at each of the dams is shown in Figure 5. Of the four substrate types indicated on the map, bedrock (embedded particles >25 cm in diameter, displayed in yellow) was the predominant substrate in the Fort Loudoun tailwater, comprising 44.3% of the total area mapped. Gravel (particles 0.2 – 6 cm in diameter, displayed in light green) was the predominant substrate type below Watts Bar Dam, and comprised 67.0% of the substrate. Of the four tailwaters mapped, Chickamauga Dam exhibited the greatest area of cobble-boulder substrate (the optimal Lake Sturgeon spawning substrate, 6 – 25 cm diameter, indicated by dark green) as a percentage of the total area mapped at 30.5%. However, there was a more even distribution of each of the four substrate types in the tailwater below Chickamauga Dam, and gravel substrate covered 29.6% of the area mapped. Gravel was the predominant substrate type and covered 35.6% of the tailwater below Nickajack Dam, which also exhibited the greatest coverage of fine particles (< 0.2 cm diameter, displayed in red) at 30.7%. When I visually asses the trends displayed in Figure 5, I note that there is an increasing trend in the total area of the best spawning substrate (cobble-boulder) between Fort Loudoun, the upstream tailwater, and Chickamauga and Nickajack, the downstream tailwaters, even when total width of the river is taken into consideration.

# DISCUSSION

The overall goal of this project was to describe the distribution of four size classes of substrate in four TVA tailwaters to assess the suitability of these areas for future Lake Sturgeon spawning events. Subsequently, my assessment will aid future management attempts to identify tailwater areas for artificial spawning reef installation. After my first attempt at interpreting the sonar imagery, my initial accuracy rates (29-33%) were inadequate. Future research using the techniques I have detailed here will benefit from revising the collection of the ground imagery. A stratified random sampling design (e.g. balance-acceptance sampling, Roberston et al. 2013) which avoids the issue of 'clumping' reference locations and can account for differing areal measurements of the various substrate classes in addition to a larger sample size of reference points will greatly improve the accuracy of future mapping studies. I attribute my low initial accuracy to differences between the resolution of the imagery I collected and the resolution necessary to utilize my initial, fine scale classification scheme. As my initial accuracy measurements were unacceptable, I revised my technique by including the video imagery in the second digitization procedure to improve my confidence in the results at a cost of consuming the reference data in map generation without reserving additional reference data to assess the accuracy of the second edition maps. This is why I do not report accuracy measures such as the results of additional error matrices. Using this hybrid approach, I improved my ability to describe the available substrates among Upper Tennessee River tailwater environments, while simultaneously streamlining my assessment of suitable spawning habitat for Lake Sturgeon.

As I have generated a census of the available substrate at these dams, I did not require statistical testing to interpolate results. I noted that cobble-boulder substrate area was greater in the tailwaters of the two most downstream dams, Chickamauga and Nickajack. Annual resampling efforts have found that larger, older Lake Sturgeon appear to inhabit the reservoirs below Chickamauga and Nickajack Dams relative to the reservoirs downstream of Watts Bar and Fort Loudoun Dams (M. Cantrell, U.S. Fish and Wildlife Service, unpublished data). This is likely an artifact of the reintroduction process, as the majority of Lake Sturgeon have been reintroduced into Fort Loudoun reservoir near Knoxville, TN, upstream of Fort Loudoun Dam. I think that the Lake Sturgeon have moved downstream from the reintroduction location so that the fish that have made it the farthest from the reintroduction point (i.e., to Nickajack and Guntersville Reservoirs, downstream of Chickamauga and Nickajack Dams, respectively) are likely to be the oldest fish. As older fish are typically larger, these Lake Sturgeon are also the ones likely to reach reproductive maturity and attempt spawning first (Becker 1984). My results suggest that if that scenario became reality, the Lake Sturgeon that aggregated in the tailwaters below Chickamauga and Nickajack Dams would encounter the greatest areas of high quality spawning substrate. The conditions I have presented in my maps here suggest that those first early spawning attempts by Lake Sturgeon in the Tennessee River would be supported by the relatively greater availability of suitable spawning substrate in the tailwaters of those two dams.

To date, spawning by reintroduced Lake Sturgeon in the Tennessee River has not been documented. Once aggregations of reproductively mature Lake Sturgeon have been found, management actions can be taken to further augment successful reproduction. The construction of artificial spawning reefs, a management tool that has been used with success to augment Lake Sturgeon spawning in other systems, may be useful in the support of natural Lake Sturgeon recruitment to the Tennessee River (LaHaye et al. 1992; Johnson et al. 2006; Roseman et al. 2011; Bouckaert et al. 2014; McLean et al. 2015). Artificial reefs can be developed in areas where reproductively mature Lake Sturgeon aggregate and the relevant water conditions are suitable for spawning. As I did not find dramatic differences at a coarse scale in the overall area of optimal spawning substrate among the dam tailwaters I surveyed, I recommend continued monitoring of these tailwaters and other potential migration barriers in the Tennessee River system for the presence of Lake Sturgeon when water conditions are suitable for spawning. Once an area has been found to support spawning Lake Sturgeon, further management actions, such as mapping substrate at finer resolutions and constructing artificial reefs can then be undertaken. Future high-resolution substrate mapping efforts will also benefit from assessing seasonal differences in the distribution of substrate in response to dam management schedules which may be a confounding factor in substrate surveys of tailwaters. The data I have provided here represent a baseline assessment of the substrate across these tailwaters where future Lake Sturgeon spawning events may occur. Water velocity, temperature, and depth all play critical roles in governing Lake Sturgeon spawning and these factors should be considered in dam management schedules, providing another avenue of support for future Lake Sturgeon recruitment.

## ACKNOWLEDGEMENTS

The information, data, or work presented herein was funded in part by the Office of Energy Efficiency and Renewable Energy (EERE), U.S. Department of Energy, under Award Number DE-EE0002668 and the Hydro Research Foundation.

States Government or any agency thereof. The views and opinions of authors expressed herein do not necessarily state or reflect those of the United States Government or any agency thereof.

I would like to acknowledge the valuable technical assistance I received from Dr. Adam Kaeser, and I deeply appreciate his free publication of training materials pertaining to the procedures I used. I would also like to acknowledge the constructive input I received on this project from the Southeastern Lake Sturgeon Working Group. Financial support for this project came from the Hydro Research Foundation graduate student fellowship and the Organization of Fish and Wildlife Information Managers student scholarship awarded to D.J. Walker. I greatly appreciate the members of the Fisheries Research Laboratory at the University of Tennessee Institute of Agriculture for their assistance in collection of the data, especially Todd Amacker, Kenneth McMahan, Hayley Gotwald, Keith Garner, and Justin Wolbert.

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# APPENDIX A

***Tables***

***Table 1.1***

Initial classification scheme used when digitizing substrate patches in the first edition substrate maps and video reference imagery.

| Substrate | Characterization | Spawning Habitat Score |
|---|---|---|
| Bedrock | > 75% exposed bedrock | 2 |
| Mixed Rocky | ≤ 50% coarse + fine matrix | 3 |
| Rocky Coarse | Discernible individual particles > 25 cm diameter | 4 |
| Rocky Fine | Particles 25 > x > 1 cm diameter | 4 |
| Riprap | Artificially placed bank stabilizing rock | 4 |
| Fine | > 75% sand, silt, clay particles ≤2 mm | 1 |
| Biological | Algae, aquatic macrophytes, zebra mussel reefs | 1 |
| Anthropogenic | Anthropogenic substrate, not riprap (e.g. concrete) | 1 |
| No Data/Sonar Shadow | No sonar image data | |
| No Data - Dam | No image at beginning of transect | |

***Table 1.2***

Final substrate classification scheme.

| Particle | Size (cm diameter) | Score |
|---|---|---|
| Cobble-Boulder | 6 – 25 | Highest |
| Gravel | 0.2 – 6 | |
| Bedrock | >25 | |
| Fine | <0.2 | Lowest |

*Table 1.3*

Accuracy of first edition substrate maps for each of the four dams mapped as the percent agreement between digitized substrate patches and reference imagery.

| Dam | Overall Accuracy |
|---|---|
| Fort Loudoun | 33% |
| Watts Bar | 24% |
| Chickamauga | 33% |
| Nickajack | 33% |

***Table 1.4***

Total area ($m^2$) of each substrate type at each dam calculated from the second edition substrate maps.

| | Cobble-Boulder | Gravel | Bedrock | Fine | Total |
|---|---|---|---|---|---|
| Fort Loudoun | 94429.2 | 101673.7 | 246252.6 | 113608.1 | 555963.6 |
| Watts Bar | 163131.5 | 605106.2 | 78464.6 | 55861.6 | 902563.9 |
| Chickamauga | 216603.5 | 210083 | 119593.5 | 164631.6 | 710911.6 |
| Nickajack | 220524.2 | 303475.1 | 66797.9 | 261346.3 | 852143.5 |

# APPENDIX B

## *Figures*

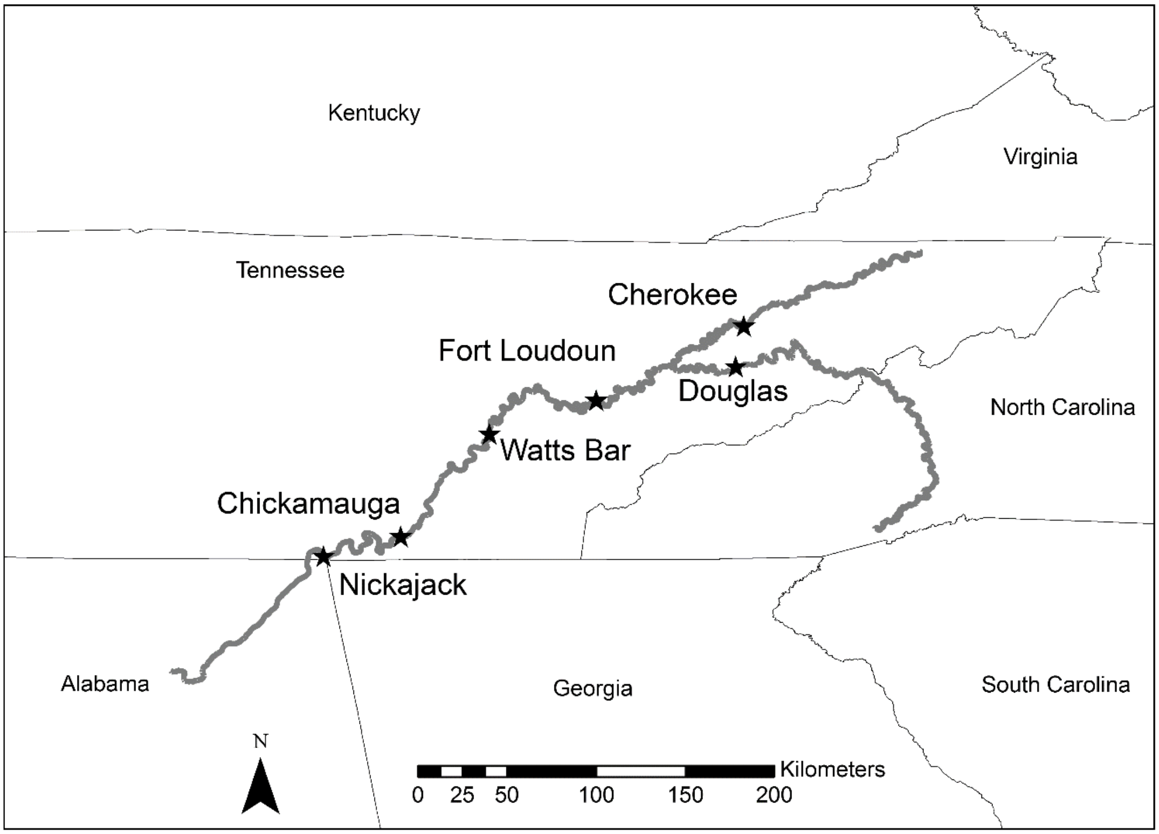


***Figure 1.1***

Map showing location of TVA hydroelectric dams on the Upper Tennessee, French Broad, and Holston Rivers. The four dams where I conducted sonar surveys are Fort Loudoun, Watts Bar, Chickamauga, and Nickajack dams.

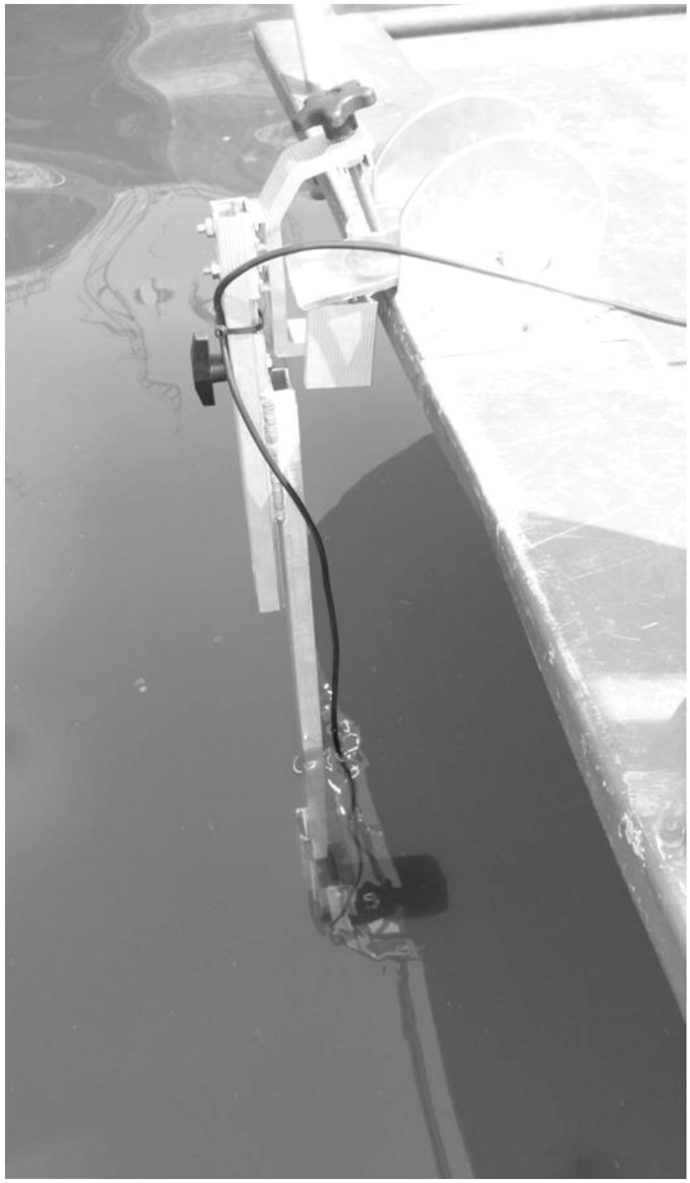

***Figure 1.2***

The bow-mounted sonar transducer arm. The transducer is removable, and the arm is adjustable for depth as well as capable of being raised out of the water for travel at speed. Photo credit: Todd Amacker.

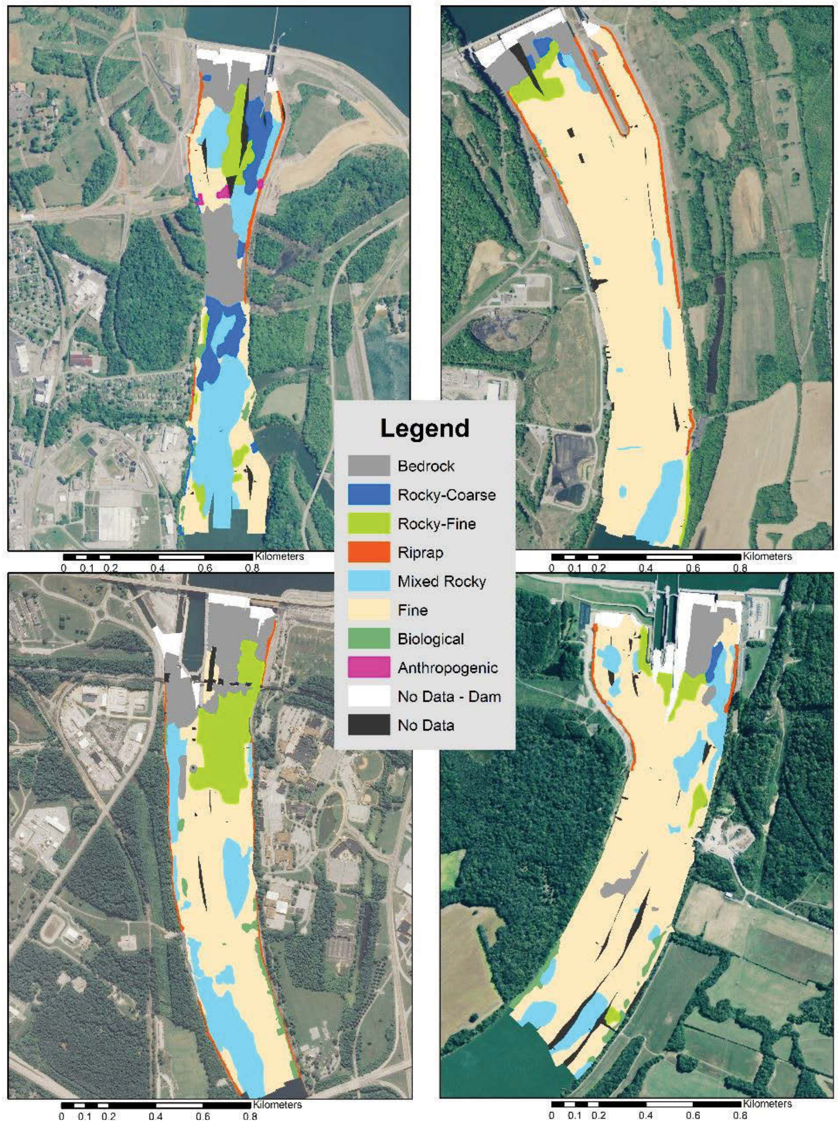


*Figure 1.3*

First edition substrate maps. I digitized each map by hand at the raster resolution using the classification scheme outlined in Table 1. Dams are shown clockwise from top left: Fort Loudoun dam, Watts Bar dam, Chickamauga dam, Nickajack dam. All four maps are displayed at 1:17000 scale, and the maps are oriented so that the upstream portion of the tailwater is at the top of the image.

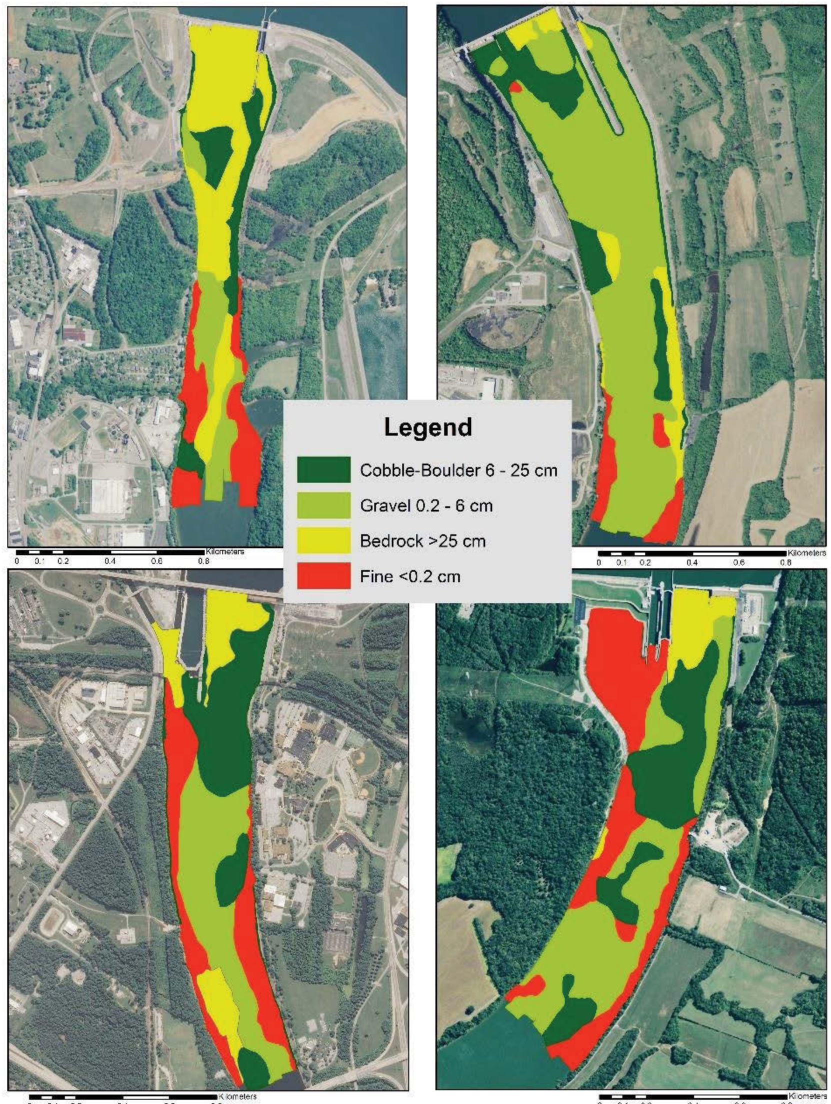


***Figure 1.4***

Second edition substrate maps. The classification scheme used in the digitization of these maps is detailed in Table 2. Dams are shown clockwise from top left: Fort Loudoun dam, Watts Bar dam, Chickamauga dam, Nickajack dam. All four maps displayed at 1:17000 scale, and the maps are oriented so that the upstream portion of the tailwater is at the top of the image.

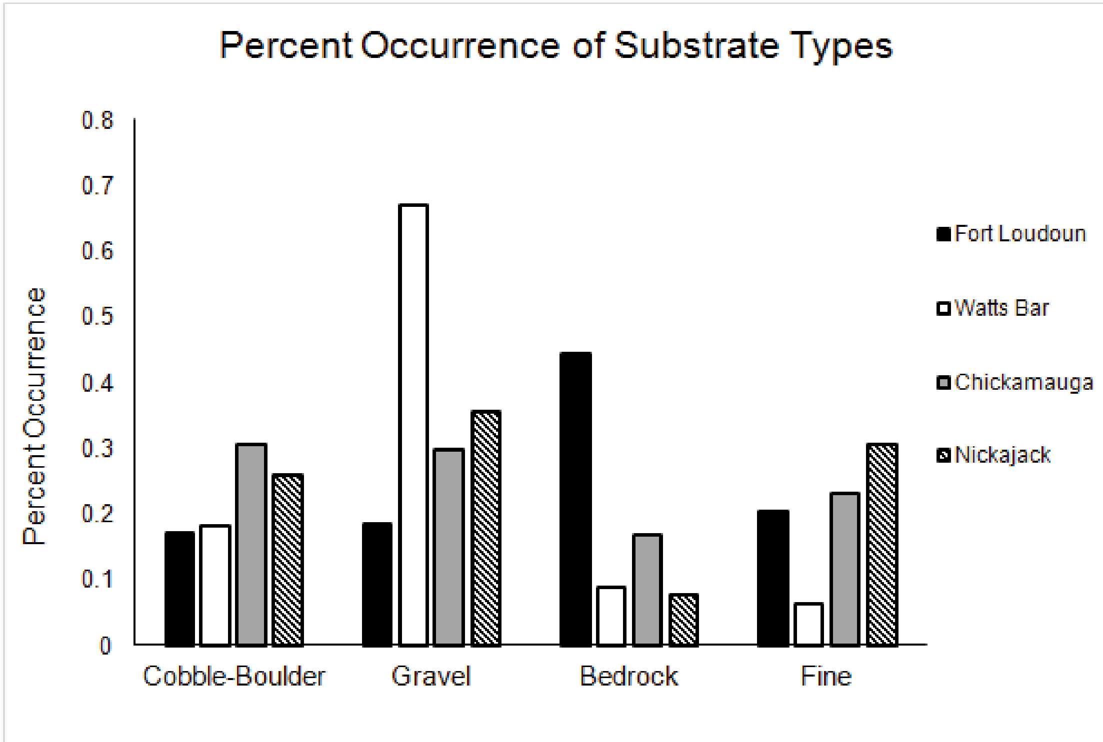


***Figure 1.5***

Areal measurements of the various substrate classes identified in the second edition maps as a percentage of the total area of the tailwater mapped.

# CHAPTER II
# CHARACTERIZING LAKE STURGEON SUMMER REFUGIA IN THE UPPER TENNESSEE RIVER

## ABSTRACT

The Lake Sturgeon reintroduction to the Tennessee River system was undertaken to reestablish a population near the southern extent of the historical range of the species. Lake Sturgeon, like the rest of their acipenserid relatives, have a complex life cycle which includes migrations to and from areas of specific physical habitat characteristics during specific portions of the cycle. Examples documented in Lake Sturgeon populations from other river systems include aggregations in river reaches that are approximately > 10-100 km long during summer thermal maxima. In this study, I confirmed the location of a suspected summer holding area important to the Lake Sturgeon reintroduced to the Upper Tennessee River by relocating acoustically-tagged individuals at a density of >3 sturgeon per river kilometer. I measured five habitat variables (temperature, depth, dissolved oxygen, conductivity, and substrate) from both Lake Sturgeon locations as well as randomly selected comparison locations. These measurements revealed that at the Lake Sturgeon locations I sampled, the fish appeared to be occupying habitat similar to locations I measured at random throughout the holding area. The similarity between the measurements from the Lake Sturgeon locations and the surrounding area precluded the use of statistical classification techniques. However, future research will be able to build on the data I have provided here to generate rigorous classifiers capable of predicting other areas of important Lake Sturgeon habitat throughout the Upper Tennessee River.

# INTRODUCTION

In the guiding document of the reintroduction of Lake Sturgeon (*Acipenser fulvescens*) to the Tennessee and Cumberland Rivers, the Southeastern Lake Sturgeon Working Group (SLSWG 2015) detailed a series of management and research objectives that remain to be addressed to aid and improve the restoration of the species. First among these research needs is to assess habitat in the river system available for the species to utilize. The current habitat suitability model for Lake Sturgeon categorizes their habitat requirements by their physiological needs during two life history stages: foraging habitat and spawning habitat (Threader et al. 1998). The suitable spawning habitat parameters were determined to maximize successful fertilization of eggs and development of larvae, and are thus relevant on a yearly basis to the species during the springtime spawning season. Foraging habitat is occupied by this species after spring spawning, and many fish likely remain in their foraging habitats until they transition to their fall-winter staging areas immediately preceding the next year's spawning migration, if they undertake a sequential migration at all.

In the previous chapter, I investigated the quality and quantity of potential future spawning ground habitats, so my next research goal was to quantify the summer foraging habitat utilized by Lake Sturgeon in the Upper Tennessee River. This research objective is significant because it describes habitat use by Lake Sturgeon during the stressful period of summer during which Lake Sturgeon realize their thermal maxima. The Tennessee River is located near the southern extent of the species' historical range, so it has been hypothesized that the Lake Sturgeon reintroduced to the warmer climates of the Southeastern U.S. may endure temperature-related stress during the summer thermal maxima. Therefore, it is important to thoroughly document the conditions in which these fish spend the summers so that other reaches can be assessed for summer habitat suitability, which will facilitate protection of those areas found to be important to the persistence of the species during stressful periods of the year.

In my first study, I conducted a habitat inventory so that locating the fish during the period of interest was not necessary. For my research objectives for the current study, I had to first locate individual Lake Sturgeon, and then record the relevant habitat characteristics used by the fish upon detection in addition to randomized locations for comparison data. To locate the Lake Sturgeon in their summer habitat, I re-located fish that had been implanted with acoustic tags (Vemco, Bedford, NS) during a prior study (Saidak 2015). Once I had confirmed a location as potentially critical summer habitat based upon the density of tagged-fish present, I measured a variety of habitat variables from both the locations where Lake Sturgeon were detected and from randomly selected locations that were used as comparison points. I used interpolation to generate spatially-explicit raster layers of the habitat variables of interest, and then random resampling within the potential area of critical habitat to increase my overall sample size. I then evaluated the measures of the five variables from Lake Sturgeon-present and randomly-located points to 1) determine whether Lake

Sturgeon are exhibiting habitat selectivity, and 2) to identify which, if any, of the habitat variables I measured accurately delineate Lake Sturgeon habitat from the surrounding areas via statistical classification.

## METHODS

### ***Acoustic tracking***

A previous study involving Lake Sturgeon of the Tennessee River (Saidak 2015) had successfully implanted 26 Lake Sturgeon captured during the annual trot-line monitoring efforts on Fort Loudoun and Watts Bar Reservoirs with Vemco acoustic transmitters during the fall, winter and spring of 2013-2014. This process required anesthetizing the fish before making a surgical incision in the ventral surface of the abdomen. Next, sterilized acoustic transmitters were inserted into the body cavity of the fish, and the incision was sutured. The models of tag used for each fish (v13 (diameter = 13 mm), or v16 (diameter = 16 mm), both 69 kHz transmission) was determined by the size of the fish being tagged, with smaller fish receiving the smaller v13 tags and *vice versa* to ensure that the ratio of tag weight to fish weight never exceeded the recommended value of 2%. The Lake Sturgeon were allowed sufficient recovery time in oxygenated holding tanks, and were released after strong swimming behavior was observed. Subsequently, a combination of passive and active tracking of the tagged fish during the summer of 2014 (Saidak 2015; Walker, Appendix F) detected all acoustically-tagged fish at least once.

During the current study, I first re-located acoustically tagged Lake Sturgeon inhabiting Fort Loudoun and Watts Bar reservoirs by surveying the reservoirs with a hydrophone (VR100, Vemco, Bedford, NS). I completed the surveys between May 31 – July 31, 2016. I deployed the omni-directional hydrophone off the starboard side of an 8-m aluminum boat submerged to a depth of 0.1 m using a fixed deployment arm. I conducted all surveys in the downstream direction at a speed of 5 kph, and followed the thalweg of the navigational channel until the hydrophone detected transmissions from a tag.

Once the hydrophone detected a tag signal of sufficient strength to be decoded, I piloted the boat in a zig zag manner to locate the tag relative to my previous position in the river, and as the strength of signal increased I piloted the boat in a circular manner to triangulate the signal source. I considered increased signal strength (as viewed on the hydrophone readout) to be an indicator of increased proximity to the tag. I recorded as the location of the tagged Lake Sturgeon the point of greatest signal return, took water chemistry measurements as outlined below, and then resumed my downstream direction of survey.

***Identification of summer holding area reach***

To identify where Lake Sturgeon were aggregating during the summer months, I plotted all the GPS coordinates of the points of greatest signal return in the GIS program ArcMap 10.4.1 (ESRI, Redlands, CA). I then identified a cluster of 8 Lake Sturgeon detections within a reach length of 2.7 river kilometers (rkm) and an area of 134 ha on Watts Bar Reservoir. This translates to a density of nearly 3 Lake Sturgeon individuals per rkm (Figure 1). I did not encounter any other densities of similar magnitude over the 148 rkm I surveyed with the hydrophone between both reservoirs, so I determined this to be an area of potentially critical summer refugia (summer holding area or SHA). Thus, I focused my habitat characterizations in this reach.

***Habitat variables***

After locating the tagged fish, I recorded depth (m), temperature ($^{o}$C), dissolved oxygen (mg/L), and specific conductivity (µS/m) using a YSI 6600 data sonde (YSI Environmental, Yellow Springs, OH). The data sonde was connected to the handheld readout unit with a cable 100-m long, which was greater than any depth encountered during the survey. I deployed the data sonde to the maximum depth in the location, then retrieved it via the cable for a distance of 1 m to suspend the data sonde above the sediment layer, and recorded measurements after the readings had stabilized (approximately 30 s later).

After recording Lake Sturgeon location and water chemistry measurements, I plotted the data in ArcMap. Once I had determined a SHA area of interest, I utilized the balance-acceptance sampling (BAS) stratified-random sampling design algorithm in the R package *SDraw* to plot random sample locations in the immediate area from which to collect comparison habitat measurements (McDonald 2016; R Core Team 2016). To determine my stratified-sampling sample size, I performed a power calculation using the mean dissolved oxygen measurement (which I suspected to be a critical factor determining suitable habitat from other areas) from the Lake Sturgeon locations and a power of $\alpha = 0.05$. The indicated sample size was 52, which I plotted within a reach which was larger than the final SHA reach. The total number of comparison locations from which I took actual measurements within the SHA was 24 (Appendix G).

In addition to collecting measurements on habitat water chemistry parameters from both Lake Sturgeon locations and randomly selected reference areas, I also surveyed the SHA reach for substrate type using a BioSonics DT-X portable scientific echosounder unit (BioSonics, Inc., Seattle, WA). I configured the boat-mounted sonar unit to ping every 0.1 sec, and used the software Visual Habitat (BioSonics, Inc., Seattle, WA) to categorize the substrate detected in each ping using principal components analysis (PCA). The input data for the PCA was the signal return strength received by the sonar after each ping, where harder substrates correspond to greater signal return strength. I classified the substrate into three substrate classes corresponding to rocky (substrate = 1), sandy (substrate = 2), and silt-clay (substrate = 3) substrate based upon prior substrate information I gathered from a preliminary assessment of substrate using benthic grab sampling in the SHA area (D. Walker, Appendix H).

Next, I generated separate, overlapping interpolated raster layers of the 5 habitat variables of interest (temperature, depth, dissolved oxygen, conductivity, and substrate type). I performed this procedure using the 24 reference locations (source data for the temperature, dissolved oxygen, and specific conductivity variables) (see Appendix I through L for rasters) and the approximately 10,000 observations gathered with the echosounder for the depth and substrate variables (see Appendix M for distribution of echosounder pings). I used inverse-distance weighting to interpolate values for each of the five variables among the 24 reference points and in between the areas not covered with the echosounder transects. Once I had 5 overlapping raster layers covering the SHA, I then overlaid 100 randomly located points on the 5 layers, again using the BAS stratified random sampling algorithm. I removed points that did not overlap all 5 interpolated raster layers, and then extracted values for each of the remaining new points from the 5 underlying rasters. I performed a similar extraction using the 8 Lake Sturgeon locations, adding the interpolated substrate and depth measurements to the temperature, dissolved oxygen, and conductivity measurements collected in the field. The resulting data set consisted of the 5 predictor variables and the dependent variable determined by Lake Sturgeon presence ($n = 8$) or absence ($n = 81$). To assess any potential correlation among the 5 predictor variables, I generated a correlation plot testing the relationships among each of the 5 variables illustrated in a circle plot.

***Statistical analysis***

In order to achieve my first research objective, I first plotted the distribution of the five habitat variable measurements taken from Lake Sturgeon-present and -absent locations in boxplots. I visually evaluated whether the Lake Sturgeon were exhibiting habitat selectivity across any of the five variables by comparing the distribution of each of the five variables against the reference data. This would be indicated by divergent ranges and/or medians of the measurements taken at the eight Lake Sturgeon-present locations.

If the result of my visual analysis of the distribution of habitat variable values suggested that the Lake Sturgeon were exhibiting habitat selectivity, my next analytical procedure would be to evaluate the data for potential statistical classifiers. The results of the correlation assessment (referenced previously) indicated strong correlations among the habitat variables temperature, depth, and substrate, thus precluding parametric statistical classifiers. In the event of habitat selectivity, I would use non-parametric classification techniques such as logistic regression and random forest.

# RESULTS

### *Acoustic tracking*

Over the course of the surveys, I detected 25 Lake Sturgeon tagged during the previous telemetry study (Figure 1). The individual fish which I detected were all originally caught and tagged from the two reservoirs I surveyed, suggesting no new fish had moved upstream beyond Watts Bar Dam in the 2 years between surveys. I encountered only three tagged Lake Sturgeon in Fort Loudoun Reservoir, and the remainder were in Watts Bar Reservoir. The general locations where I detected the signals were the same as the areas of greatest annual Lake Sturgeon captures in the previous 5 years of trotline sampling as well, suggesting that these locations in the upper-to-mid lengths of each tailwater support suitable non-spawning habitat for this species, nearly year-round. As I encountered only 3 Lake Sturgeon in Fort Loudoun Reservoir, the following results are derived from measurements taken of the 8 fish in the SHA identified in Watts Bar Reservoir.

***Habitat variable measurements***

The correlation matrix among the five predictor variables indicates that there is some negative (Pearson) correlations within the data (Figure 2). Of these correlations, the temperature and depth variables exhibit the strongest negative correlation (Pearson's correlation = 0.6), while a weaker negative correlation is present between substrate and temperature, likely an artifact of the separate relationships between depth and substrate and depth and temperature. These correlations precluded the use of parametric statistical analyses. The average (standard deviation) water chemistry measurements associated with the Lake Sturgeon locations within the SHA (n = 8) were: 7.20 (2.34) mg/L dissolved oxygen, 25.94 (1.49) $^{o}$C temperature, 199.13 (9.61) µS/m specific conductivity, and 12.39 (5.24) m depth. When compared to the reference measurements extracted from the interpolated raster layers (n = 81), it appears that of the water chemistry variables, the Lake Sturgeon were distributed over a narrower range of values for all variables except dissolved oxygen (Figure 3). General trends in the habitat occupied by Lake Sturgeon in the SHA include a much greater range but similar median of dissolved oxygen values than the surrounding area, a narrower range, but greater than median, temperature and depth than surrounding locations, and a narrower and lower range of conductivity than surrounding locations. For the substrate variable, there does not appear to be an appreciable trend in the substrate occupied by Lake Sturgeon compared to surrounding areas: both were typified by substrate containing elements of sandy material (substrate = 2.0) and finer substrate particles (substrate = 3.0), though the Lake Sturgeon locations tended towards more sand content.

## DISCUSSION

Upon review of studies reporting physiological requirements of acipenserid species, these fishes are adversely affected by the interaction between rising temperatures and falling dissolved oxygen saturations, and sturgeons in particular generally exhibit a greater negative metabolic response to changes in these habitat variables than other fishes (Secor and Niklitschek 2002). Hydroacoustic and telemetry studies consistently located acipenserid species occupying narrow ranges of available habitat. For example, sturgeons will inhabit increased depth during summer thermal maxima periods, including Chinese Sturgeon *Acipenser sinensis* (Zhang et al. 2014), Gulf Sturgeon *Acipenser oxyrhincus destoi* (Sulak and Clugston 1999; Hightower et al. 2002; Stewart et al. 2012), and Northern U.S. and Canadian Lake Sturgeon populations (Holtgren and Auer 2004; Smith and King 2005; Barth et al. 2009). In the case of the Gulf Sturgeon, listed as threatened under the Endangered Species Act in 1991 (USFWS 1991), the summer-fall holding areas of spawning rivers have been identified as areas deserving of special protection and enforcement from local state wildlife authorities as critical habitat necessary for the persistence of the species. The evidence concerning summer refuge and foraging habitat indicates that acipenserids will aggregate in relatively long reaches>10 rkm if suitable habitat exists during the summer-fall period of the year (i.e., reduced temperature and/or increased dissolved oxygen).

Given the evidence presented in this study, I conclude that I have identified one such area of importance to the Lake Sturgeon reintroduced to the Upper Tennessee River system: the SHA on Watts Bar Reservoir. This conclusion is supported by the evidence that Lake Sturgeon individuals persist in this area, as they have been reliably detected and/or captured on trotlines for more than 5 years. Furthermore, I have provided an extensive accounting for the ranges of physical habitat variables that are found within this SHA during the critical summer thermal maxima period. The measurements I recorded from Lake Sturgeon locations, however, appear to suggest trends in habitat utilization which are counterintuitive to physiological evidence regarding Lake Sturgeon. For example, all North American species generally prefer and perform optimally under cool (<25 $^{o}$C) temperatures (Cech and Doroshov 2004). The strong inverse relationship between temperature and dissolved oxygen content suggest that at least some of the individuals detected were outside optimal habitat locations. It may be that in this instance, the data instead describe the individual behaviors of Lake Sturgeon in the process of transit, or that the fish are tolerating low dissolved oxygen conditions in the cool water of thermally stratified reservoirs.

The results of the boxplot analysis did not suggest that the Lake Sturgeon are exhibiting habitat selectivity, and includes results that are difficult to apply biological relevancy to (i.e. the lower dissolved oxygen levels and greater temperatures found at some Lake Sturgeon-present locations). These results precluded the use of more advanced statistical analysis or modeling techniques

Future studies are required to expand on the data I have provided here to develop a broad knowledge base regarding Lake Sturgeon foraging habitat in the Upper Tennessee River system during summer thermal maxima. Future efforts should seek to add both more Lake Sturgeon observations and more surveys of the total riverine and reservoir habitat available to this species, to improve on the interpretability of the distinction between suitable and unsuitable Lake Sturgeon habitat. With a more robust dataset, statistical classification methods such as classification trees and/or logistic regression could then be employed to determine the habitat variables of greatest importance in characterizing suitable Lake Sturgeon summer foraging habitat from unsuitable areas. Of particular utility in this undertaking would be the random forest procedure (Breiman 2001). This procedure has several advantages, first of which is that it employs bootstrapping in developing training datasets, and thus can handle differential sample sizes between Lake Sturgeon presence-absence observations (James et al. 2015). This process also develops non-correlated predictive classification trees by employing an additional layer of bagging (i.e., sampling) during the selection of predictive variables at each node in each tree built in conjunction with bootstrapping the training and test observations. This sampling of the predictors is necessary to avoid any single predictor masking the effects of other significant predictors when generating the constituent trees within the forest. Random forests can adequately handle a mixture of categorical and continuous predictors and do not make rigid assumptions about the distribution of the data (James et al. 2015). Finally, random forests (and classification tree methods more broadly) generate easily-interpretable dendrograms, which would aid in the communication of the results from researchers to fisheries managers and the public more broadly.

## ACKNOWLEDGEMENTS

I gratefully acknowledge my partners in the Southeastern Lake Sturgeon Working Group for their material contributions to the completion of this project, working with both myself and other graduate students. I am deeply thankful to Drs. Mark Bevelhimer and Brenda Pracheil at Oak Ridge National Laboratory for their material contributions and helpful suggestions. I would not have been able to complete this project without the determined efforts of my advisor, as well as C. Chapman, J. Coombs, J. Haney, K. Garner, M. Harris, and J. Wolbert helping to maintain equipment and conduct field work.

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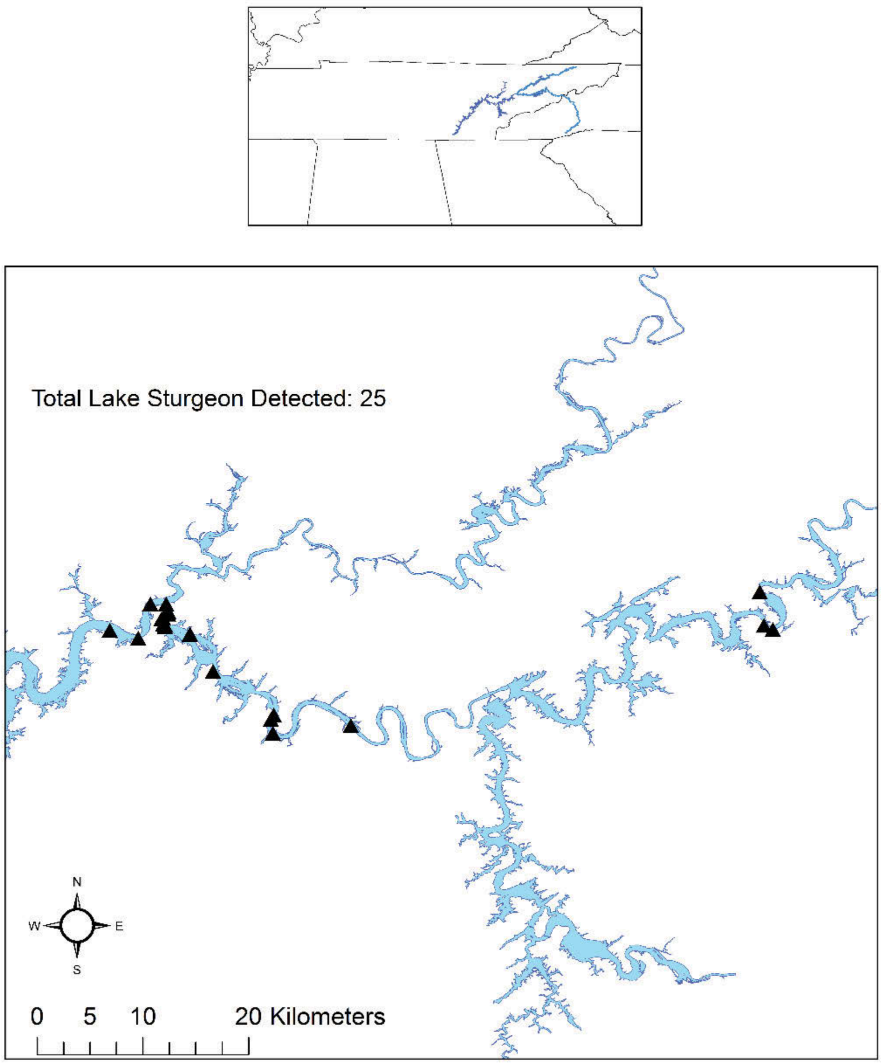
# APPENDIX C

### *Figures*




### *Figure 2.1*

Locations of all Lake Sturgeon (n = 25) detected during acoustic telemetry surveys, 31 May – 31 July 2016. Summer holding area indicated by cluster of detections in the western portion of map, Watts Bar Reservoir.

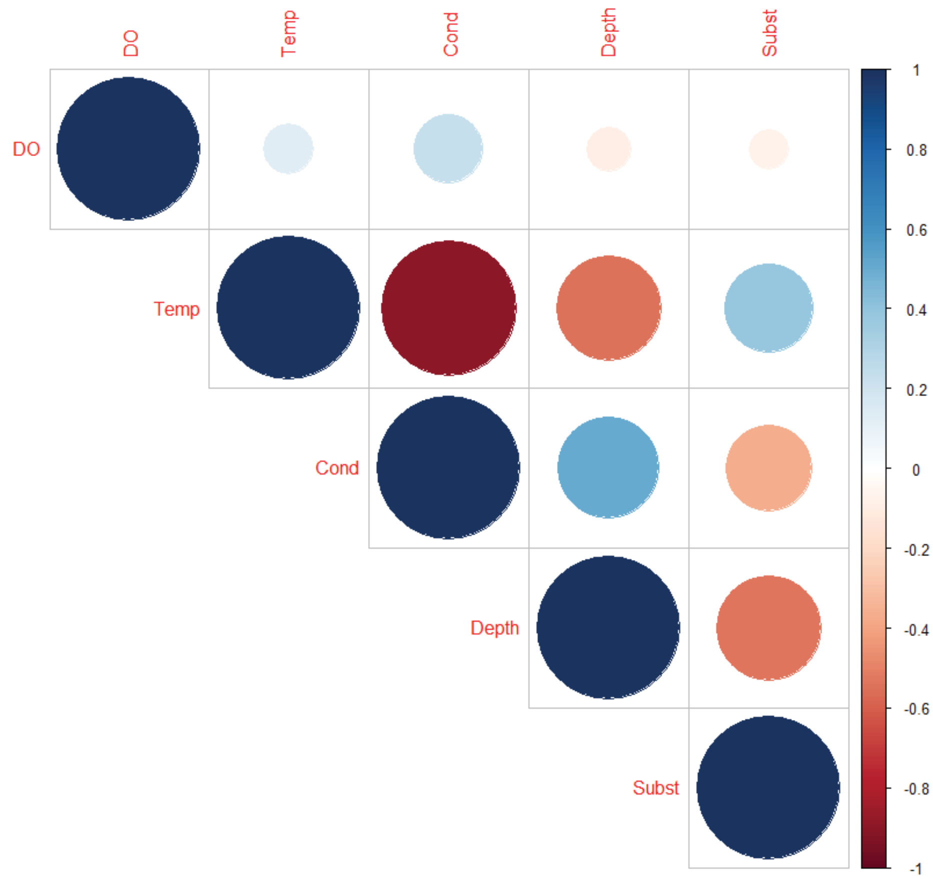



***Figure 2.2***

Correlation plot visualizing the Pearson's correlation among the five physical habitat variables measured at Lake Sturgeon-present and -absent locations within the summer holding area.

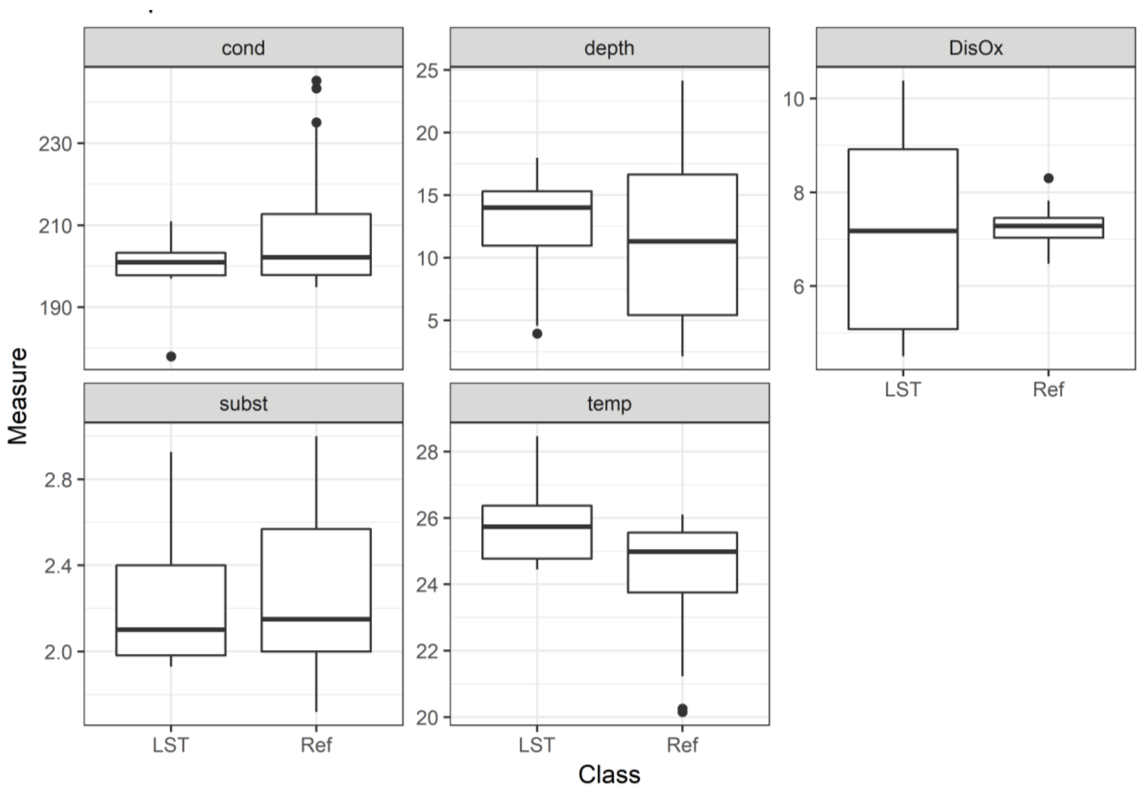



***Figure 2.3***

Boxplots comparing the median (dark line through box), lower quartile (lower whisker end), upper quartile (upper whisker end), second quartile (lower box boundary), and third quartile (upper box boundary) for habitat measurements at Lake Sturgeon present (left box) and absent (right box) locations. Outliers are indicated with dots.

# CHAPTER III
# POPULATION DYNAMICS OF REINTRODUCED LAKE STURGEON IN THE UPPER TENNESSEE RIVER

A version of this chapter was submitted for publication to the North American Journal of Fisheries Management on 13 September 2017 under the same title. It is still under review as of 8 October 2017.

## ABSTRACT

Efforts to reestablish the Lake Sturgeon to the Tennessee River drainage have been underway for nearly 20 years. Since 2011, annual semi-quantitative sampling efforts have reliably encountered reintroduced Lake Sturgeon in in the reaches encompassing Watts Bar and Fort Loudoun Reservoirs. I generated the first estimates of population size and survival of reintroduced Lake Sturgeon in these reservoirs using the POPAN Jolly-Seber open population model in Program MARK. I observed $AIC_C$ to evaluate multiple model parameterizations and identify the best performing model. The best model estimated 5,643 reintroduced Lake Sturgeon inhabiting Fort Loudoun and Watts Bar Reservoirs in 2011. Estimated annual survival was 42% (95% C.I.: 15 – 75%). I next used this range of survival estimates to evaluate whether reintroduction efforts at current annual stocking numbers alone would reach the reintroduction goal of 20 year classes >15 years of age before total mortality consumed previous year classes. I found that under the current stocking regime without natural recruitment, only under low mortality conditions does this population achieve the reintroduction goals through stocking alone. This research highlights two management needs: to continue monitoring survival and abundance of Lake Sturgeon released into the Upper Tennessee River, and to dedicate resources to ensuring natural recruitment occurs and succeeds.

# INTRODUCTION

The SLSWG has outlined in its management plan a goal of 20 different age classes of fish >15 years of age present in the system (SLSWG 2015). This goal was adopted on two assumptions: 1) that Lake Sturgeon inhabiting the warm waters of Tennessee would begin to reach sexual maturity at around 15 years old (e.g. Peterson et al. 2002), and 2) that many year classes of individuals above that age would promote variations in spawning individuals from year to year, given the periodic nature of individual Lake Sturgeon spawning attempts. To date, no Lake Sturgeon spawning events have been documented in the Tennessee River system, although systematic efforts to observe spawning have been limited (D. Walker, unpublished data). Because the reintroduction began in the year 2000, the earliest that this goal could be met is the year 2035.

Since 2011, annual semi-quantitative sampling has been conducted to capture Lake Sturgeon reintroduced to the Tennessee River, with the overall goal of estimating various parameters such as population size and survival (SLSWG 2015). Prior to release, all Lake Sturgeon have one or two lateral scutes removed in a predetermined pattern to differentiate year classes. However, body modifications to sturgeon species can be unreliable as a method of ageing due to tissue regeneration and variation in individual variability in growth rates (Smith et al. 2002). Given the large numbers and relatively small size of Lake Sturgeon at release, and high costs in material and man-hours, tagging for individual recognition does not take place prior to release. Beginning in 2011, standardized trotlines were selected as the best sampling gear for monitoring this population after other methods (e.g., gill netting) proved ineffective. Trotlines are deployed along the channel of the Tennessee River to capture Lake Sturgeon, and every captured Lake Sturgeon receives a unique passive integrated transponder (PIT; 125, 130, or 134 kHz) tag. The annual sampling and tagging of captured Lake Sturgeon with individually-identifying PIT tags constitutes a capture-mark-recapture population assessment design (Williams et al. 2002). For the purposes of population assessment in this study, I am considering the capture of reintroduced Lake Sturgeon and implantation of unique identifying tags as the first capture event, though these events could be considered a recapture of the reintroduced animals.

The objectives of this study were to 1) generate population size and survival estimates for reintroduced Lake Sturgeon in Fort Loudoun and Watts Bar reservoirs of the Tennessee River, and 2) to test by simulation whether under the current stocking regime, one of the stated goals of reintroduction (i.e., 20 year classes >15 years of age) could be achieved without relying on uncertain future natural recruitment.

## METHODS

### *Study area*

Fort Loudoun and Watts Bar Reservoirs constitute the two upstream-most reservoirs on the Tennessee River main stem. Watts Bar Dam is located at Tennessee River mile (TRM) 530, and Fort Loudoun Dam is located at TRM 602.

The confluence of the French Broad and Holston Rivers at TRM 652 marks the beginning of the Tennessee River main stem. The location of each Lake Sturgeon capture or recapture included in this study, relevant TVA dams in the area, and the Upper Tennessee River and several of its major tributaries are shown in Figure 1. I used a semi-quantitative, standard sampling design. Trotlines were used to capture Lake Sturgeon and were 125 m long with 0.3-m drop lines occurring at 1-m intervals. Circle hooks (#2 size) were baited with 12.5-cm pieces of Common Carp (*Cyprinus carpio*) or Buffalo fish (*Ictobius spp*.) muscle tissue. The number and location of deployed trotlines varied from year to year based on the judgement of the field crews deploying them and the physical conditions they encountered during each sampling season (e.g., intentional avoidance of areas of dense aquatic vegetation to avoid loss of gear). Additionally, sampling locations can vary based upon the current research needs of individual members of the SLSWG. Sampling occurred each fall when surface water temperatures fell to approximately 15-18$^{o}$C from the summer maxima. The timing of this condition varies, but the majority of sampling efforts occurred in November or December of each year. Sampling is typically limited to a 5-day week on the Upper Tennessee River before the effort is directed downstream. Trotlines were deployed perpendicular to the direction of water flow and located along the margins of the navigation channel to ensure that the lines extended across a range of depths. The trotlines soaked overnight for approximately 18 hours and were retrieved the next day.

### *Lake Sturgeon handling and tagging*

For all captured Lake Sturgeon I recorded body measurements, including weight (g), TL and fork length (FL) in mm. Then, I assessed each fish for missing scutes to indicate year-class. Recaptured fish were detected with a PIT tag reader (Biomark 601, Biomark Inc., Boise, ID) by scanning the entire body for PIT tags. If I did not detect a PIT tag, I considered the fish to be a novel catch. I injected a PIT tag with a unique identifying number into the muscle tissue adjacent to the dorsal fin of every new Lake Sturgeon captured. If I detected a PIT tag, I recorded the unique number. For both novel and recaptured Lake Sturgeon, I recorded tag numbers with the individual's associated length, weight, and age data. I verified that all newly injected PIT tags were detectable both before and after injection with the PIT tag scanner. I then allowed every Lake Sturgeon time to recuperate in an aerated or oxygenated holding tank until I observed strong swimming behavior. Once the fish were able to swim under their own power efficiently, I returned all Lake Sturgeon to the reservoirs within 100 m of the site of capture. In 5 years of trotline sampling, I encountered only one Lake Sturgeon mortality directly associated with capture, representing a mortality rate associated with sampling of less than 0.33%.

### *Abundance and survival estimation*

The first goal of this study was to generate an estimate of survival and population size for Lake Sturgeon reintroduced to the Upper Tennessee River system (i.e., Fort Loudoun and Watts Bar Reservoirs). I limited my analyses to the fish recaptured in Fort Loudon and Watts Bar Reservoirs as these are the only sampling reaches where ≥1 recapture has occurred during this time period in the Tennessee River system. To achieve this objective I used the program MARK (White and Burnham 1999; Cooch and White 2017) to perform maximum likelihood estimation of parameters using the capture histories I recorded during 2011-2015. I used the package ‘RMark’ (Laake 2013; Laake and Rexstad 2017) in the program R (R Core Team 2016) as my user-interface to program MARK. In doing so I took advantage of certain benefits of combining both R and MARK, such as the capability to programmatically develop the capture histories, design matrices, and other input data required for analysis in MARK, and the ability to use scripts to document and replicate analyses. Because I had a preexisting familiarity with the R language and environment, I chose to use RMark even though it does not replicate every model present in MARK (Laake and Rexstad 2017).

Since I were interested in estimating population size as well as survival, I used a Jolly-Seber model within program MARK (Jolly 1965; Seber 1965; Williams et al. 2002). Within RMark, I implemented the POPAN formulation of the original Jolly-Seber model (Schwarz and Arnason 1996), as this was the only open-population model available which generated an estimate of population size in this interface, and the other formulations within native MARK were likely to generate similar parameter estimates (Cooch and White 2017).

I assumed that this is an open population based upon prior research. A previous acoustic tagging and monitoring study found evidence that the Lake Sturgeon of the Upper Tennessee River were capable of downstream passage though the hydroelectric dams on the Tennessee River (Saidak 2015), which I consider to be a plausible avenue of permanent emigration as there currently is no evidence of Lake Sturgeon traveling upstream of any TVA hydroelectric dam they encounter. Additionally, the acoustic monitoring research did not suggest that downstream passage was a common phenomenon (one incidence over 18 months of monitoring) which supports the assumption that the population remains closed during the week-long sampling intervals (i.e., sampling is instantaneous). Furthermore, a new release of Lake Sturgeon from aquaculture facilities occurred each year during sampling, suggesting possible recruitment of new individuals into my study area before later sampling events. For the purposes of this study, I assume the current design to meet the rest of the assumptions of the Jolly-Seber model, including homogeneity of catchability and survivability of tagged and untagged fish, tag retention throughout the study, and successful detection of all tags if the individual is a recapture (Williams et al. 2002; Schwarz and Arnason 2017). I acknowledge that future tagging and recapture studies are necessary to validate these assumptions.

I developed individual capture histories for each fish for which I had complete identification, location, length, and weight records ($N_{total} = 213$, $N_{recapture} = 3$). When I developed my models, I included categorical covariates in the maximum likelihood analysis. I plotted length and age histograms, and observed breaks in each distribution at TL = 650 and 950 mm (Figure 2). As mentioned previously, the external bodily modifications which I used to identify fish age may be unreliable, but the evidence presented in Figure 2 suggests I can capture both length and age variation with a single length-classification categorical variable, and the distribution of the length histogram supports multiple year classes present in the population. Therefore, I categorized each fish as short, average, or long length based on its TL (Table 1). I placed each fish into a weight category determined by whether it was greater or less than the average weight of all fish included in the analysis (mean weight = 2.5 kg; Table 1). Table 2 illustrates how many fish occurred in each combination of length and weight classifications. Given the difficulty in identifying sub-adult Lake Sturgeon sex by external observations alone, I did not include sex as a covariate. After fully coding my data and capture histories, I generated a series of 64 models testing all permutations of the parameters. I used Akaike's Information Criterion adjusted for small sample sizes ($AIC_C$) to evaluate all 64 models simultaneously (Akaike 1974; Hurvich and Tsai 1989).

***Population dynamics simulation***

Prior efforts have failed to document reproduction by reintroduced Lake Sturgeon in the Upper Tennessee River (D. Walker, unpublished data). One potential explanation is that the population has not reached sufficient numbers of sexually mature individuals. Therefore, I simulated future annual reintroduction numbers from the numbers reintroduced into the Upper Tennessee River between 2000 and 2016 as a proxy measure for annual Lake Sturgeon recruitment. I assumed that the number of Lake Sturgeon stocked each year came from a normal distribution, with a mean and standard deviation calculated from the numbers reintroduced (mean number of fish stocked each year (2000-2016) = 8862, S.D. = 3616.25). From this distribution, I randomly selected annual stocking numbers for the years 2017-2035 (Figure 3). I assumed that future stocking numbers will be within this distribution to maintain adequate levels of genetic diversity when and if natural spawning begins. I used the stocking numbers from the years 2000-2016 and the simulated numbers from 2017-2035 to represent the annual recruitment each year to the Upper Tennessee River system in the absence of evidence of natural recruitment for the purposes of simulating the effects of total mortality on this population.

I evaluated a total of 64 POPAN Jolly-Seber Models in RMark. These models included the terms Phi, or survival; PENT, or probability of entering the sampling population from the hypothetical superpopulation characteristic of the POPAN formulation; p, or the probability of capture; and N, or the abundance of fish in the sampling population immediately before sampling began. I also included the three-level length and two-level weight categorical covariates.

After identifying the best model, I recorded the annual survival estimate and its 95% confidence interval. I then randomly applied a survival rate from within that 95% confidence interval to each year-class each year between the years 2000-2035. To model how each age class declined due to natural and fishing mortality each year, I used a density-dependent Ricker-type logistic model:

$$N_{t+1} = N_t \cdot e^{-Zt}$$

Where "$N_t$" is the abundance of a single age class at time "t", "Z" is the inverse log of the survival estimate selected for a year-class, and "t" is a constant as new calculations were applied from year to year (i.e., all time increments were +1 year) (Allen and Hightower 2010). I ran each simulation from the year 2000-2035, and I performed 1000 simulations. I then plotted the mean and standard errors of abundances of each year-class persisting in the year 2035.

## RESULTS

### *Abundance and survival estimates*

Of the 64 population size models I evaluated, the top ten are listed in Table 3. The model with the lowest $AIC_C$ score included the weight-code covariate with the Phi term, and the interaction of weight and length covariates term on the estimate of abundance. I refer to this model as the 'Phi.weight.p.N.lengthxweight' model.

The 'Phi.weight.p.N.lenghtxweight' model produced a total population size estimate of 5,643 Lake Sturgeon present in the Upper Tennessee River System immediately before the first sampling event in 2011 (Table 4). The majority of individuals in this population (3,543, or 62.8%) fell into the light-average length category. The second most abundant group (853 individuals, or 15.1%) were categorized as light-short. A similar number of individuals were in the heavy-long category (532, or 9.4%) and the heavy-average-length category (682, or 12.1%). No individuals in the recapture data I analyzed qualified as heavy-short. The best model for my  data only produced significant Phi, or survival, estimates for the heavy-average-length and light-average-length classifications. The survival estimate for fish over the average weight and of average length approached 1, but the lower confidence interval included 0, so I disregard this result as spurious for modeling purposes. The survival estimate for the majority of the fish estimated to be in this population, those of average length but below average weight, was 42.6% (95% C.I. = 15.5 – 75.1%). As the majority of the Lake Sturgeon I have encountered in the Upper Tennessee River system fall into this class, and the estimates appear to be valid, this was the survival estimate that I applied in the population dynamics simulations.

***Population dynamics simulation***

I ran 1000 simulations, in each of which I randomly selected a value from the 95% C.I. range of survival (0.15 – 0.75, by 0.01 units) estimated by the best POPAN Jolly-Seber model in the first analysis and converted it to total mortality (Z) by computing the inverse-log. I did not attempt to separate natural from fishing mortality for the purpose of this analysis, because the species is protected in Tennessee. I applied a unique mortality rate to each year class beginning with its introduction to the population over each year from the year of introduction (i.e., the ‘birth year’ of the year class) to the year 2035. The results of the simulation under this original condition is demonstrated in Figure 4A, showing the average abundance estimate for each age class ± 1 standard deviation. Because I applied only a logarithmic population decline equation to each year class, all year-classes became extirpated roughly 5 years after reintroduction, thus the only year-classes present at 2035 (Figure 4A) are those added to the population within the last 5 years. Since the age and length data from this population (Figure 2) indicate the presence of more than 5 year-classes, I then tested two other simulation scenarios to identify conditions that would achieve the reintroduction goal.

The next condition I tested through simulation was to fix the mortality rates applied to each year class at specific values, and maintain the real and simulated numbers of reintroduced fish used in the first simulation. I set the survival rate for fishes in their first three years of life in the Upper Tennessee River system at 75%, and I set the survival rate for fish greater than three years of age at 99%. I chose the elevated rate for older Lake Sturgeon on the assumption that once Lake Sturgeon in the Upper Tennessee River reach that age and concurrent size, they are unlikely to face many natural predators, and should be large enough to effectively leave or avoid areas of inhospitable habitat. After 1000 simulations, the sum of the average number of Lake Sturgeon of all age classes present in the Upper Tennessee River system in the year 2035 is 25,833 individuals (Figure 4B). Of this total population, the simulated number of individuals 15 years of age or older is 1,025.

While the simulations under the conditions of reduced mortality projected that some individuals of reproductive age would persist under very low mortality conditions, 1,025 reproductive individuals present in the population after 35 years of reintroduction efforts is low. I therefore tested how that number would change by doubling the number of Lake Sturgeon reintroduced from the years 2018 and beyond while still maintaining the set age-class specific survival rates of the previous simulation. According to my simulations, if Lake Sturgeon reintroductions were to be doubled in the remaining years, the average number of fish present in the Upper Tennessee River would be 42,222 individuals in the year 2035 (Figure 4C). This represents an increase of 160% in total number of individuals present under the current stocking regime while assuming equally high rates of survival. Similarly, the average number of fish of reproductive age after 1000 simulations was 1,629, representing an increase of 159%.

## DISCUSSION

A foundational concept in natural resources management is the idea of adaptive management, where resource managers continually evaluate the progress of their actions and make changes, if necessary. Therefore, an undertaking such as the reintroduction of Lake Sturgeon to the Tennessee River, which requires large amounts of private, state, and federal resources and coordination to begin and maintain, must be able to self-evaluate and adapt to new insights. My results are encouraging, with some concern. A survival rate of 42%, and potentially much higher, for Lake Sturgeon susceptible to the current sampling protocols is positive. Further research is necessary to track and assess smaller, younger year classes of Lake Sturgeon so that we have a fuller understanding of mortality rates endured by reintroduced fish throughout their life history. My results indicate that once Lake Sturgeon reach a size susceptible to trotlines, mortality is not an overwhelming factor determining population structure. Furthermore, even though the confidence interval generated by my Jolly-Seber model for the largest fish is statistically spurious, it is highly likely that Lake Sturgeon in the Tennessee River can reach sizes beyond which natural predation is no longer a contributing factor to mortality. This translates to an obvious management goal of maintaining and expanding suitable growth conditions, such as maintaining suitable temperature and dissolved oxygen regimes and monitoring Lake Sturgeon diets to ensure that they achieve the large sizes necessary to preclude natural predation.

While my current population assessments suggest positive trends, the population simulation results strongly suggest that the population goals guiding the reintroduction are not achievable through stocking alone. I was able to generate total population size estimates of fish in the target year classes in the year 2035 only by deliberately lowering total mortality experienced by all age-classes. Under similar mortality conditions and with twice the mean stocking rate for future years, I estimate an average of 1,629 reproductively mature individuals in the Upper Tennessee River in the year 2035. I selected this year to be the endpoint of my simulations as it is the first year that the goal condition of 20 year classes >15 years of age could be realized. In an era of shrinking budgets and increasingly diverse conservation needs, I assume that achieving the restoration goals in as timely a manner as possible is a priority.

However, a spawning population of 1,629 reproductively mature Lake Sturgeon dispersed across a range of up to 296 rkm (the distance from Watts Bar Dam to Cherokee Dam, the longest upstream route a Lake Sturgeon in the study area can travel on its own assuming it successfully passes through the shipping lock at Fort Loudoun Dam) represents a density of approximately 5 reproductively mature Lake Sturgeon per river kilometer. Given the periodicity of individual Lake Sturgeon spawning attempts, the annual density of reproductive fish decreases further. For comparison, the Kootenai River (Idaho) White Sturgeon (*Acipenser transmontanus*), the only land-locked population of the typically anadromous species, is listed as endangered under the U.S. Endangered Species Act (USFWS 1994; Anders et al. 2002; Paragamian et al. 2005). While adult Kootenai White Sturgeon persist, natural annual recruitment continues to fail, potentially due to loss of quality spawning habitat. One recovery goal for this population is the presence of a stable population of 7,000 adult individuals, the number of fish estimated present before the closure of a hydroelectric dam and beginning of recruitment failure (Parmagnian and Hansen 2008). This translates to a density of over 36 reproductive Kootenai River White Sturgeon per rkm across 190 rkm of habitat, or a 7-fold increase over my projected density of reproductive Lake Sturgeon present in the Upper Tennessee River in the year 2035.

My population simulation model is deterministic, and I performed the simulations under the assumption of no natural recruitment contributing to year-classes. My simulation results should be interpreted as a baseline assessment of a worst-case scenario for this reintroduction. They should also serve to highlight the necessity of ensuring natural reproduction by reintroduced Lake Sturgeon occurs to supplement and eventually supplant stocking of hatchery-raised fish. Lake Sturgeon are an intensive species to raise in aquaculture, and natural reproduction is both a goal of the restoration as well as a superior method of population management, at the very least in terms of return on investment for fisheries managers. A previous study has found suitable spawning substrate downstream of several TVA hydroelectric dams on the Tennessee River (Walker and Alford 2016), which could contribute to overcoming some of the reproductive hurdles faced by the Kootenai River White Sturgeon population. The next step in the restoration of this population should be detection of spawning Lake Sturgeon and determination of whether larvae have enough suitable riverine habitat to drift and survive in this system of reservoirs. Once specific spawning areas are located, managers can take actions to protect the habitat and the fish when they aggregate and the larvae as they drift downstream. Finally, I recommend that more extensive sampling be done to increase the capture-recapture data available for this population, as well as further study to test the population modeling assumptions I made here. By increasing both the overall number of Lake Sturgeon captured, in particular the number of recaptures, future population modeling efforts can expect to reduce the confidence intervals of their estimates, and improve the accuracy of the data and conclusions driving the effective adaptive management of this population.


## ACKNOWLEDGEMENTS

I would like to acknowledge the invaluable input and assistance I received from our partners in the Southeastern Lake Sturgeon Working Group during the course of this study, as well as instructive comment from M. Bevelhimer and B. Pracheil from the Oak Ridge National Laboratory. I am especially grateful to the Wisconsin Department of Natural Resources, without whom the reintroduction of Lake Sturgeon across the Southeast would not be possible. Finally, I am grateful to our partners and friends in the Fisheries Lab at the University of Tennessee for their support in the field: J. Coombs, K. Garner, and J. Wolbert.

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# APPENDIX D

***Tables***

***Table 3.1***

Size definitions used to classify Lake Sturgeon in Jolly-Seber models.

| Categorical Variables | TL, mm | Weight, g |
|---|---|---|
| Short | >650 | |
| Average | 650-950 | |
| Long | <950 | |
| Light | | >2507 |
| Heavy | | <2507 |

***Table 3.2***

. Number of Lake Sturgeon encountered in each length and weight category pair.

| Classifications | | | | |
|---|---|---|---|---|
| | | Weight | | |
| | | Heavy | Light | Total |
| | Average Length | 41 | 112 | 153 |
| Length | Long | 32 | 1 | 33 |
| | Short | 0 | 27 | 27 |
| | Total | 73 | 140 | 213 |

*Table 3.3*

Top 10 POPAN Jolly-Seber models ranked from lowest AICc to highest.

| Model Terms | # of Parameters | AICc | Delta AICc | Weight | Deviance |
|---|---|---|---|---|---|
| Phi(~weight.code)p(~1)pent(~1)N(~TL.code * weight.code) | 10 | 181.87 | 0 | 9.49E-01 | -445.43 |
| Phi(~TL.code)p(~1)pent(~1)N(~TL.code * weight.code) | 11 | 189.91 | 8.05 | 1.70E-02 | -439.61 |
| Phi(~1)p(~1)pent(~1)N(~TL.code * weight.code) | 9 | 190.17 | 8.30 | 1.50E-02 | -434.93 |
| Phi(~TL.code)p(~TL.code)pent(~1)N(~TL.code * weight.code) | 13 | 190.51 | 8.64 | 1.26E-02 | -443.52 |
| Phi(~TL.code)p(~TL.code * weight.code)pent(~1)N(~TL.code) | 13 | 191.89 | 10.02 | 6.33E-03 | -442.14 |
| Phi(~TL.code)p(~TL.code * weight.code)pent(~1)N(~TL.code * weight.code) | 16 | 196.67 | 14.80 | 5.81E-04 | -444.29 |
| Phi(~1)p(~TL.code)pent(~1)N(~TL.code * weight.code) | 11 | 206.17 | 24.31 | 5.00E-06 | -423.35 |
| Phi(~weight.code)p(~TL.code)pent(~1)N(~TL.code * weight.code) | 12 | 206.91 | 25.05 | 3.45E-06 | -424.85 |
| Phi(~1)p(~TL.code * weight.code)pent(~1)N(~TL.code) | 11 | 207.44 | 25.57 | 2.65E-06 | -422.078 |
| Phi(~weight.code)p(~TL.code * weight.code)pent(~1)N(~TL.code) | 12 | 207.71 | 25.84 | 2.32E-06 | -424.05 |

***Table 3.4***

Population size and survival estimates of the best POPAN Jolly-Seber model.

| | Parameter | Estimate | Standard Error | LCI | UCI |
|---|---|---|---|---|---|
| **Nhat 2011** | Heavy:avg.length | 682 | 420 | 239 | 2110 |
| | Light:avg.length | 3543 | 2049 | 1273 | 10244 |
| | Heavy:long | 532 | 330 | 185 | 1660 |
| | Light:long | 31 | 35 | 5 | 191 |
| | Light:short | 853 | 513 | 296 | 25617 |
| | Total | 5643 | | | |
| **Phi** | Heavy:avg.length | 0.99 | 1.6818e-5 | 0 | 1 |
| | Light:avg.length | 0.43 | 0.17462 | 0.1547 | 0.7505 |

# APPENDIX E

***Figures***

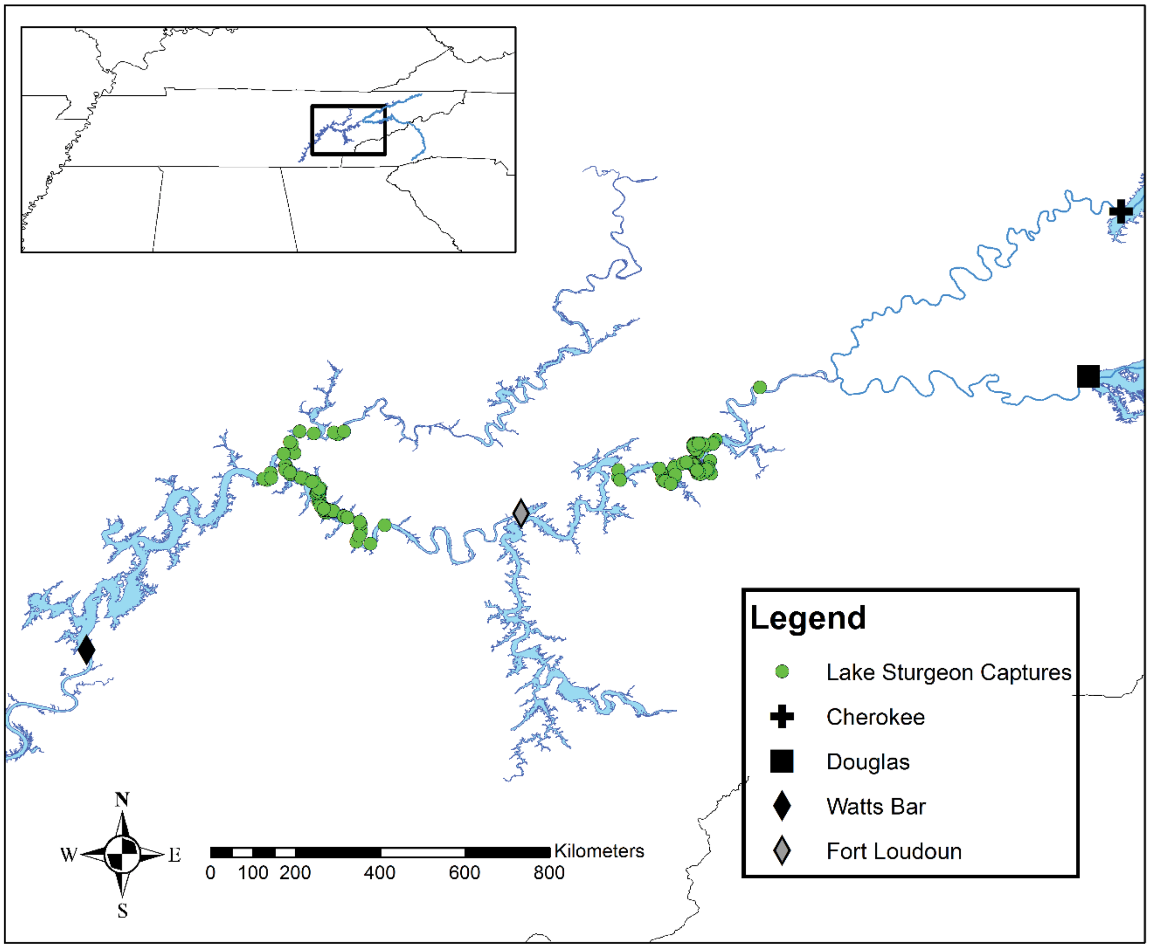



***Figure 3.1***

Map of the Upper Tennessee River system, showing capture location of 213 Lake Sturgeon analyzed in this study.

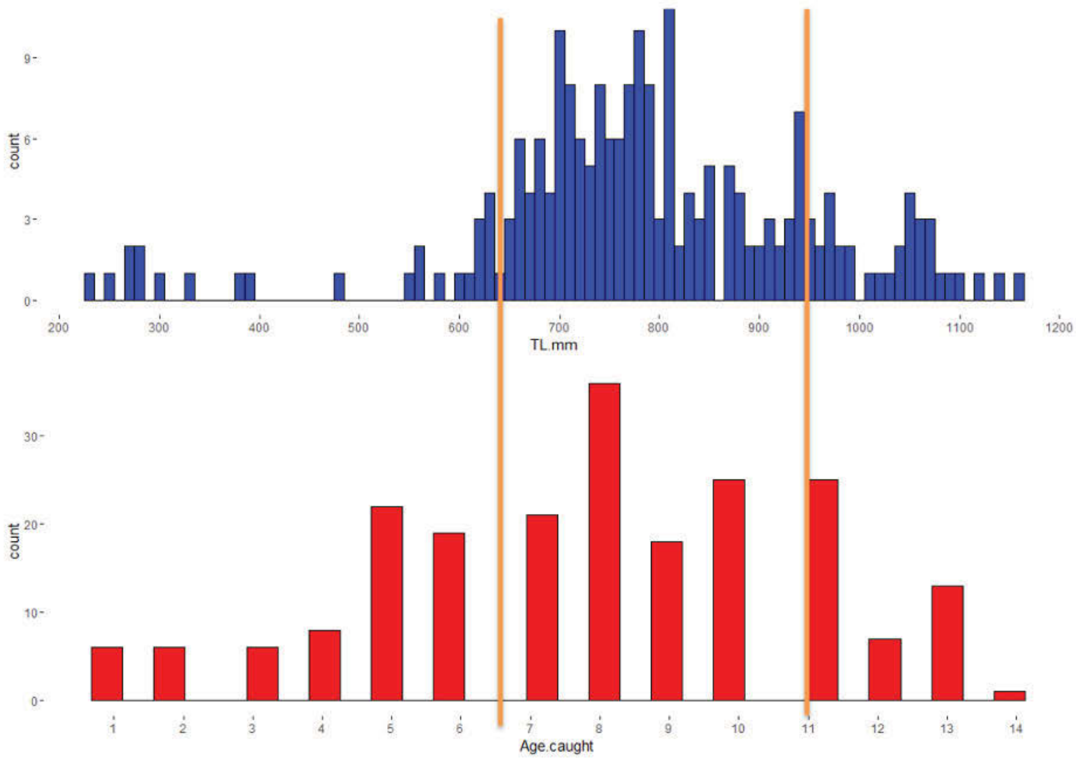



***Figure 3.2***

Length and age histograms of Lake Sturgeon included in study. Dividing lines are placed at TL = 650 mm and TL = 950 mm.

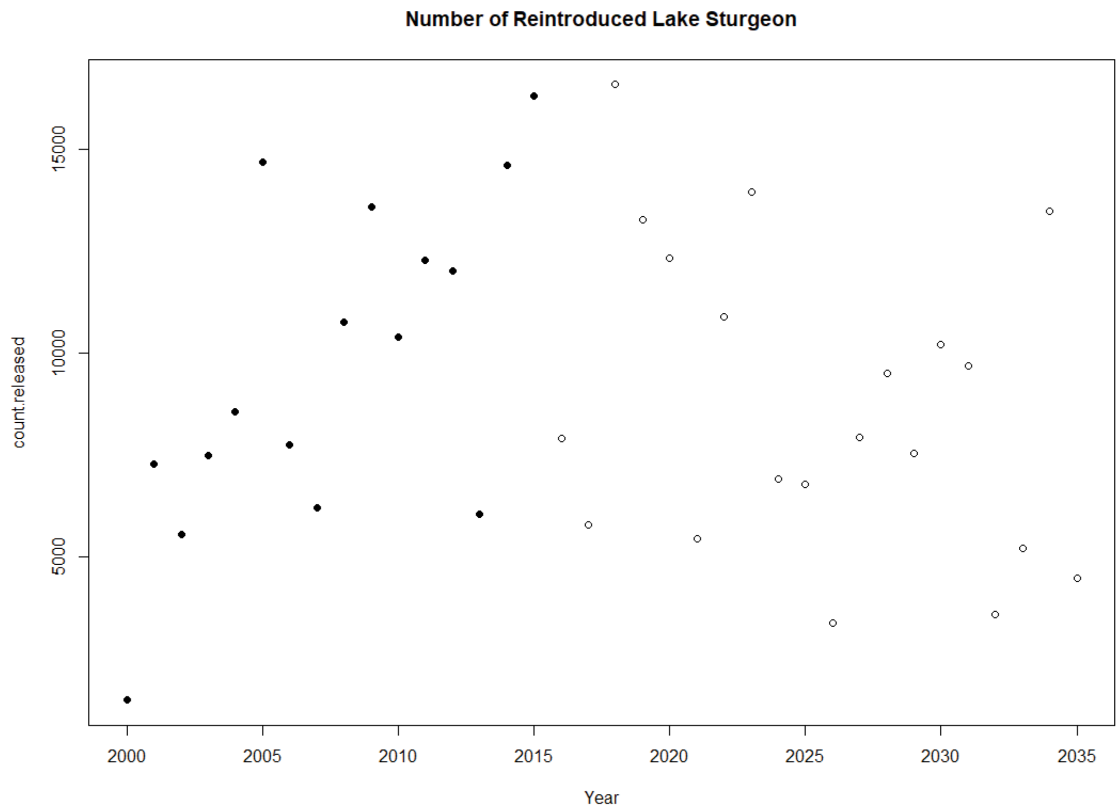



***Figure 3.3***

Scatter plot of actual (filled) and simulated (hollow) Lake Sturgeon stocking numbers.

***Figure 3.4***

Mean population sizes for each year class after 1000 simulations. Error bars are ±1 S.D. A) Year-class abundances after first simulation, with fixed $N_0$ and variable Z. B) Year-class abundances after simulation with fixed $N_0$ and fixed Z. C) Year-class abundances after simulation with twice the simulated reintroduction numbers and fixed Z.

A)

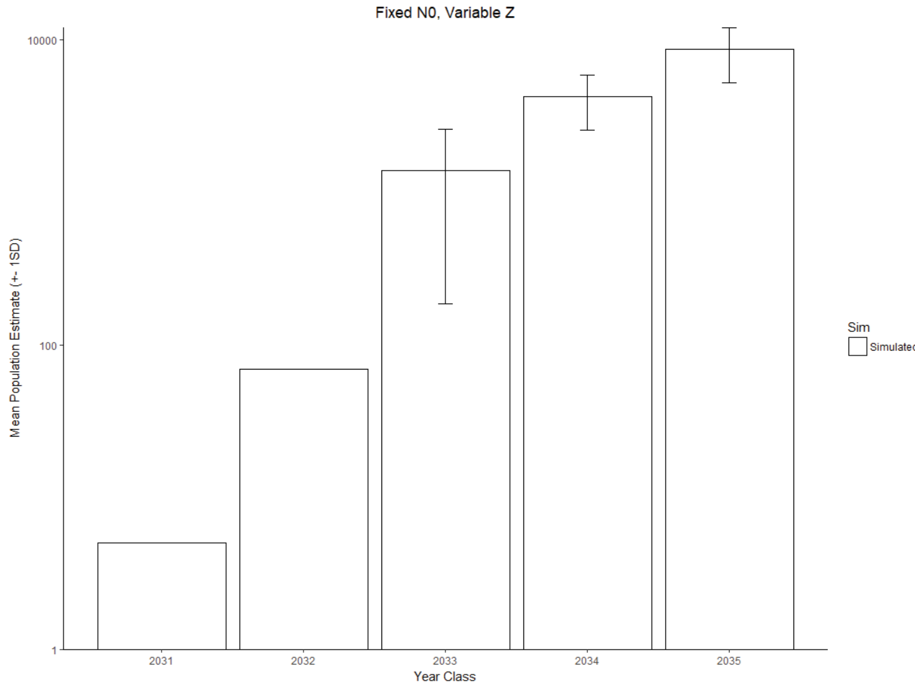



Figure 3.4 continued

B)

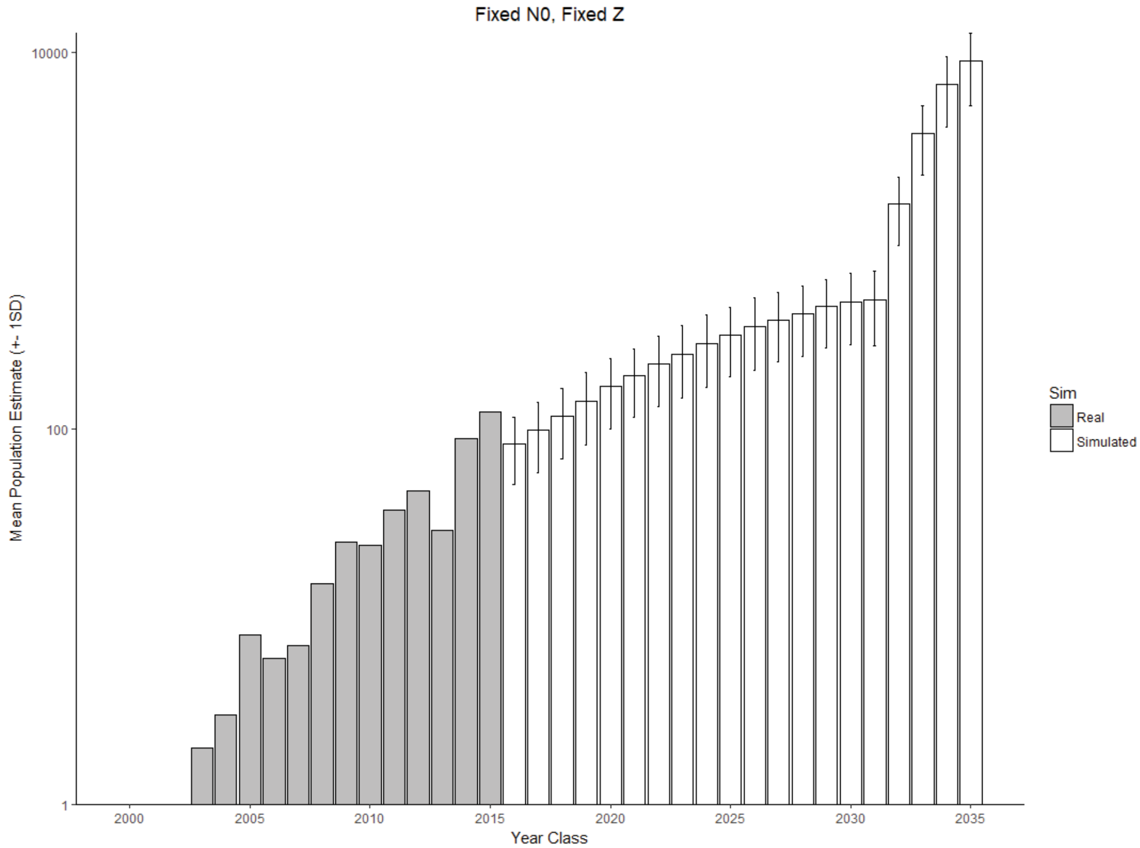
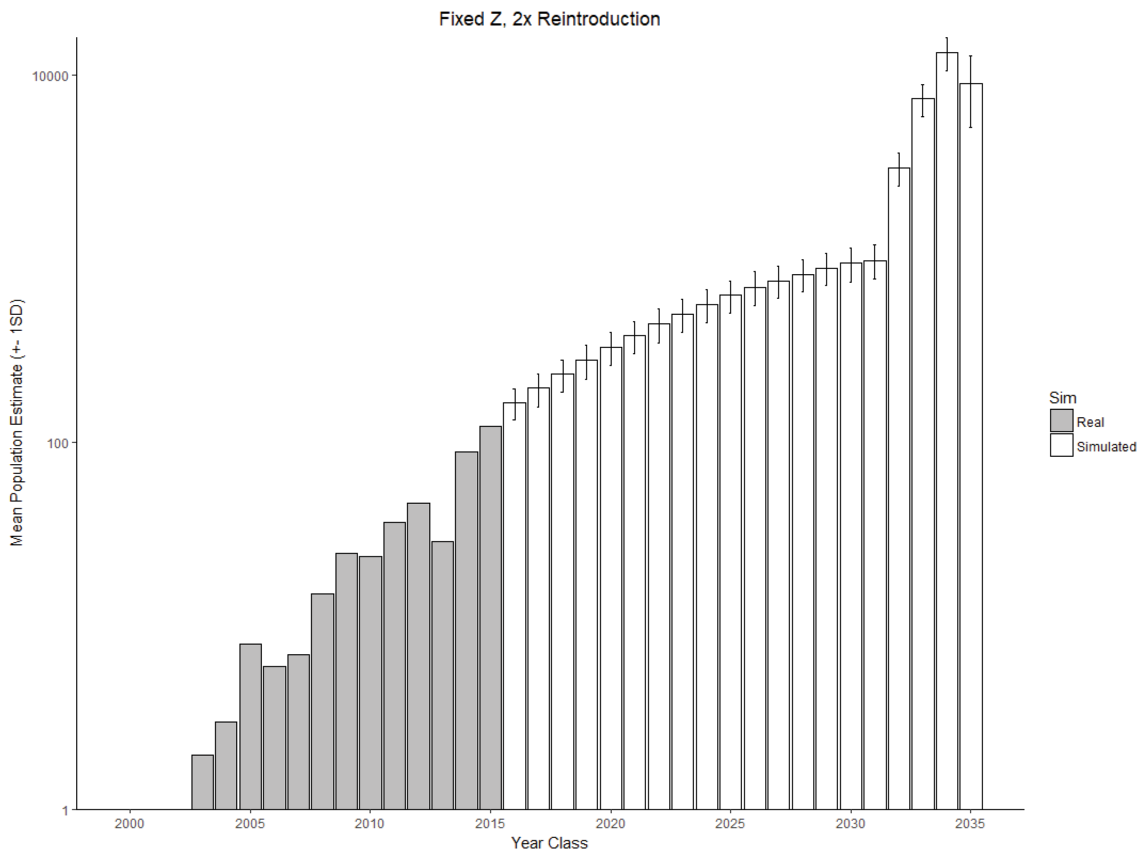



Figure 3.4 continued.

C)



Fixed Z, 2x Reintroduction
Mean Population Estimate (+- 1SD)
10000
100
1
2000
2005
2010
2015
2020
2025
2030
2035
Year Class
Sim
Real
Simulated

# CONCLUSION

In each of the three studies I have reported, I generated novel insights with applications towards the ongoing and future management of Lake Sturgeon reintroduced to the Upper Tennessee River and beyond. In the first chapter, I found evidence suggesting that if and when Lake Sturgeon currently inhabiting the various reservoirs of the Tennessee River undertake spawning migrations upriver, they are likely to encounter at least some suitable substrate over which they can spawn in the tailwaters of the major hydroelectric dams I surveyed. In the original habitat suitability model for the species, four physical habitat variables were included to determine the suitability of a reach for Lake Sturgeon spawning: depth, temperature, water velocity, and substrate. Of these four variables, three of them will be governed in the tailwaters I surveyed by the dam operation schedules, which in turn are developed days or weeks in advance to meet a variety of stakeholder demands (e.g. power generation, recreation, spring flood spillage, etc.). Therefore, from a fisheries management standpoint, substrate remains as the one variable upon which some management action could be taken to improve the success of Lake Sturgeon spawning and recruitment in these tailwaters. In the case of the Upper Tennessee River system, the agency tasked with management of the resources (the Tennessee Valley Authority) already has infrastructure in place for the deployment of cobble-boulder substrate, which is currently used as rip-rap to armor shorelines. When spawning aggregations of Lake Sturgeon are detected, it should be a relatively simple exercise to survey the substrate at the spawning ground and deploy artificial spawning reefs comprised of the rip-rap material to augment spawning suitability if necessary. If the spawning reliably occurs in the tailwaters of managed dams, not other low-head or relic dams, then managers may also be able to incorporate facilitation of suitable spawning characteristics in dam management and release schedules as well.

In my second chapter, I gathered detailed measurements in a restricted area that qualifies as a Lake Sturgeon summer holding area. This is a location where fish have been reliably detected using acoustic telemetry as well as a productive area during fall-winter sampling. I restricted the areas of sampling habitat variables for logistic reasons, namely that my limited time with the echosounder meant that only a limited area could be sampled for the substrate variable. The sparsity within my dataset precluded my use of more advanced analytical techniques to classify Lake Sturgeon-present habitat from other areas, so I present my findings here as a descriptive assessment of the summer holding area. I maintain that, given further resources and time, data can be collected such that the ensemble trees produced with random forest applications would provide a simple, interpretable diagram to effectively share the conclusions of the analysis with wide audiences who may lack the necessary statistical training to interpret results from other procedures for management applications. In future studies with greater datasets, a random forest classifier would be my primary choice of determining what specific habitat variables and ranges determine suitable Lake Sturgeon habitat within the Tennessee River system. From there, a geographic information system could be employed to correlate reference measurements taken in-stream to surrounding geomorphic, biological, anthropogenic, or other data sources that can be more easily collected via satellite surveys, and correlate those easily collected variable measurements with suitable in-stream conditions for Lake Sturgeon. The result of a broader analysis such as this would be the effective prediction of other summer holding area locations, where more Lake Sturgeon may seek shelter beyond what has currently been detected through trot-line sampling alone.

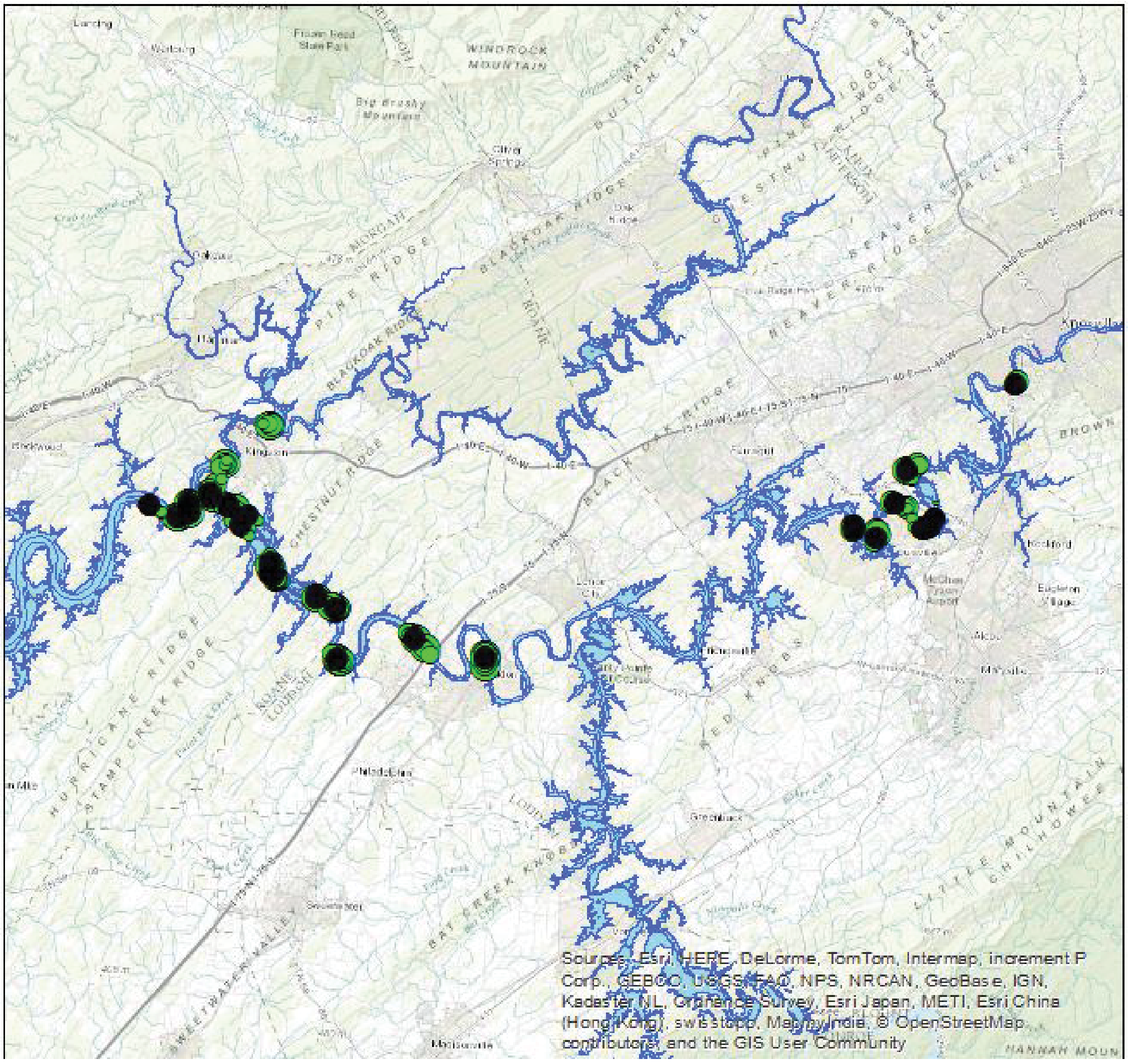
In my third study, I pivoted from assessing habitat availability and suitability for Lake Sturgeon to an estimation of the total population size and survival and mortality. Here, my findings raise the greatest alarm. Whereas I found evidence of suitable habitat for both spawning and foraging in the first two studies, both reasons to be optimistic about the restoration of this species to the Tennessee River, my conclusions regarding the population and its parameters are not as rosy. The vast majority of fish reintroduced to the Tennessee River have entered the system at Seven Islands State Birding Park, on the French Broad River. From there, they travel downstream to the upstream-most reservoir on the Tennessee River, Fort Loudoun. Our current sampling data suggest that there are many Lake Sturgeon inhabiting Fort Loudoun Reservoir, but in my analysis of the size and weight of these fish, I found that most individuals appear to be stunted in growth, reaching average lengths but not achieving correspondent weights. This conclusion matches anecdotal evidence from sampling, but greater sample sizes are necessary before rigorous conclusions can be drawn regarding the growth of the fish in Fort Loudoun Reservoir. While there is evidence of one individual successfully transiting Fort Loudoun Dam and entering Watts Bar Reservoir, there may be differences in the phenotypes expressed by Lake Sturgeon leading some individuals to travel great distances and many others to remain in limited home ranges. If this is the case, the evidence I present in Chapter 3 suggest that the fish remaining in Fort Loudoun, possibly the majority of surviving individuals in the Upper Tennessee River, are stunted in growth. Whether this stunting is density-dependent or not remains to be seen, but if reintroductions of young-of-the-year are possible anywhere downstream of Fort Loudoun Dam, it may be in the best interest of this population to release more individuals there than into the French Broad River.

I took the mortality estimate I generated with the population estimation procedure in the first part of Chapter 3 and used it as the input for 1000 simulations modeling the changes in abundance of each year-class introduced to the Tennessee River from the start of the program in 2000 to one projected endpoint, 2035. I found that only after manipulating mortality to very low levels were any fish greater than 5 years of age persisting in the system in the year 2035. While the assumptions of this model are stringent and not likely to fully represent the conditions within the population, it serves as a warning of what may occur if suitable habitat and forage bases are not present in the Tennessee River for the reintroduced fish. The overarching goal of this reintroduction is to establish a permanent population of Lake Sturgeon, both to restore a missing part of the natural ecosystem and as the future foundation of a recreational fishery and the economic boost that fishery would provide to the region. This reintroduction effort will require vast public and private resources to ensure success. I have provided a baseline assessment of what the population may look and behave like if adequate conditions are not provided for this population to succeed, and I encourage the managers involved in the restoration of this species in the strongest possible terms to incorporate my findings here, as well as future scientific investigations involving this population, in adaptive management as the reintroduction progresses.

In multiple instances I have called for additional data to supplement that which I have collected and analyzed in the previous chapters. The largest obstacle I faced during the collection of my data was of a logistical nature. It is an unfortunate case that natural resource managers, and particularly fisheries managers, have a much greater price to pay per datum than other fields flush with data. The price paid is in both monetary value (equipment, man-hours, logistics) and in time. The fastest reasonable solution to this issue would be greater investment in this type of research from interested parties. If the reintroduction of the Lake Sturgeon to the Upper Tennessee River is to be supplemented by scientific evidence, then more investment is required to offset the investment required of the researchers and their institutions. Novel, useful scientific insights can be had at exceedingly reasonable costs through the investment in future graduate student stipends and support. Looking further into the future, it may be that some of the logistical challenges I faced will be effectively overcome with the rapidly developing (and therefore rapidly cheapening) field of drones and autonomous vehicles. The capacity to conduct sonar survey transects from a central location in an occupied boat while parallel transects are being covered simultaneously by unmanned water craft would massively reduce the time (and therefore cost) needed to survey to saturation a particular area. The future research efforts that I call for should remain flexible to new technologies so that the price per datum that is paid to build on much of the baseline data I have provided in this dissertation is lowered, and that the investment that is bestowed on this work is allowed to stretch as far as possible.

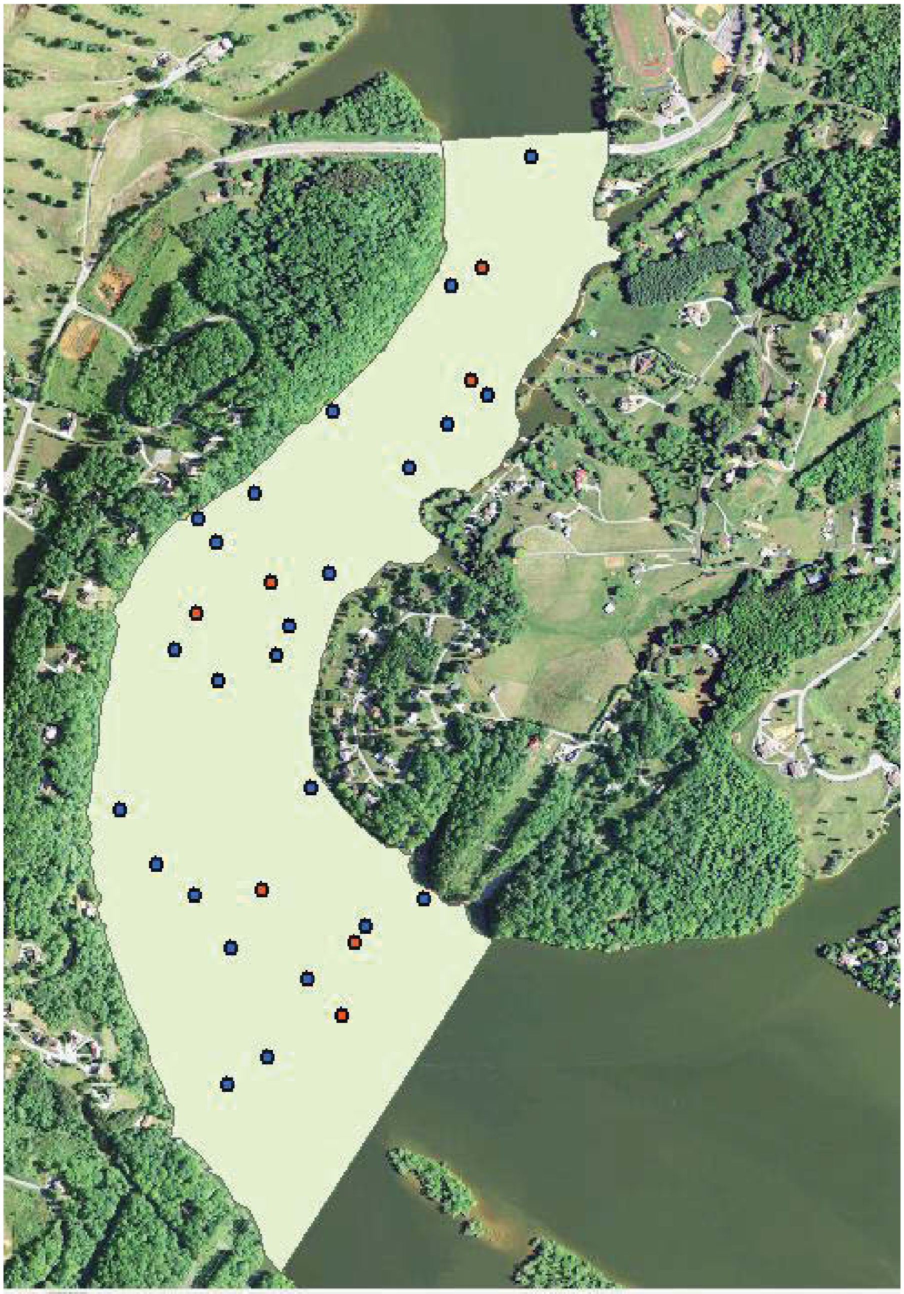
# APPENDICES

# APPENDIX F



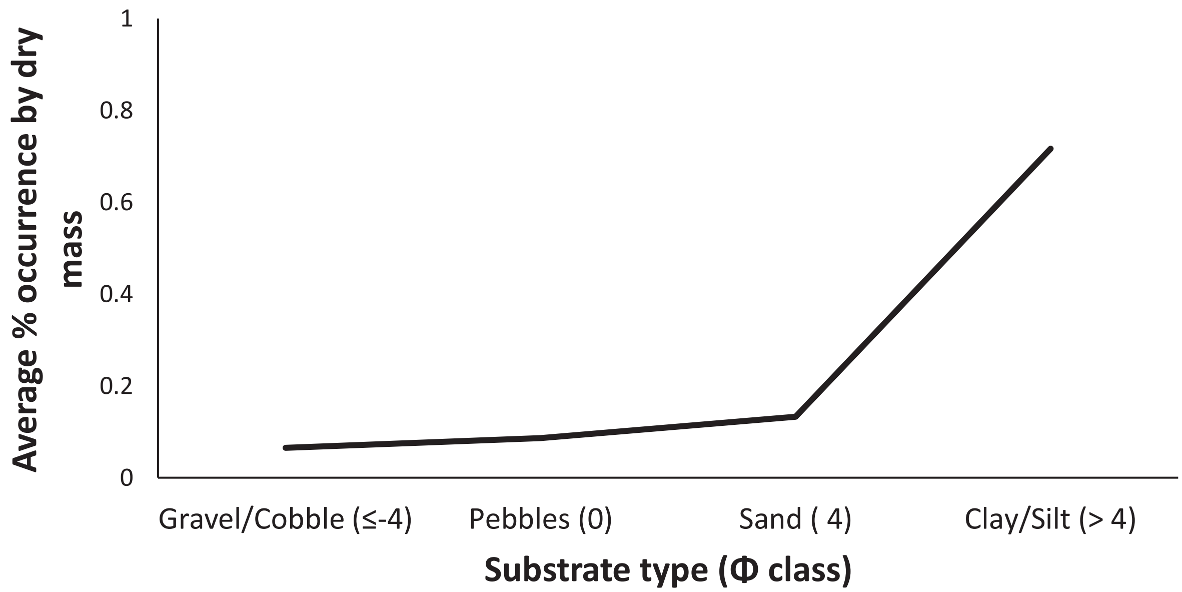

Lake Sturgeon detection locations (green dots) and substrate sampling locations (black dots) for characterization of Lake Sturgeon summer habitat substrate. Walker (2014), results unpublished.

**APPENDIX G**



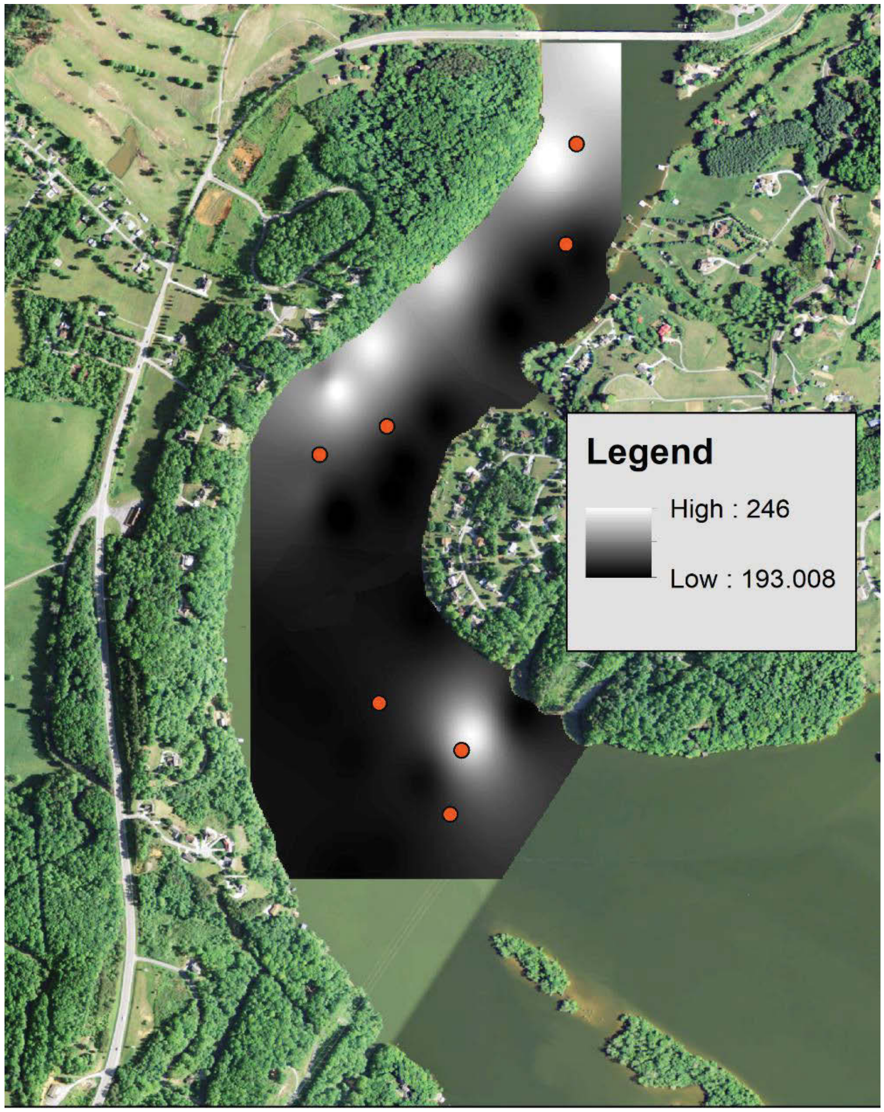
Polygon (light green) delineating Lake Sturgeon summer habitat area. Blue dots = BAS random sampling locations, orange dots = Lake Sturgeon locations.

# APPENDIX H



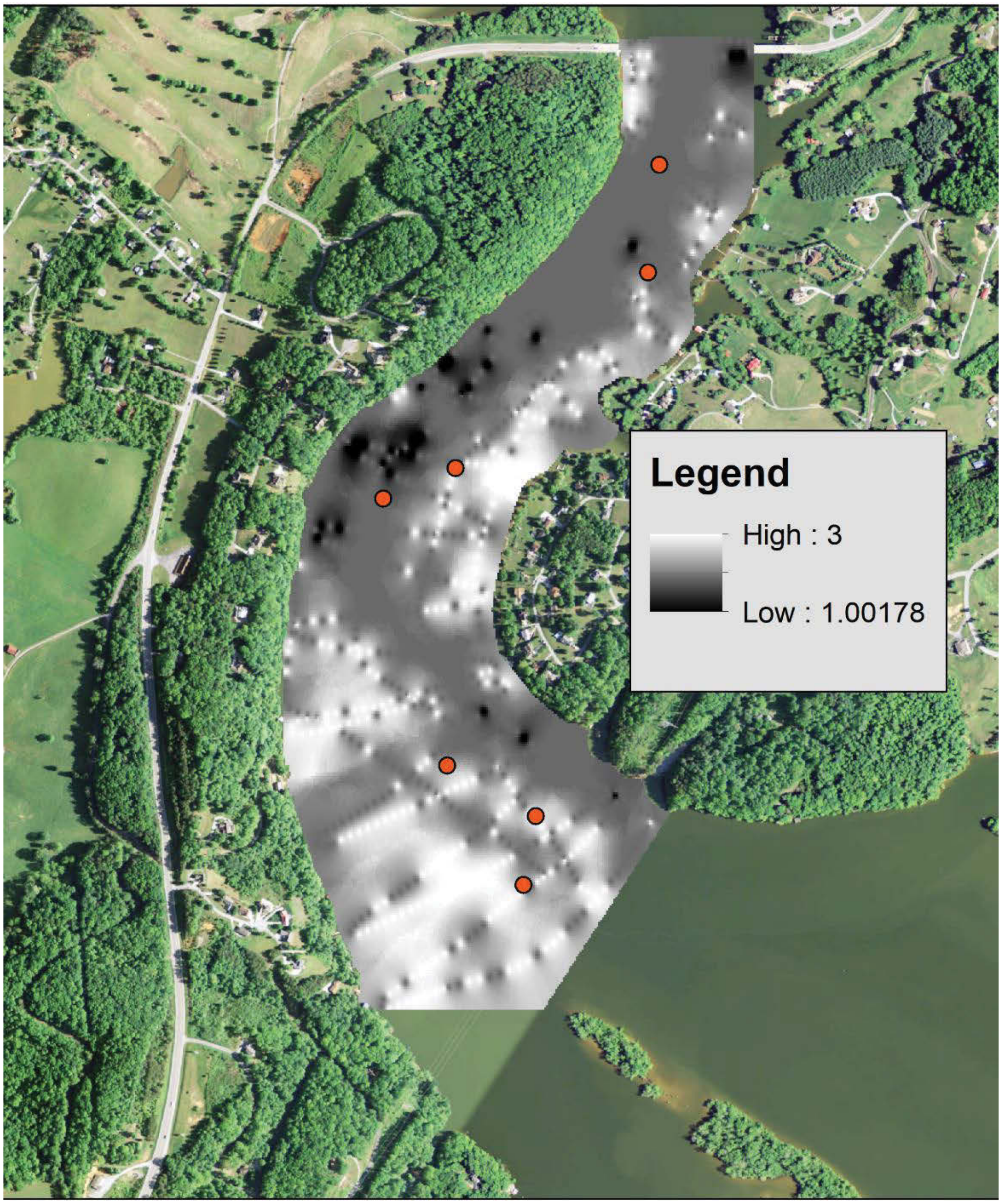

Average percent occurrence (by dry mass) of four substrate types in Lake Sturgeon locations sampled 2014. Walker (2014), results unpublished.

# APPENDIX I



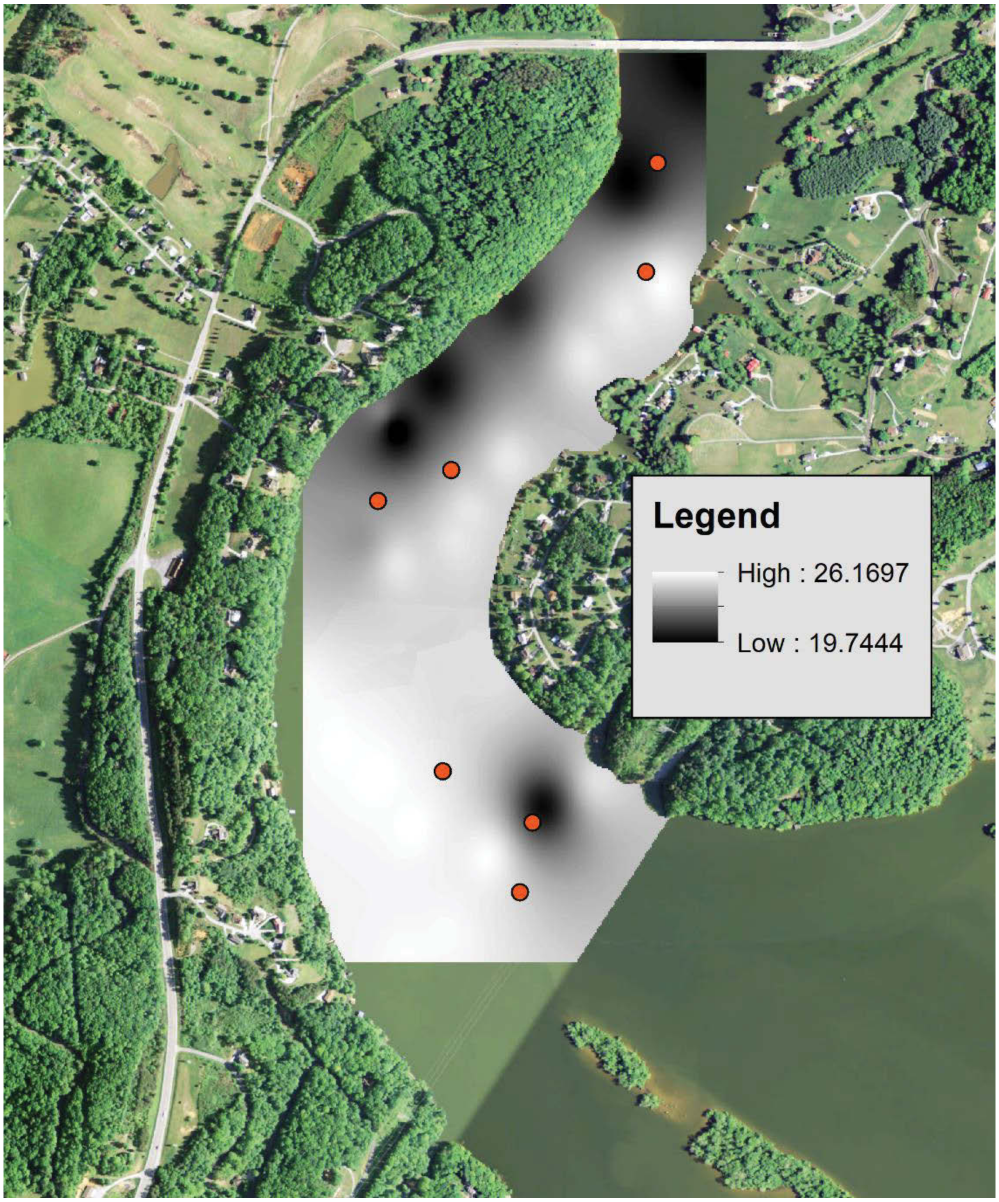

Example raster (conductivity, μS/m) generated with inverse distance weighting interpolation from the BAS assigned random sampling locations (blue dots, N = 24). Lake Sturgeon locations are indicated in orange.

# APPENDIX J



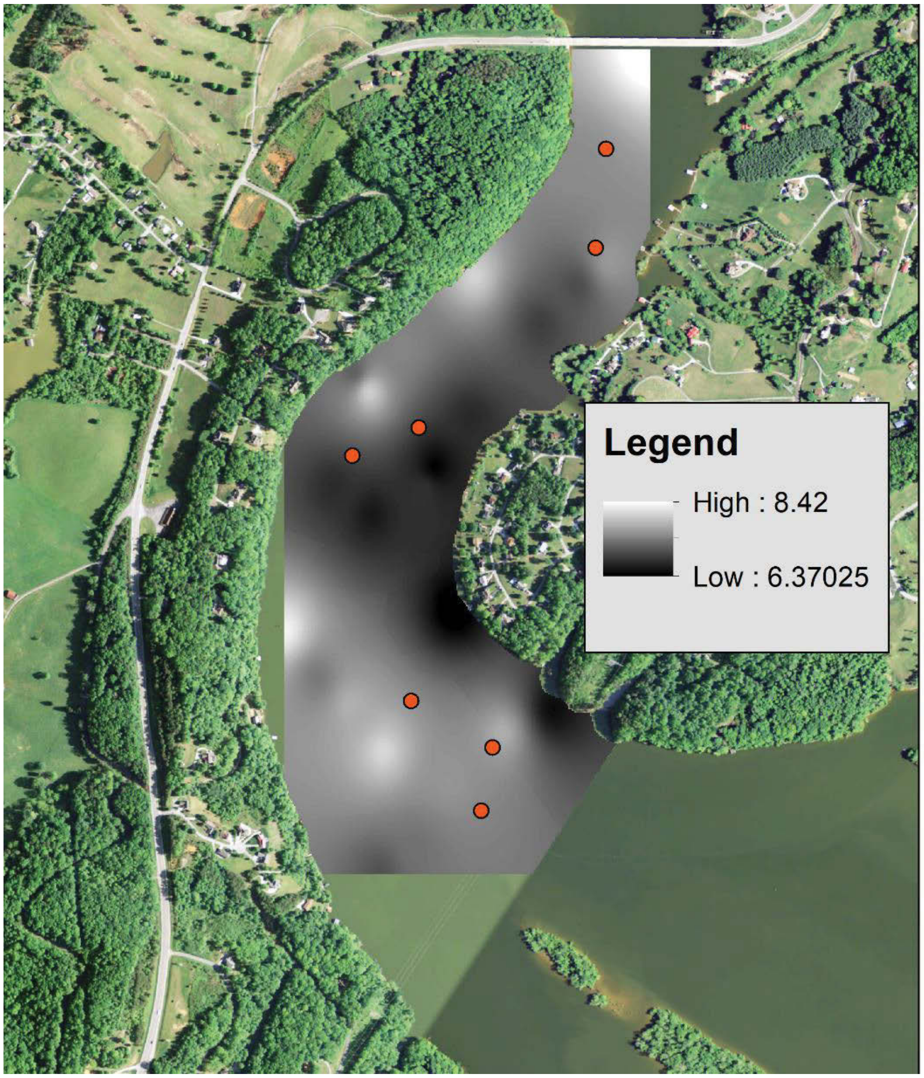

Example raster (substrate, 1 = rock, 3 = silt) generated with inverse distance weighting interpolation from the BAS assigned random sampling locations (blue dots, N = 24). Lake Sturgeon locations are indicated in orange.

# APPENDIX K



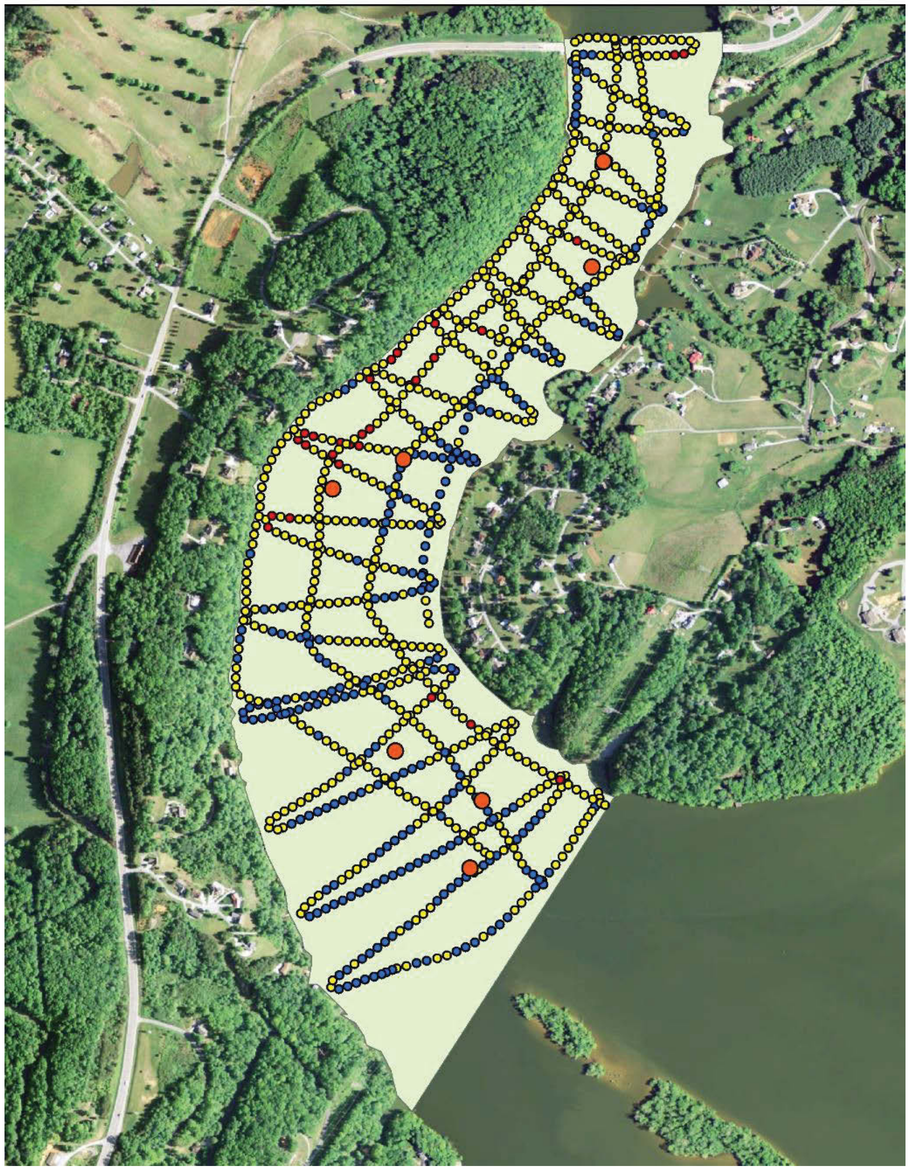

Raster (temperature, °C) generated with inverse distance weighting interpolation from the BAS assigned random sampling locations (blue dots, N = 24). Lake Sturgeon locations are indicated in orange.

# APPENDIX L




Raster (dissolved oxygen, mg/L) generated with inverse distance weighting interpolation from the BAS assigned random sampling locations (blue dots, N = 24). Lake Sturgeon locations are indicated in orange.3

# APPENDIX M



Distribution of Biosonics Echosounder survey tracks throughout the Lake Sturgeon summer habitat area. Each dot is the location of a single sonar ping, and colors correspond to substrate classification (red = rock, yellow = silt, blue = sand). Lake Sturgeon locations indicated by orange dots.

# VITA

Daniel James Walker was born in Roswell, Georgia, a suburb of Atlanta. He received his B.S. degree from Appalachian State University in Biology in May 2012. He next received his M.S. in Wildlife and Fisheries Science at the University of Tennessee in May 2014. He has dedicated his academic career to the study of fishes of conservation concern, ranging from darters in headwater streams to charismatic megafauna like the Lake Sturgeon in large rivers.