ASSESSING HYDRILLA OCCURRENCE AND SUBSEQUENT IMPACTS ON THE SPORTFISH COMMUNITY IN LAKE SINCLAIR, GEORGIA

by

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(Under the Direction of Martin J. Hamel)

ABSTRACT

Hydrilla is an invasive aquatic macrophyte that negatively impacts use of waterways for navigation, commercial fishing, irrigation, and hydroelectric power generation, as well as various aspects of sportfish community ecology. Hydrilla was introduced to Lake Sinclair, Georgia in 2018 and has spread rapidly throughout the reservoir, forcing managers to seek effective hydrilla assessment techniques, and to determine how the sportfish community is impacted from widespread hydrilla expansion. Obtaining accurate assessments of hydrilla coverage is imperative for the development of successful management strategies, and traditional assessment methods (i.e., line-transects and side-scan sonar) are expensive, time consuming, and often inaccurate. Furthermore, previous research on the impact of hydrilla on fish communities has been inconsistent across systems. Therefore, the objectives of this study were to 1.) develop satellite image classification techniques for identifying hydrilla, and 2.) examine changes in sportfish community dynamics and demographics following the hydrilla invasion.

INDEX WORDS: Remote Sensing, Sportfish Ecology, Invasive Species, Hydrilla

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by

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I dedicate this thesis to my parents. Your unconditional love and support, and your seemingly boundless wisdom have made me who I am today. I have you to thank for my continued success.

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CHAPTER 1

GENERAL INTRODUCTION AND STUDY OBJECTIVES

Introduction

Hydrilla (*Hydrilla verticillata*) is an aquatic macrophyte that negatively impacts anthropogenic use of waterways for navigation, commercial fishing, irrigation, and hydroelectric power generation. As a result, it is unpopular with stakeholders in the reservoirs where it is established (Langeland 1996). Additionally, hydrilla can impact various aspects of sportfish ecology, as well as how anglers interact with sportfish communities. However, there is no clear consensus on the nature of the relationship between hydrilla and sportfish, as previous research has documented both positive and negative impacts. For example, documented increases in the abundance of young-of-the-year and subadult sportfish have been observed in reservoirs where hydrilla is present (Tate et al. 2003), but as hydrilla density increases beyond a certain threshold sportfish growth and body condition begin to decline (Colle and Shireman 1980). Understanding the relationship between hydrilla and sportfish may inform fishery management decisions by identifying the systems or life stages in which fish are most susceptible to its impacts. A recent invasion of hydrilla in Lake Sinclair, GA has forced managing officials to consider how the sportfish community may be affected and pursue hydrilla management via herbicide application. Effective hydrilla assessment methods are needed to properly devise management strategies, but traditional assessment techniques such as side-scan sonar and transect surveys require high effort (Madsen and Wersal 2017). Technological advancements such as Unmanned Aerial Vehicle

(UAV), satellite imagery, and vegetation indices may provide faster and more accurate assessments of hydrilla's abundance (Everitt et al. 2007, Liira et al. 2010). My research seeks to determine the nature of the relationship between hydrilla and sportfish in Lake Sinclair, Georgia and to test the efficacy of remote sensing techniques for hydrilla assessment.

Background

Hydrilla is native to tropical areas of Asia but can also persist and spread in temperate climates. As a result of intentional and accidental introduction from the aquarium trade and hitchhiking on boats, hydrilla is now found on every continent except Antarctica (Cook and Lüönd 1982). Hydrilla grows when it is submersed and rooted in substrate, but pieces may break off and form free-floating mats. Additionally, hydrilla exhibits polymorphism; its physical appearance can vary depending on environmental conditions, and it can be monoecious or dioecious (Cook and Lüönd 1982, Verkleij et al. 1983, Pieterse et al. 1985). When hydrilla was first documented in the United States in 1960, it was observed in two locations in Florida; a canal near Miami and a spring near Crystal River (Blackburn et al. 1969). Since then, it has spread to freshwater lakes, ponds, rivers, and impoundments in 28 states, as well as Guam and Puerto Rico (Jacono et al. 2020).

The spread of hydrilla impacts various aspects of the ecology of sportfish. Feeding efficiency and body condition of popular sportfish such as largemouth bass (*Micropterus salmoides)*, black crappie *(Pomoxis nigromaculatus)*, bluegill (*Lepomis macrochirus)*, and redear sunfish (*Lepomis microlophus*) have been shown to decrease as hydrilla density increases (Colle and Shireman 1980, Morrow et al. 1991). Other research suggests that the presence of hydrilla does not impact fish community metrics such as diversity and abundance (Killgore et al. 1989, Hoyer et al. 2008).

Hydrilla, like many aquatic macrophytes, provides a variety of essential services to sportfish (e.g., largemouth bass) in lacustrine systems. The complex structure formed by their stems and leaves provides valuable spawning habitat (Chew 1974), shelter from predators (Savino and Stein 1982), and habitat for invertebrates that young sportfish prey on (Moxley and Langford 1982). As a result, an increase in abundance of young-of-the-year and subadult largemouth bass has been shown in systems where hydrilla is present (Tate et al. 2003, Bonvechio and Bonvechio 2006, Nagid et al. 2014). Similarly, Barrientos and Allen (2008) reported that when compared to two species of native aquatic macrophytes at similar densities, hydrilla supported the highest fish biomass and was not associated with any significant variation in species composition. Hydrilla also provides habitat for macroinvertebrates (Carniatto et al. 2014) and smaller forage fish such as bluegill which are a staple in the diets of subadult and adult largemouth bass (Barnett and Schneider 1974). Scientists and anglers alike generally accept that largemouth bass populations are dependent on aquatic macrophytes such as hydrilla.

The benefits of hydrilla on sportfish populations may change spatially. For example, Hoyer and Canfield (1996) performed an empirical analysis of largemouth bass and aquatic vegetation in Florida lakes. This study demonstrated a weak positive correlation between aquatic macrophyte abundance and the abundance and growth of young-of-the-year and subadult bass, but not adults. A study conducted in Texas suggested that reductions of aquatic plant coverage in a system below 20% can result in a reduction of largemouth bass abundance (Durocher et al. 1984).

The presence or removal of hydrilla may also impact movement dynamics of sportfish. For example, photorespiration in dense stands of hydrilla can drop dissolved oxygen to hypoxic levels at night, resulting in a decline in habitat quality that may force fish to move out of these areas (Bradshaw et al. 2015). Additionally, largemouth bass demonstrated greater diel movements, inhabited greater depths, and began using woody debris for cover following removal of hydrilla (Sammons et al. 2003).

While the impacts of aquatic macrophyte density on sportfish abundance are generally neutral or positive, the same cannot be said for sportfish growth. Maceina and Slipke (2004) and Olson et al. (1998) suggest that largemouth bass foraging efficiency is highest at intermediate aquatic macrophyte densities, and multiple studies have reported that dense submersed aquatic vegetation is related to reduced sportfish growth and body condition (Buck et al. 1975, Colle and Shireman 1980). The hypothesized mechanism for this is that as structural complexity of a habitat increases, predation success tends to decrease (Glass 1971, Stein and Magnuson 1976).

Colle and Shireman (1980) suggested a threshold response where aquatic macrophytes such as hydrilla become a hinderance to sportfish feeding efficiency at high density, leading to decreased growth and body condition. Specifically, the authors examined how the condition factor — the relative robustness of a fish — changed for largemouth bass, bluegill, and redear sunfish at varying densities of hydrilla in two Florida lakes. *Lepomis* spp. did not exhibit declines in condition until hydrilla mats were observed on ~80% of the reservoir. However, body condition of various largemouth bass size classes significantly declined in relation to hydrilla density.

Aquatic vegetation abundance also likely affects the preferred prey of largemouth bass, as the shift to piscivory occurs at smaller sizes in reservoirs where hydrilla removal efforts are present (Shelton et al. 1979, Bettoli et al. 1992). If hydrilla delays the onset of piscivory, then growth, body condition, and survival of age-1 fish may decline (Olson 1996).

In addition to its ecological impacts on sportfish communities, hydrilla can alter the interactions between recreational anglers and sportfish populations by significantly increasing the difficulty of boat navigation in an affected body of water and creating difficult fishing conditions with decreased catch rates (Langeland 1996). For example, Orange Lake, in North Central Florida, had a reputation as an excellent place to target largemouth bass, black crappie, bluegill, and redear sunfish and attracted large numbers of local and non-local anglers (Colle et al. 1987). Hydrilla was first observed in the lake in 1974 and within two years, hydrilla coverage reached 80%, causing an 85% decline in angler effort. The benefit to cost ratio of angler expenditures to aquatic plant control costs declined from 31:1 to 0.3:1. Similarly, when hydrilla coverage reached 65% surface area in Lake Seminole, Georgia, angling effort and total catch decreased by over 36% (Slipke et al. 1998). Conversely, O'Rouke (2009) indicated a positive correlation between hydrilla coverage and angler success (fish caught per hour) on Lake Tohopekaliga, FL, suggesting that declines in angler success associated with hydrilla may be due to a learning curve involved with fishing around it. Collectively, recreational freshwater fishing generates over $86 billion in overall economic output in the United States and produces more than $11 billion in state and federal tax revenues (ASA 2020). Decreases in angler purchasing can therefore have significant economic impacts to local and state economies.

Rationale and Significance

Several factors contribute to the observed ambiguity in the effects of hydrilla on fish populations. For example, fish population dynamics naturally fluctuate related to cyclical weather patterns (Kanno et al. 2015), water management (Eklund 1994), recruitment variation (Booth and Brosnan 1995), and angler harvest (Parkinson et al. 2004). Combining long-term data sets with nearby control systems may be a valuable method to examine causal relationships between hydrilla and fish population dynamics.

To improve our understanding of the impacts of hydrilla on fish communities, we must use effective techniques to assess its distribution, coverage, and biomass. Current assessment methods such as transects, rake-toss surveys, and biovolume estimates using sonar vary greatly in their complexity and comprehensiveness. Each method has its own limitations. Transects and rake toss surveys are cost-effective methods but are time-consuming, largely qualitative and provide little information on how dense hydrilla is. Additionally, sonar assessments become ineffective when hydrilla exceeds a certain density or becomes emergent (Madsen and Wersal 2017). More modern technologies may provide new opportunities to compare the efficacy of novel hydrilla assessment techniques to that of more traditional methods. Novel methods such as unmanned aerial vehicles and satellite imagery have the potential to be more efficient and effective. Reliable assessment methods are the key to developing an effective management strategy for improving the overall health of sportfish standing crops (Olson et al. 1998) and maintaining the recreational value of affected reservoirs (Weber et al. 2020). Thus, the objectives of my study were to: 1) compare changes in community metrics for several sportfish species (i.e., largemouth bass, black crappie, redear sunfish, and bluegill) before and after the introduction of hydrilla in Lake Sinclair, and 2) compare the efficacy of remote sensing techniques (i.e., drone and satellite imagery) for hydrilla assessment to that of traditional methods such as sonar biovolume estimates.

This study is the first examination of hydrilla’s impacts on sportfish communities in Lake Sinclair and aims to provide baseline data for future management of this system, as well as to determine the most accurate and cost-effective methods for Georgia Power’s future hydrilla assessment efforts.

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CHAPTER 2

EMPLOYING REMOTE SENSING TECHNIQUES FOR MONITORING HYDRILLA OCCURRENCE

Introduction

Numerous approaches to aquatic plant assessment exist that vary substantially in cost, accuracy, and effort required. Point intercept surveys are adequate for determining where a species is present and absent, while line transect methods can be used to estimate species distribution. However, neither of these methods provides an accurate coverage or biovolume estimate, which is a shortcoming because biovolume estimates are required for the calculation of herbicide application dosages and stocking densities for biological control species such as triploid grass carp (*Ctenopharyngodon idella*). Biovolume estimates are often obtained via physical assessments (e.g., plant rake coring or dredges) (Madsen and Wersal 2017), but these methods require very high effort. Side-scan sonar provides a lower-effort way to obtain biovolume estimates and is widely used by managers to perform aquatic vegetation assessments (Madsen and Bloomfield 1993, Silva et al. 2008).

Remote sensing is an emerging assessment tool that has been historically dismissed as having low sensitivity and being cost prohibitive; however, more recent work has shown remote sensing to be an efficient and effective technique for emergent and floating aquatic plant assessments (Everitt et al. 2007, Liira et al. 2010, Midwood and Chow-Fraser 2010). Recent technological advancements, such as 3-meter resolution satellite imagery and affordable unmanned aerial vehicles (UAV), have been combined in a Geographic Information System (GIS) to estimate volume and coverage of various vegetation types in terrestrial environments (Housman, Chastain, and Finco 2018, Li et al. 2019). However, the utility of this approach in aquatic environments is unknown.

In recent years, interest in the commercial use of small UAVs has grown (Freeman and Freeland 2015), particularly for agricultural and forestry applications (Grenzdörffer et al. 2008). Small UAV systems have been used successfully for forest fire detection and monitoring (Casbeer et al. 2005, Merino et al. 2006), forest disease mapping, and quantification of dendrometric parameters such as canopy height when equipped with lidar sensors (Torresan et al. 2017). In the agricultural industry, UAVs have proven useful for assessing soil water content (Gago et al. 2015), nuisance weed assessments for precise herbicide applications (Ehsani and Maja 2013), and harvest logistics optimization (Grenzdörffer et al. 2008).

Though most commonly used for agriculture and forestry applications, the versatility and functionality of UAVs has also made them a valuable tool in marine settings for commercial and research purposes. For example, UAVs in combination with machine learning proved effective for rapidly estimating density, biomass, and size distributions of commercially important edible jellyfish (Raoult and Gaston 2018). Similarly, researchers have had success using UAV technology to obtain density estimates for ecologically important sea cucumbers (Kilfoil et al. 2020) and several elasmobranch species (Kiszka et al. 2016) in shallow lagoons.

Satellite imagery analysis is the dominant remote sensing technique and has traditionally been useful for long-term aquatic plant monitoring and large-scale coverage estimates. This technique requires minimal effort but is considered to have low sensitivity and is not plant species-specific. In addition, it has historically been considered one of the more expensive methods, given that images and specialized analysis software – requiring trained personnel – must be purchased (Madsen and Wersal 2017). The newer developments of subscription-based satellite imagery services and intuitive software may change this perspective, allowing for coverage estimates to be obtained from a personal computer in a fraction of the time required to obtain an estimate from the ground, and without the costs associated with field equipment use.

Recent studies have indicated that modern high-resolution satellite imagery analysis with GIS can identify different crop types (Yang et al., 2011) and discern between stands of hardwood and pine trees (Blinn et al., 2012). However, the smaller size of aquatic plant patches and reflectance introduced by water makes species-specific monitoring with satellite imagery difficult in aquatic systems. A two-pronged approach to aquatic vegetation monitoring that involves detection with satellite imagery and confirmation of species with remote UAV systems may provide managers with a quick, effective, and economical means to assess and monitor aquatic plants.

Hydrilla (*Hydrilla verticillata*) is an invasive aquatic plant that was introduced into the United States in 1960. Hydrilla negatively impacts anthropogenic use of waterways for navigation, commercial fishing, irrigation, and hydroelectric power generation, and as a result is unpopular with many stakeholders in reservoirs where it is established (Langeland 1996, Pimentel et al. 2000). Managing hydrilla is also costly. Between 1982 and 2012, state and local agencies in Florida spent approximately $750 million on the management of hydrilla (Madsen and Wersal 2017). As hydrilla and other invasive aquatic plants continue to spread, the importance of understanding the dynamics of aquatic vegetation in a system is increasing. Therefore, the objectives of this chapter are to 1) Compare the efficacy of novel hydrilla assessment methods (i.e., satellite and UAV imagery) to that of side-scan sonar, and 2)

Determine if modern UAV imagery can differentiate between other non-hydrilla species of aquatic plants.

Materials and Methods

Study Site

Lake Sinclair (Figure 2-1) is a large reservoir on the Oconee River in central Georgia, spanning Baldwin, Hancock, and Putnam counties. It has a surface area of 6,204 hectares, a maximum depth of 27 meters, and 671 kilometers of irregular shoreline with many coves and tributaries. Lake Sinclair was created by the Georgia Power Company in 1953 and is used for hydroelectric power generation and recreation. The reservoir is an oligotrophic system characterized by sandy or rocky bottom and scarce aquatic vegetation. (Smith and Manoylov 2013).

Sampling to test the efficacy of hydrilla assessment techniques was conducted in a cove (referred to hereafter asShoulderbone Cove) in the northernmost extent of Lake Sinclair in 2020 (Figure 2-2). This 88 ha cove was selected due to its abundant hydrilla, lack of non-hydrilla species that may add uncertainty to initial assessment, and lack of previous herbicide application.

Data Collection

Site visits to Shoulderbone Cove occurred on July 17 and October 16 of 2020. Hydrilla germination during 2020 was delayed throughout the lake due to high turbidity. July was selected as it was immediately following germination, whereas October was selected because it was late in the growing season when hydrilla was fully emergent but prior to winter die off.

Unmanned Aerial Vehicle imagery and side-scan sonar assessments were conducted during each site visit. Satellite imagery of the cove on both July 17 and October 16 of 2020 were obtained from the commercial imagery service Planet Labs (www.planet.com). The UAV was flown along the shoreline of Shoulderbone Cove at an altitude of ~ 46 meters and a speed of 11 kilometers per hour, taking a photo every 3 seconds. UAV flight paths were plotted using DJI Ground Station Pro and performed with a DJI Phantom 4 Pro equipped with a 1" RGB CMOS Sensor and an 8.8mm-24mm lens. The length of each flight was determined by battery life, and the UAV was recalled for a battery change once it reached ~30% to avoid loss of control or crashes. Rake toss surveys were performed each time the UAV battery was changed to confirm that the species being observed was hydrilla. All UAV images were then georeferenced using AgiSoft Metashape Pro (AgiSoft, 2016) before being loaded into GIS for analysis.

Side-scanning was performed using a Lowrance HDS9 head unit and a Lowrance 3-in-1 transducer. BioBase Software's Habitat+ Plan (Navico 2014) was used to generate spreadsheets of aquatic plant density from side-scan data in terms of percentage of the water column occupied by hydrilla at georeferenced points. A scholastic subscription to Planet Labs' Planet Explorer was used to obtain 3-m resolution surface reflectance Dove satellite images of Shoulderbone Cove. Surface-reflectance imagery was selected specifically because we were interested in reflectance values (pixel values) associated with hydrilla and other cover types. Images of Shoulderbone Cove from September of 2018, 2019, and 2020 were also collected.

Data from a Georgia Department of Natural Resources (GADNR) shoreline vegetation survey were used to locate sites with other species of aquatic vegetation present and to test the potential of UAV imagery for differentiating between species. The aquatic plant species evaluated include water willow (*Justicia americana*), *Lyngbya*, and Southern naiad (*Najas guadalupensis*), and instances of emergent and submerged hydrilla.

Data Analysis

Supervised pixel-based image classifications were used as an empirical method for determining hydrilla identification in satellite imagery. All analyses were performed in ArcGIS Pro (ESRI, 2021) and RStudio (R Core Team, 2021). Prior to analysis, all satellite images underwent spectral transformation via Normalized Difference Vegetation Index (NDVI)

$$NDVI = \frac{NIR - Red}{NIR + Red},$$

where *Red* is associated with the third band in the satellite imagery (620-750 nm) and *Near-Infrared (NIR)* is the fourth band (760-850 nm). This transformation was used to maximize sensitivity to vegetation and minimize the influence of external factors such as background reflectance and atmospheric effects (Fang et al. 2003). The NDVI transformation yields pixel values between -1 and 1, with values approaching 1 indicating the highest density of green vegetation. The NDVI was selected for this analysis because it has proven to be robust under a range of atmospheric conditions (Gamon et al. 1995). One-hundred and twenty equally sized training samples were created as a polygon layer split across three cover types: water, hydrilla, and forest (to account for overhanging tree limbs). A polygon-level zonal statistics table containing a numerical ID, cover type, and mean pixel value for each polygon was then generated and exported for analysis. Pixel values associated with the three cover types were significantly different (F = 4,998; df = 2; p = <0.001; Figure 2-3), thus training samples could be used to train a supervised pixel-based image classification. Randomly generated accuracy test points, a confusion matrix, and georeferenced UAV images were used to test the accuracy of the classification. The confusion matrix uses the randomly generated points to determine the rate at which misclassifications occur, and what type of error causes each instance of misclassification.

Heat maps of the biovolume estimates obtained from side-scan sonar in July and October were created using the outputs from BioBase. Areas indicated as hydrilla by side-scan and image classification were digitized to obtain coverage estimates in hectares. This was done by loading the BioBase data into ArcGIS Pro, splining the kriging grid created by the georeferenced points to smooth it, and then clipping to the lake boundary (Navico, 2014). The symbology could then be adjusted so that different densities were represented by different colors in the map.

The UAV images of different vegetation species were processed using similar methods to satellite images. However, a different vegetation index was used because the UAV images only contained red, green, and blue bands and did not reflect the near-infrared band of the light spectrum. Instead, the Coloration Index (CI), which uses red and blue spectral bands, was applied:

$$CI = \frac{(Red - Blue)}{Red},$$

where *Red* is associated with the first band of the UAV imagery (620-750 nm) and *Blue* is associated with the third (450-495 nm). Training samples were then created, and zonal statistics tables were exported for analysis.

Results

In July 2020, hydrilla had begun to grow in Shoulderbone Cove but had not yet reached its peak density or grown enough to form mats on the water’s surface. On this sampling event, side-scan sonar analysis resulted in an estimated coverage of 6.06 ha of hydrilla (Figure 2-4), while satellite imagery classification indicated only 3.64 ha (Figure 2-5), but cross referencing with UAV imagery revealed that these were overwhelmingly false detections in which the classification misidentified shadows cast by trees as hydrilla. In October 2020, hydrilla was emergent throughout Shoulderbone Cove and was ubiquitous throughout the entire water column. At this time, side-scan sonar imagery detected 3.43 ha of hydrilla in the cove (Figure 2-6), while satellite imagery classification indicated 9.35 ha of hydrilla (Figure 2-7). Hydrilla was visually confirmed to be present in large quantities in several locations but was completely undetected by side-scan. This lack of detection may be due to a combination of extremely dense stems and leaves and the sloping bathymetry of the cove preventing sonar from "bouncing back."

To test the accuracy of the satellite image classification, we employed UAV imagery and the Create Random Points function of ArcGIS Pro. To limit these points to the area surveyed with the UAV, a 20 m buffer was created inside the shoreline of Shoulderbone Cove. Two-hundred random points – a minimum of 10 m apart - were created inside this buffer. Each random point was then buffered by 3 m to account for the resolution of the satellite imagery, and the image classification at each point was then compared to UAV imagery using a confusion matrix to determine the rate of agreement between the image classification and UAV imagery. The confusion matrix indicated that the image classification identified hydrilla 96% of the time, with non-detections (indicating that there is not hydrilla where there is) occurring 4% of the time. When the classification identified a cover type as hydrilla, it was correct 87% of the time, with false-detections (indicating that hydrilla is present where it is not) occurring 13% of the time (Table 2-1).

Analysis of the multi-plant species UAV images indicated that the pixel values associated with emergent hydrilla were significantly different from those of all other species, including water willow – an emergent species (F = 82.29; df = 4; p = <0.0001; Figure 2-9). Post-hoc comparisons using Tukey's honestly significant difference test indicated a difference in the pixel values of *Lyngbya* sp. and submerged hydrilla (diff = -0.107; p-adj = 0.009), but southern naiad and submerged hydrilla were virtually indistinguishable from one another (diff = -0.0164; p-adj = 0.983; Figure 2-6).

Discussion

Remote sensing techniques have strong potential for use in aquatic plant assessments. Satellite imagery was used to train an 83% accurate image classification for hydrilla detection in our study system. Given the high level of accuracy, this technique will be an excellent tool for long-term monitoring, as the same classification schema and training samples can be used repeatedly for the study area through time. However, it should be noted that these techniques cannot be reliably applied unless hydrilla is emergent.

UAV imagery shows promise for identifying emergent hydrilla and differentiating it from other species. With further testing on a reservoir with more abundant emergent vegetation, these methods could likely be refined further. However, submerged vegetation poses a challenge for species identification; the pixel values associated with submerged hydrilla and southern naiad were virtually identical, making it unlikely that a classification could accurately differentiate between the two. Because water strongly absorbs red light, the Coloration Index that uses only the red and blue bands is ineffective. However, if a UAV was equipped with a sensor with hyperspectral capabilities, other vegetation indices such as NDVI or the Green Difference Vegetation Index (GDVI) could be explored.

Remote sensing techniques require much lower effort than traditional techniques. Untrained individuals took 45 to 60 minutes to generate a satellite image classification when provided with instructions. For data collection effort, it takes approximately 90 minutes to conduct a UAV flight of Shoulderbone Cove, while by comparison it takes approximately 240 minutes to survey the same area using side-scan sonar. However, all hydrilla assessment techniques have trade-offs. Although satellite imagery is more efficient, it lacks the sensitivity to distinguish between different aquatic plant species, so ground truthing is required for the initial training of an image classification. Additionally, satellite imagery is ineffective when hydrilla is not yet emergent and cannot estimate biovolume, so these techniques should not be used to attempt to calculate herbicide application dosages or stocking densities for biological control species.

Side-scan sonar is a more efficient way of obtaining biovolume estimates (Silva et al. 2008, Greene et al. 2018), but it becomes ineffective when vegetation density surpasses a certain threshold. Additionally, side-scan sonar cannot identify aquatic plant species, and as such, must be accompanied by validation techniques if this information is needed. Side-scan sonar is also limited by boat navigation. Two areas on the north side of Shoulderbone Cove were too shallow to be accessed by boat and no data were collected despite satellite imagery indicating that hydrilla was present.

A two-pronged approach using high resolution satellite and UAV imagery may provide the best approach for future hydrilla monitoring and coverage assessments. Satellite imagery is highly effective at detecting emergent vegetation, meaning that it can be used to quickly identify areas where aquatic vegetation had not occurred in previous years but is now present. Coupling this information with UAV imagery that can differentiate between emergent plant species (e.g., hydrilla and water willow) would make for a more efficient method of identifying new instances of potential hydrilla spread and performing the necessary ground truthing. This development would allow lake managers to identify new infestations more quickly, and expeditiously make informed management decisions.

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Table 2-1: Confusion matrix created using randomly generated points (n = 200) to test the accuracy of satellite image classification. UAV imagery was used to verify or refute the classified cover type at each point.

	Satellite: Hydrilla not present	Satellite: Hydrilla present
UAV: Hydrilla not present	96%	13%
UAV: Hydrilla present	4%	87%

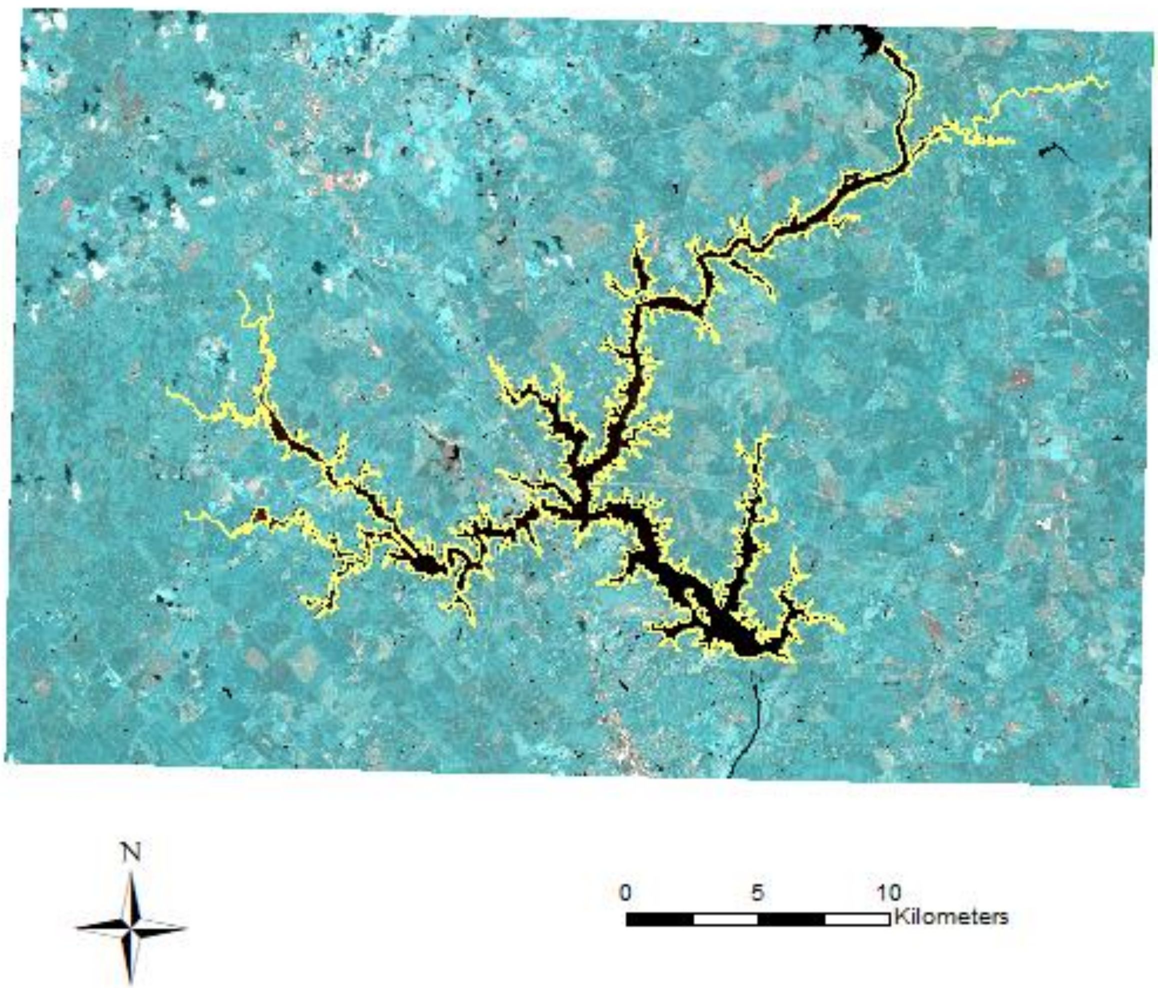

Figure 2-1. Map of Lake Sinclair, GA.

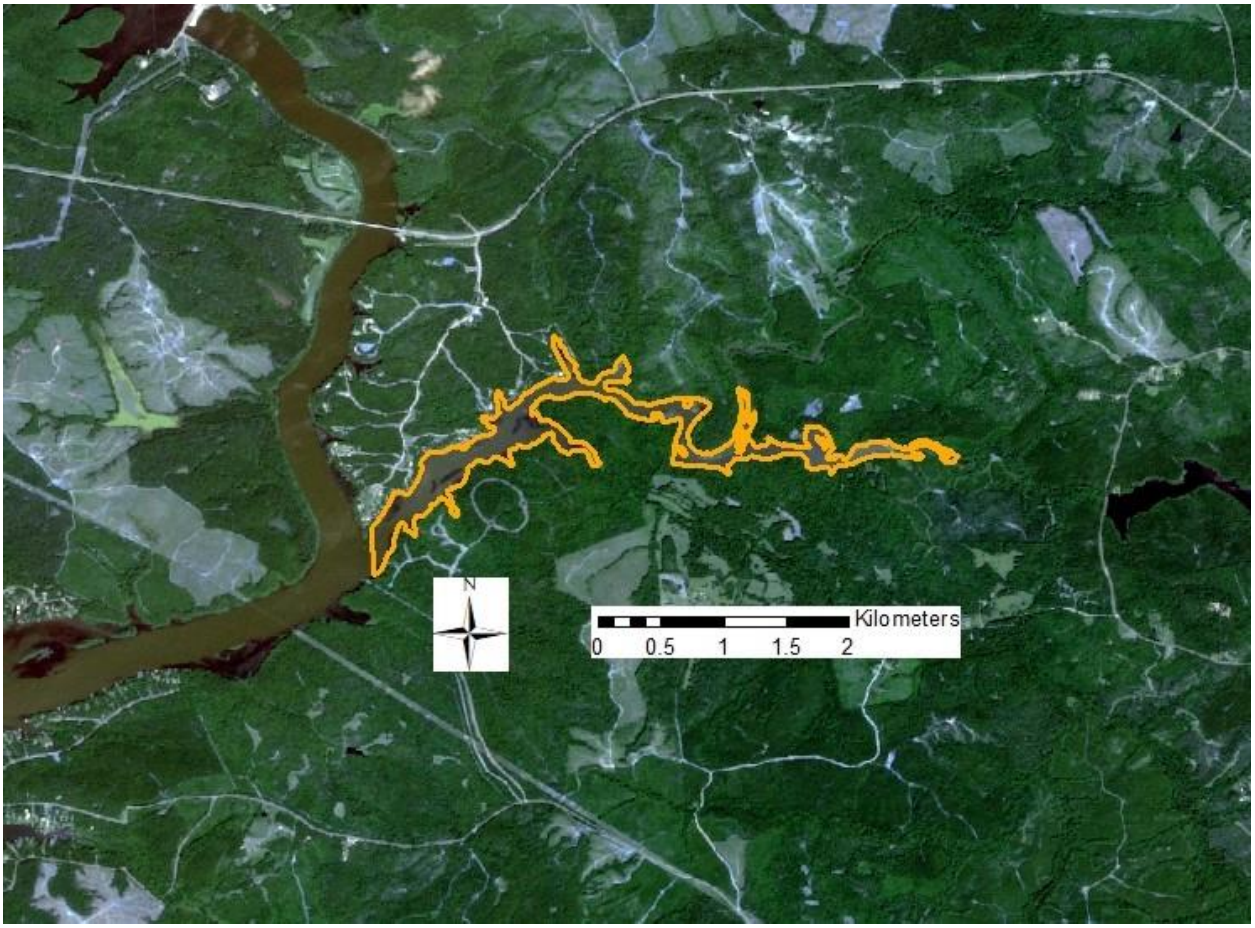

Figure 2-2. Map of Shoulderbone Cove, the 88 ha study site located in the northernmost extent of Lake Sinclair, GA.

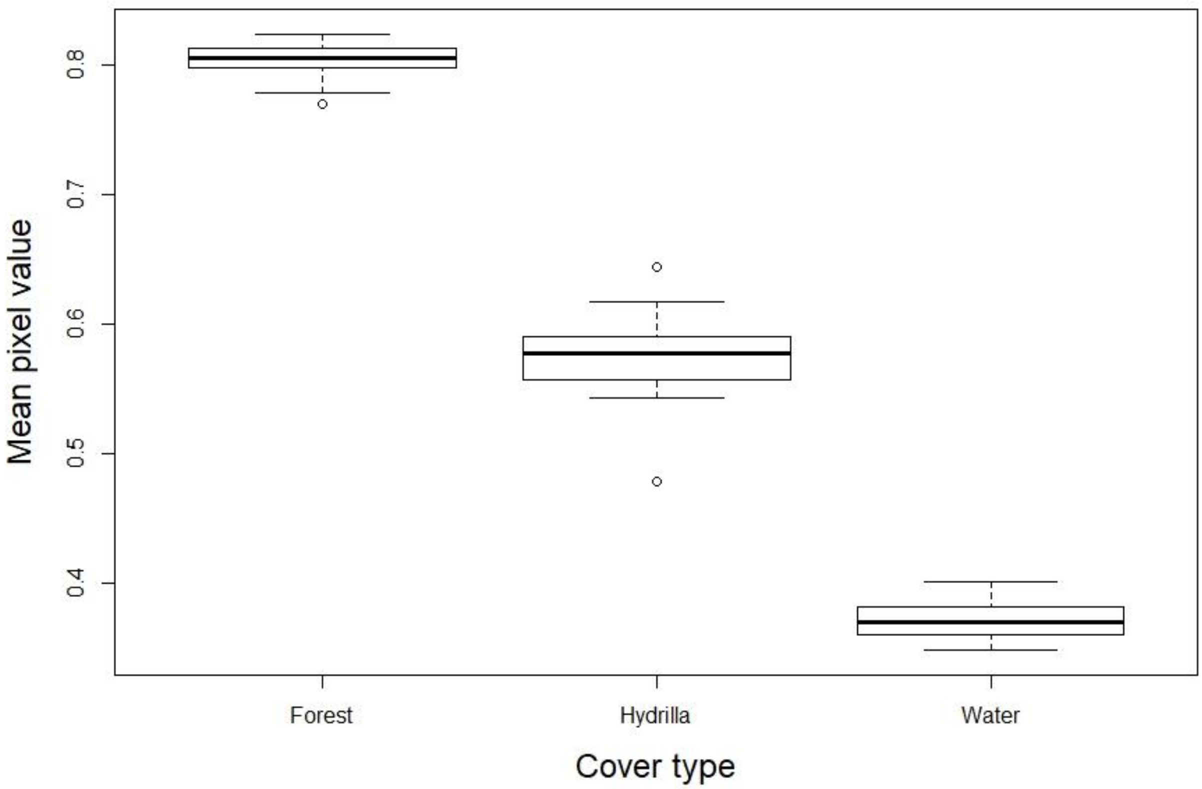

Figure 2-3: Boxplot of mean normalized difference vegetation index (NDVI) pixel values for sample polygons across 3 cover types.

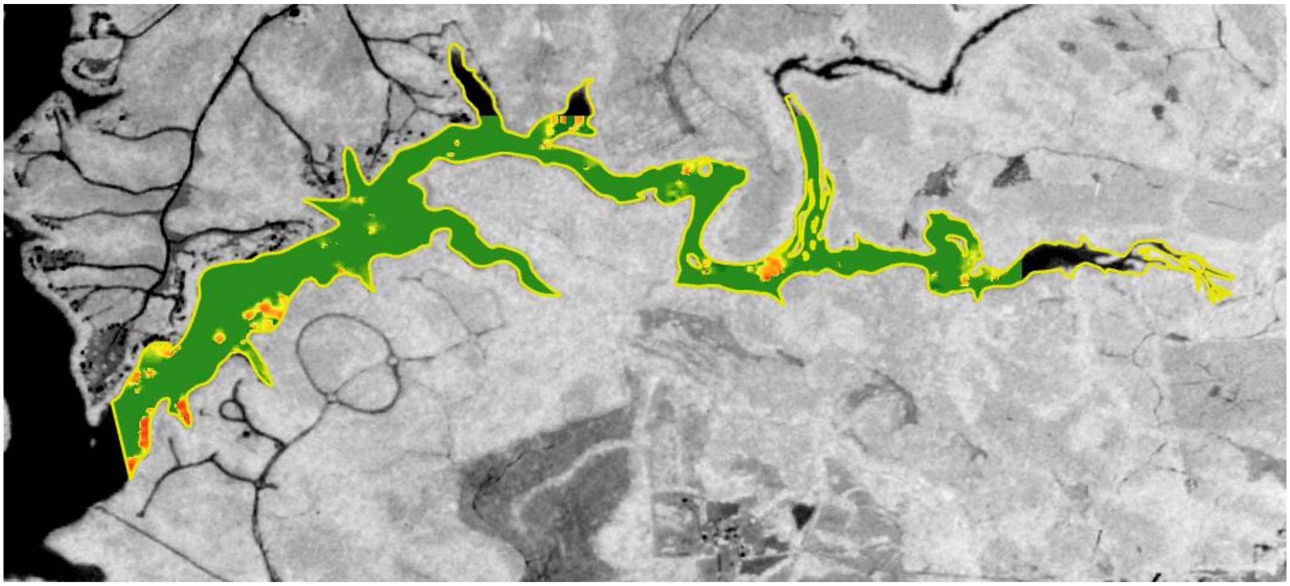

Figure 2-4: Side-scan sonar heat map of hydrilla occurrence in Shoulderbone Cove; July 2020. Green represents areas with little to no aquatic vegetation growth, yellows and oranges indicate areas with intermediate aquatic vegetation growth, and red indicates high aquatic vegetation growth.

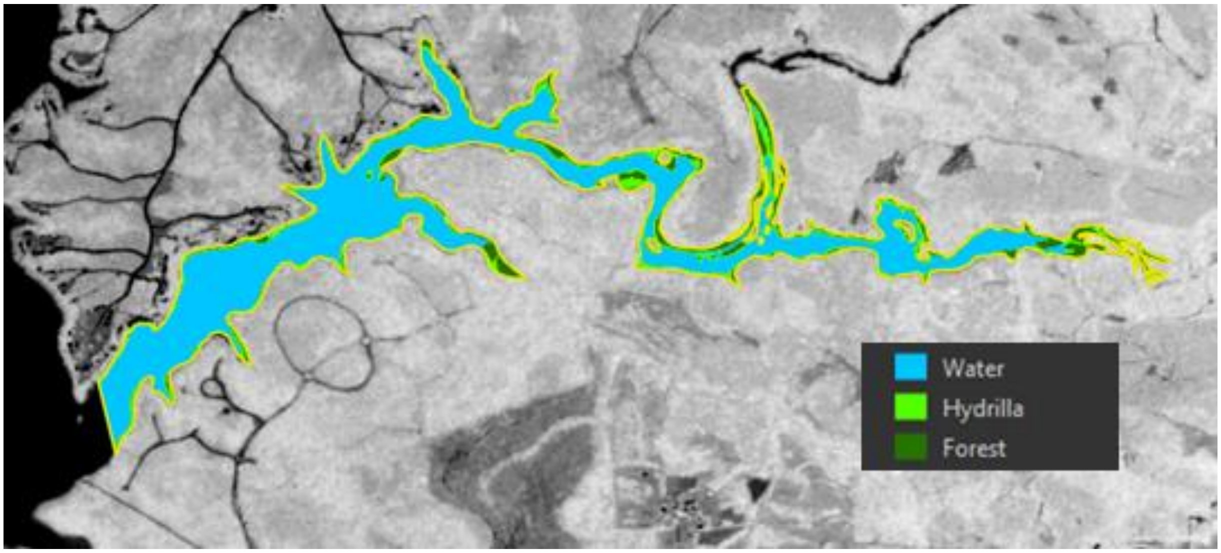

Figure 2-5: Satellite image classification for hydrilla occurrence in Shoulderbone Cove in July 2020.

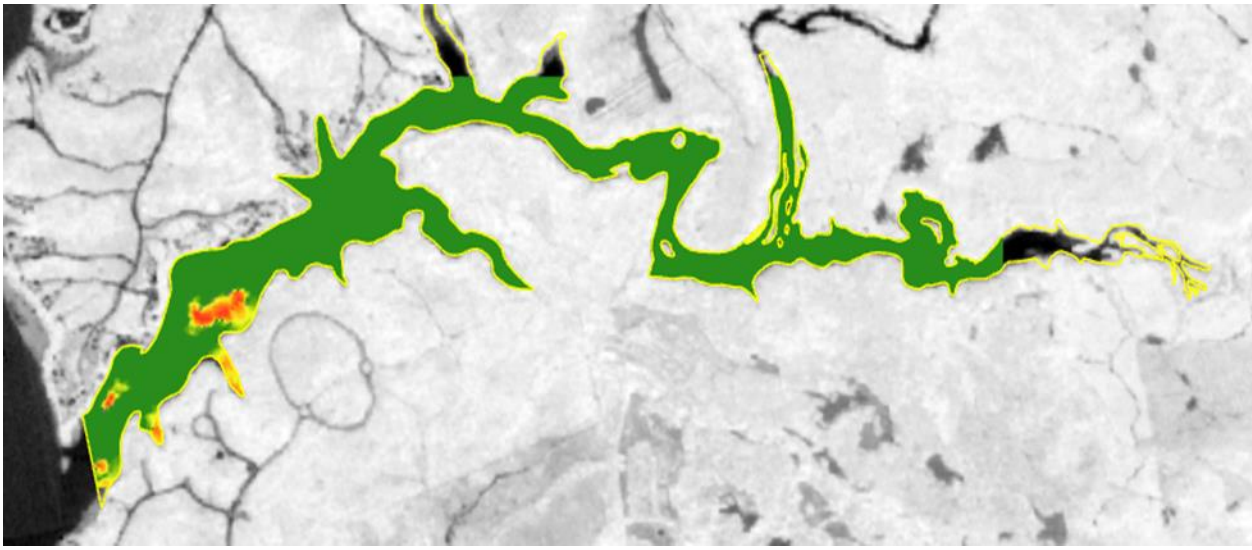

Figure 2-6: Side-scan sonar heat map of hydrilla occurrence in Shoulderbone Cove in October 2020. Green represents areas with little to no aquatic vegetation growth, yellows and oranges indicate areas with intermediate aquatic vegetation growth, and red indicates high aquatic vegetation growth.

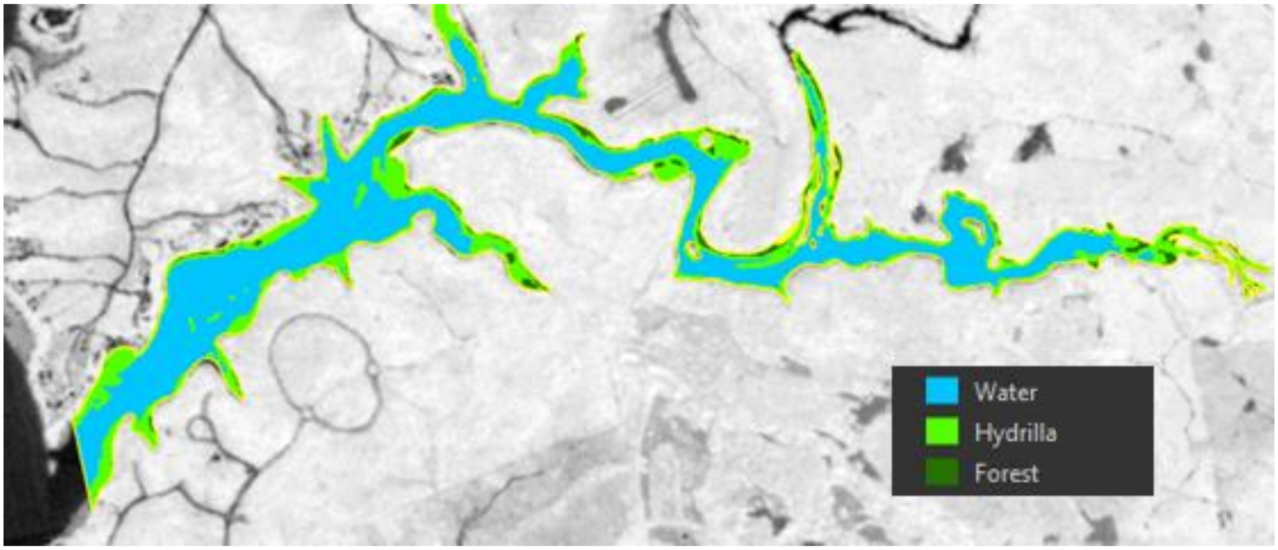

Figure 2-7: Satellite image classification for hydrilla occurrence in Shoulderbone Cove in October 2020.

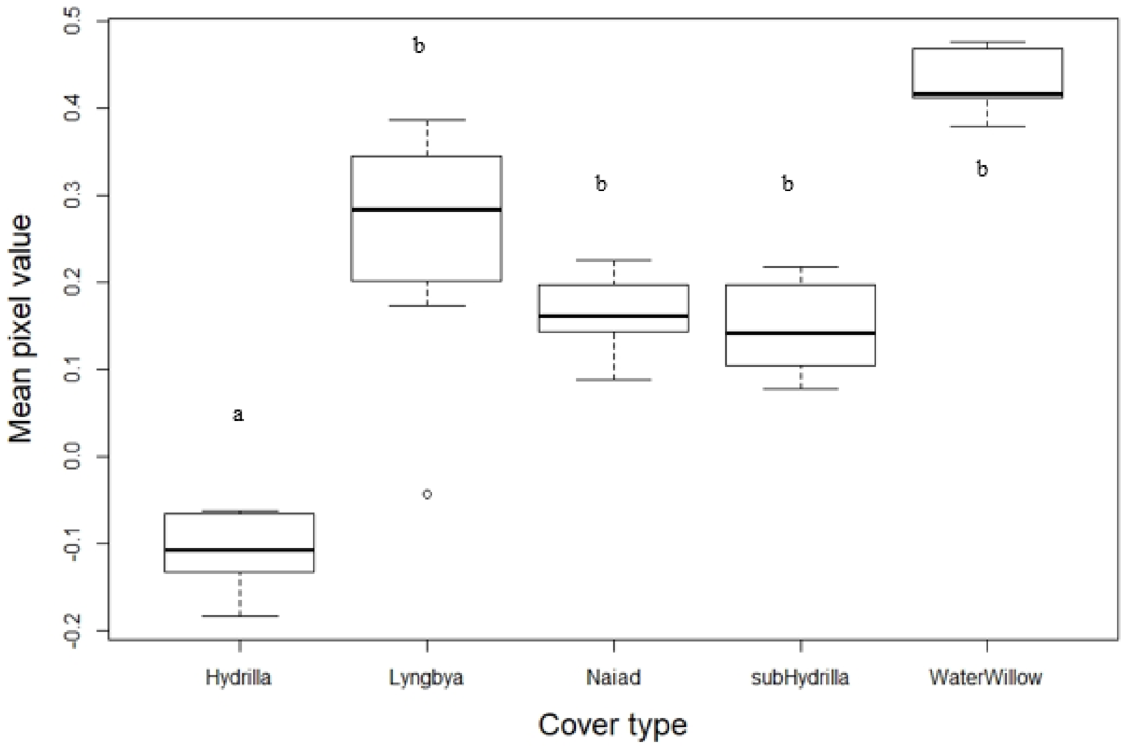

Figure 2-8: Box plots of mean coloration index pixel values associated with hydrilla, *Lyngbya*, southern naiad, submerged hydrilla, and water willow. Statistically significant differences are indicated with lowercase letters.

CHAPTER 3

EXAMINING THE INITIAL RESPONSE OF A SPORTFISH COMMUNITY TO HYDRILLA INVASION

Introduction

Since its initial introduction in Florida in 1960 (Blackburn et al. 1969), hydrilla (*Hydrilla verticillata*) has spread rapidly throughout the United States and now persists in freshwater lakes, ponds, rivers, and impoundments in 28 states as well as Guam and Puerto Rico (Jacono et al. 2020). Hydrilla grows vertically in the water column when it is rooted in substrate and forms dense mats when it reaches the surface, hindering anthropogenic use of waterways for navigation, commercial fishing, irrigation, and hydroelectric power generation (Cook and Lüond 1982).

Hydrilla is an aquatic macrophyte that provides many essential services to sportfish in lacustrine systems. The complex structure formed by their stems and leaves creates valuable spawning habitat (Chew 1974), shelter from predators (Savino and Stein 1982), and habitat for macroinvertebrates that young sportfish prey on (Moxley and Langford 1982, Carniatto et al. 2014). Hydrilla also provides habitat for smaller forage fish (e.g., bluegill (*Lepomis macrochirus*) and redear sunfish (*Lepomis microlophus*), which are a staple in the diets of subadult and adult largemouth bass (*Micropterus salmoides*) (Barnett and Schneider 1974). As a result, an increase in abundance of young-of-the-year and subadult largemouth bass has been shown in systems where hydrilla is present (Tate et al. 2003, Bonvechio and Bonvechio 2006, Nagid et al. 2014). Similarly, Barrientos and Allen (2008) reported that when compared to two species of native aquatic macrophytes at similar densities, hydrilla supported the highest fish biomass and was not associated with any significant variation in fish species composition. Other research suggests that hydrilla presence does not have an impact on diversity and abundance of native plants and fauna (Killgore et al. 1989, Hoyer et al. 2008).

While the impacts of aquatic macrophyte density on sportfish abundance are generally neutral or positive, the same cannot be said for sportfish growth. Maceina and Slipke (2004) and Olson et al. (1998) suggest that largemouth bass foraging efficiency is highest at intermediate aquatic macrophyte densities, and multiple studies have reported that when submersed aquatic vegetation coverage is too high it causes stunting and negatively impacts sportfish growth and body condition (Buck et al. 1975, Colle and Shireman 1980). For example, feeding efficiency and body condition of popular sportfish such as largemouth bass, bluegill sunfish, redear sunfish, and black crappie (*Pomoxis nigromaculatus*) tends to decrease as hydrilla density increases (Colle and Shireman 1980, Morrow et al. 1991). A possible explanation for this is that predation success decreases as the structural complexity of a habitat increases (Glass 1971, Stein and Magnuson 1976); more structure provides cover for prey species making it difficult for predatory species, such as largemouth bass, to access them.

Hydrilla abundance also likely affects the availability of different forage species that largemouth bass prey on, as the shift to piscivory occurs at smaller sizes in reservoirs where hydrilla removal efforts are present (Bettoli et al. 1992, Shelton et al. 1979). If hydrilla delays the onset of piscivory, then growth, body condition, and survival of Age-1 fish may decline (Olson 1996).

The variation in how hydrilla affects sportfish communities is likely caused by differences in climate and system characteristics and is largely responsible for the lack of scientific consensus on the topic. For example, Durocher et al, (1984) suggested that reductions of a Texas reservoir's aquatic plant coverage below 20% will reduce largemouth bass abundance. Hoyer and Canfield (1996) performed an empirical analysis of largemouth bass and aquatic vegetation in Florida lakes. A weak positive correlation between aquatic macrophyte abundance and the abundance and growth of young-of-the-year and subadult bass was found, but there was no relationship between aquatic macrophyte abundance and adult largemouth bass abundance or growth. Another study performed in Florida indicates that hydrilla at high densities in small systems may lead to hypoxic conditions and reduced habitat quality (Bradshaw et al. 2015).

Hydrilla was recently established in Lake Sinclair, Georgia, a deeply channelized reservoir (maximum depth 27m) maintaining high flow rates throughout the water column due to pump-back for hydroelectric power generation. As a result, complete hydrilla coverage will likely occur only in shallow bays, unlike smaller or shallower systems where total hydrilla coverage occurs throughout. Evaluating the recent introduction of hydrilla into this novel system provides a unique opportunity to further our understanding of hydrilla and sportfish interactions. Therefore, the objective of this chapter is to examine changes in the community metrics of several sportfish species (i.e., largemouth bass, black crappie, redear sunfish, and bluegill) in Lake Sinclair, GA before and after hydrilla became established, as well as with a neighboring, hydrilla-free reservoir.

Methods

Study Site

Lake Sinclair and Lake Oconee are two reservoirs on the Oconee River in central Georgia, both of which are popular for recreational fishing and boating and are used for hydroelectric power generation. Lake Sinclair (Figure 3-1) was created by the Georgia Power Company in 1953 and is contained within Baldwin, Hancock, and Putnam counties. It has a surface area of 6,204 ha, a maximum depth of 27 m, and 671 km of irregular shoreline with many coves and tributaries. (Smith and Manoylov 2013). Lake Oconee (Figure 3-1) is also a Georgia Power managed reservoir constructed in 1979 and converges with Lake Sinclair at Wallace Dam (Bachoon et al. 2009). It is the second-largest reservoir in Georgia at an area of 7709 ha with shorelines primarily characterized by residential development and several golf courses (Sherchan and Bachoon 2011). Both reservoirs are oligotrophic systems characterized by sandy or rocky bottom and typically scarce aquatic vegetation (Smith and Manoylov 2013). These reservoirs are connected by a hydroelectric dam, and exchange approximately 8% of their water volume daily as a result of pumpback for power generation (O'Rouke, 2022). As a result, Lake Sinclair and Lake Oconee are effectively connected and share very similar hydrology and physical characteristics. Hydrilla first became established in Lake Sinclair in 2018 but has not yet spread to neighboring Lake Oconee.

Data Collection

Georgia Department of Natural Resources (GADNR) biologists performed annual fish sampling on Lake Sinclair and Lake Oconee each year from 2013-2020 using gill netting and electrofishing. Sampling of Lake Sinclair occurred in the fall, and Lake Oconee was sampled in the spring each year. Electrofishing surveys occurred at the same stationary sites each year on Lake Sinclair (n=10) and Lake Oconee (n=14) (Figure 3-2). Each station had a defined starting point and was sampled parallel to shore. Lake Sinclair electrofishing sites were sampled for 900 seconds each, while Lake Oconee sites were sampled for 1800 seconds. Individual length and weight were recorded for all target species collected.

Annual gill net sampling occurred at the same stationary sites on Lake Sinclair (n=10) and Lake Oconee (n=12) (Figure 3-2). Nets were 61.0-m monofilament experimental nets that were 1.8 m deep, consisting of five 12.2-m panels with mesh sizes of 1.9 cm, 2.5 cm, 3.8 cm, 5.1 cm, and 5.4 cm. Nets were set perpendicular to shore within one hour of sunset and fished overnight. Each site was randomly selected to have the small or large mesh closest to shore. Targeted species were measured for total length (TL) in millimeters (mm) and weighed to the nearest gram (g). We removed otoliths from all largemouth bass captured in fall 2020 and spring 2021 for Lake Sinclair and Lake Oconee, respectively.

Otolith processing followed the grinding method described in Sakaris and Bonvechio (2020), in which the otoliths were browned on a hot plate, cracked near the nucleus, mounted on a microscope slide with Crystalbond, ground flat on 1000 grit sandpaper, then polished using lapping film and Type B immersion oil for viewing. Each otolith was independently aged by three viewers, and measurements from the nucleus to each annulus and the edge of the structure were obtained using ImagePro Plus software.

Data Analysis

All data analyses were performed using RStudio statistical software (R Core Team 2019) in combination with the FSA: Fisheries Stock Analysis package (Ogle et al. 2021). The long-term dataset from GADNR supported analyses of body condition (relative weight; Wr) (Wege and Anderson 1978), size structure (proportional size distribution; PSD) (Neumann and Allen 2007), and relative abundance (catch-per-unit-effort; CPUE). We used the Dahl-Lea method for back-calculations (Lea 1910) to compare estimates of the mean length of largemouth bass at age-1 before and after the introduction of hydrilla to Lake Sinclair. Catch per unit effort was calculated for electrofishing only, while body condition, size structure, and back-calculated length analyses utilized both electrofishing and gill net data.

We used a Before-After-Control-Impact experimental design, where “Before” was defined as 2013-2017, “After” was 2018-2020, and the control was Lake Oconee. Outliers generated in relative weight calculations were identified using the interquartile range (IQR) criterion and removed using the boxplot() and which() functions. Analysis of variance (ANOVA) was performed using the aov() function to test for significant changes in electrofishing CPUE, Wr, and length at age-1 (largemouth bass only) among years and between reservoirs. If the ANOVA indicated a significant result, Tukey’s post hoc comparisons were performed using the TukeyHSD() function in base R to determine whether the change was associated with the years after hydrilla’s introduction and if it was occurring only in Lake Sinclair. We assessed each response variable for normality using a Shapiro-Wilk’s test ($\alpha = 0.05$) (Shapiro and Wilk 1965), and if necessary, performed several data transformation techniques to satisfy assumptions of parametric statistics. If normality could not be achieved through transformations, the Kruskal-Wallis nonparametric test (Kruskal and Wallis, 1952) was used.

Proportional Size Distribution (PSD) analyses were conducted using Chi-Squared tests with Pearson residuals, as both the predictors and response variable are categorical (Neumann and Allen 2007). The years examined were 2016, 2018, and 2020. These years were selected because they provide defined “before”, during”, and “after” time periods. Pearson residuals represent how much an observed value deviates from the expected value generated by a Chi-Square test, with any value approaching +2 or -2 being considered significant.

Results

Relative weights (Wr) of black crappie, bluegill, and redear sunfish were highly cyclical in both reservoirs through time, and any significant difference after 2018 cannot be definitively linked to hydrilla (Figures 3-3, 3-4, 3-5). Largemouth Bass in Lake Sinclair exhibited a significant increase in relative weight in 2018 and 2019 when compared to 2014, 2015, and 2017 but returned to pre-2018 levels in 2020 (Figure 3-6). No significant change occurred in Lake Oconee across all years. (Figure 3-7).

Proportional Size Distribution analysis of bluegill in 2018, the year of hydrilla's introduction, indicated the number of "quality" size and larger (Q+) bluegill captured in Lake Sinclair was significantly higher than in 2016. Two years later, the number of Q+ bluegill captured was significantly lower than in 2016 (X^2 = 27.96; df = 2; p < 0.001) (Figure 3-8**)**. However, this same trend was reflected in the control reservoir, Lake Oconee (X^2 = 21.461; df = 2; p <0.001) (Figure 3-9**)**, suggesting that this decline cannot be attributed to the presence of hydrilla.

The size distribution of redear sunfish in Lake Sinclair shifted toward a smaller size from 2018 to 2020. In 2018, the observed catch of "sub-stock to quality" (SS-Q) size individuals was significantly lower than in 2016 while the number of Q+ fish captured was significantly higher. In 2020, the opposite occurred, with numbers of SS-Q redear sunfish being higher and Q+ being lower (X^2 = 21.465; df = 2; p < 0.001) (Figure 3-10). This trend was not reflected in Lake Oconee (Figure 3-11). The Kruskal-Wallis nonparametric test was only employed for black crappie catch per unit effort data, and CPUE values did not change significantly following hydrilla's introduction for any of the four species examined (Table 3-1). However, mean annual CPUE has continually increased for redear sunfish in Lake Sinclair since 2018 (Figure 3-12).

Growth of juvenile largemouth bass did not change following hydrilla's introduction. The mean length of age-1 largemouth bass was similar after hydrilla was introduced (F =1.527, df = 1, p=0.22) (Figure 3-13). This same trend was observed in Lake Oconee (Figure 3-14).

Discussion

We detected few changes in sportfish relative abundance, size structure, and growth following the introduction of hydrilla in Lake Sinclair. These results add to the growing body of literature that indicate varying responses of the fish community to the introduction of non-native hydrilla.

The lack of change in mean length of age-1 largemouth bass in Lake Sinclair is inconsistent with the findings of several previous studies involving invasive aquatic vegetation such as hydrilla. A decline in size of age-1 fish has been frequently observed due to a delayed shift to piscivory as hydrilla spreads throughout shallow areas where young fish frequent (Shelton et al. 1979, Bettoli et al. 1992, Valley and Bremigan 2002). Several factors may have influenced our results. The introduction of hydrilla was a recent occurrence (2018); therefore, a lag time may be occurring before ecological effects are realized. Additionally, many studies that have identified negative impacts on the sportfish community occurred in systems with much higher hydrilla coverage than Lake Sinclair. It is likely that Lake Sinclair has not yet reached a level of infestation where a threshold response in the fish community would occur. Furthermore, fish sampling site selection were based on fixed sites in areas that are conducive for increased sample gear efficiency. Although these data are likely sufficient to make inferences for long-term trends, they may not be adequate for documenting fine-scale (i.e., temporal or spatial) changes. Continued monitoring of both areas with and without hydrilla colonization will be necessary to observe long-term trends as hydrilla continues to spread throughout the shallow coves of Lake Sinclair.

The significant proportional increase in SS-Q size redear sunfish is notable considering that this change was not reflected in Lake Oconee. Redear sunfish utilize aquatic vegetation beds for spawning and recruitment (Barnett and Schneider, 1974), and the relative increase in vegetation abundance in Lake Sinclair since hydrilla's introduction may be increasing young-of-year survival – and proportion of smaller individuals - by reducing the foraging efficiency of piscivorous predators (e.g., largemouth bass). Bluegill also utilize aquatic vegetation for spawning and recruitment but are capable of spawning 2-3 times per year in the Southeastern U.S. (Partridge and DeVries 1999), while redear sunfish only spawn once per year (Sammons et al. 2011). The frequency that bluegill spawn may mask effects that hydrilla had on the population's size structure, while the historically low abundance of sub-Quality size redear sunfish may be inflating the actual magnitude of the apparent increase in captures.

Lake-specific characteristics in Lake Sinclair are likely preventing noticeable impacts on the sportfish community. For example, when hydrilla coverage exceeds 50% of a reservoir, declines in black crappie growth rates (Maceina and Shireman 1985) and bluegill body condition (Colle and Shireman 1980) have been observed due to reduced feeding efficiency. The bathymetry and hydrology of Lake Sinclair makes this level of infestation unlikely, as hydrilla growth is limited by availability of light. Most of the channelized portions of Lake Sinclair are resistant to invasion by hydrilla because of depth. Additionally, the pump-back cycle used for hydroelectric power generation leads to high turbidity throughout the water column and low benthic light levels in many areas of Lake Sinclair. As a result, hydrilla coverage of Lake Sinclair is still well below 10%. These factors should limit widespread hydrilla coverage throughout the reservoir and may continue to prevent negative biological impacts from occurring in sportfish in subsequent years.

Understanding how sportfish communities in a reservoir respond to hydrilla requires long-term monitoring efforts, even if hydrilla is not yet established in the system. For example, hydrilla is not present in Lake Oconee but given the nature of the pump-back system and high probability of recreational anglers moving between reservoirs because of their proximity, it is possible that hydrilla could become established in the future. In this event, continued contributions to the long-term dataset established by the Georgia DNR will facilitate the design and implementation of before-after studies to document the ecological and economic impacts of hydrilla's spread within the system if it occurs.

Understanding how various stakeholders perceive hydrilla is an important component for managing aquatic vegetation and sportfishing alike. Anglers targeting largemouth bass were found to prefer fishing in areas with normal to high abundance of aquatic vegetation, including hydrilla (Slipke et al. 1998), and several popular bass fishing magazines feature articles about fishing in and around hydrilla. This may be due to the anecdotal idea that hydrilla is good for bass fishing, without understanding how hydrilla can negatively impact the ecosystem and other user groups. Education on hydrilla's negative effects can help to dispel this misconception. Additionally, informing the public on how to properly clean vessels and trailers to prevent spreading hydrilla to other systems is an essential step, as stopping hydrilla from becoming established in a reservoir is the best way to manage it.

Continuing to expand our understanding of how hydrilla affects sportfish communities could lead to more successful management of aquatic ecosystems. This study contributes to the growing base of literature on hydrilla's effects by examining the initial response of a sportfish community following hydrilla introduction into a novel system. Despite colonization in shallow bays throughout the reservoir, the fish community was largely unchanged. Assessing long-term effects will require continued monitoring coupled with hydrilla assessments to elucidate what levels of hydrilla coverage and biomass trigger a threshold response in the sportfish community.

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Table 3-1: Analysis of variance results for comparisons of fish CPUE for each year for all four study species. There were no significant differences among years.

Species	*F*	*Df*	*p*
Largemouth Bass	0.704	7	0.669
Black Crappie	1.039	7	0.411
Bluegill	0.782	7	0.605
Redear Sunfish	0.671	7	0.696

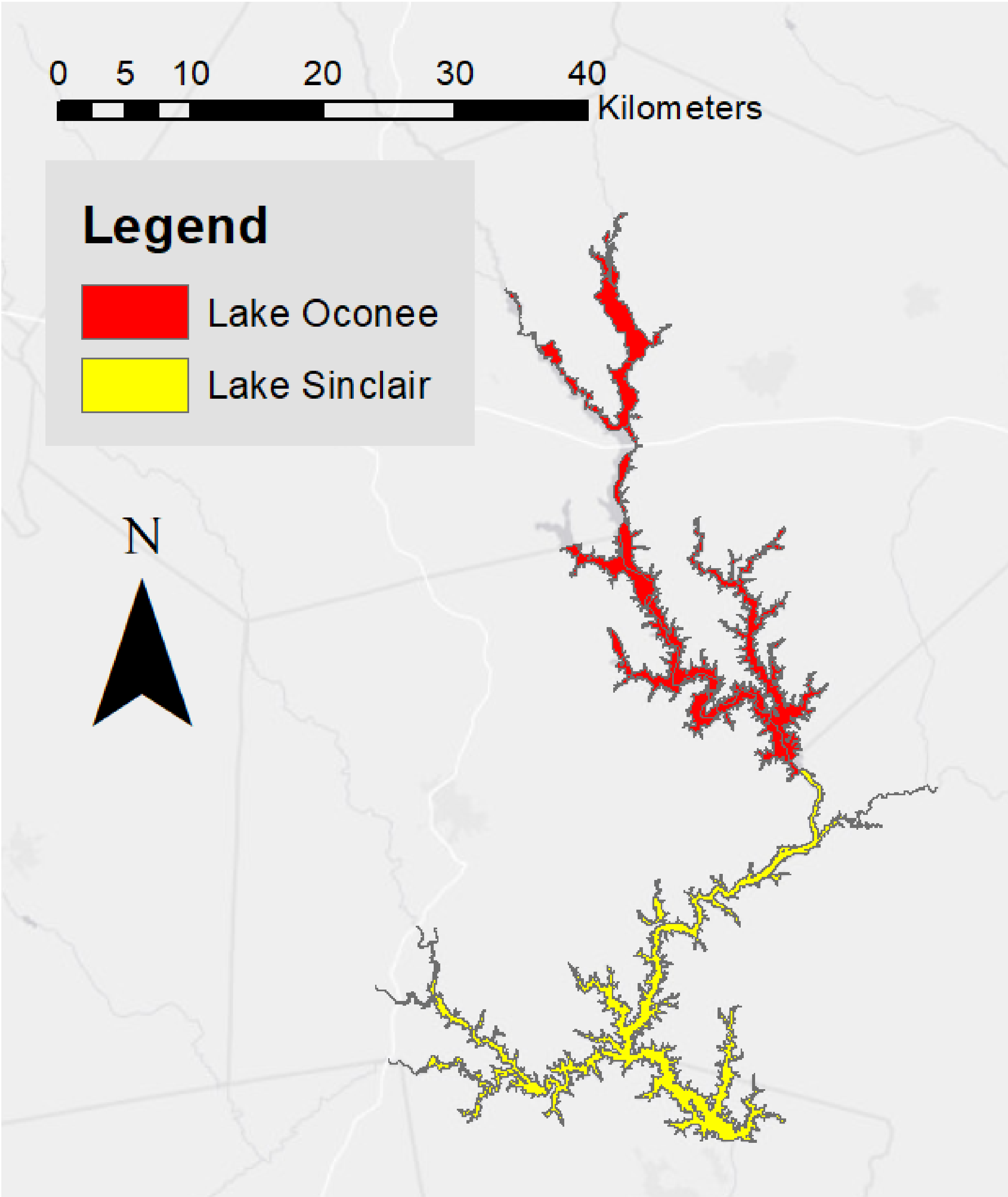

Figure 3-1: Map of Lake Oconee (Red) and Lake Sinclair (Yellow) in central Georgia, separated by Wallace Dam.

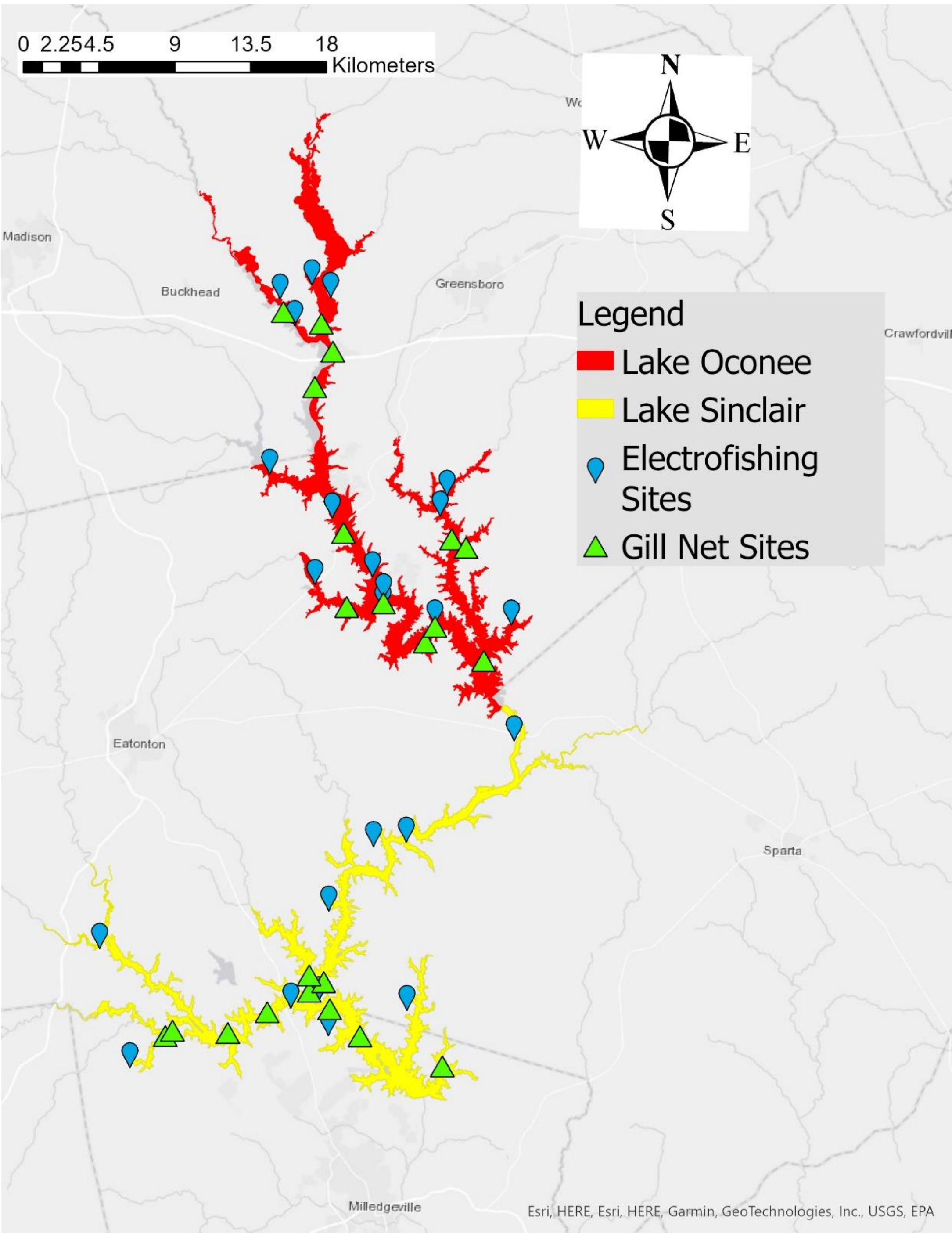

Figure 3-2: Standardized electrofishing and gill net sampling sites from the Georgia Department of Natural Resources on Lake Oconee and Lake Sinclair, Georgia.

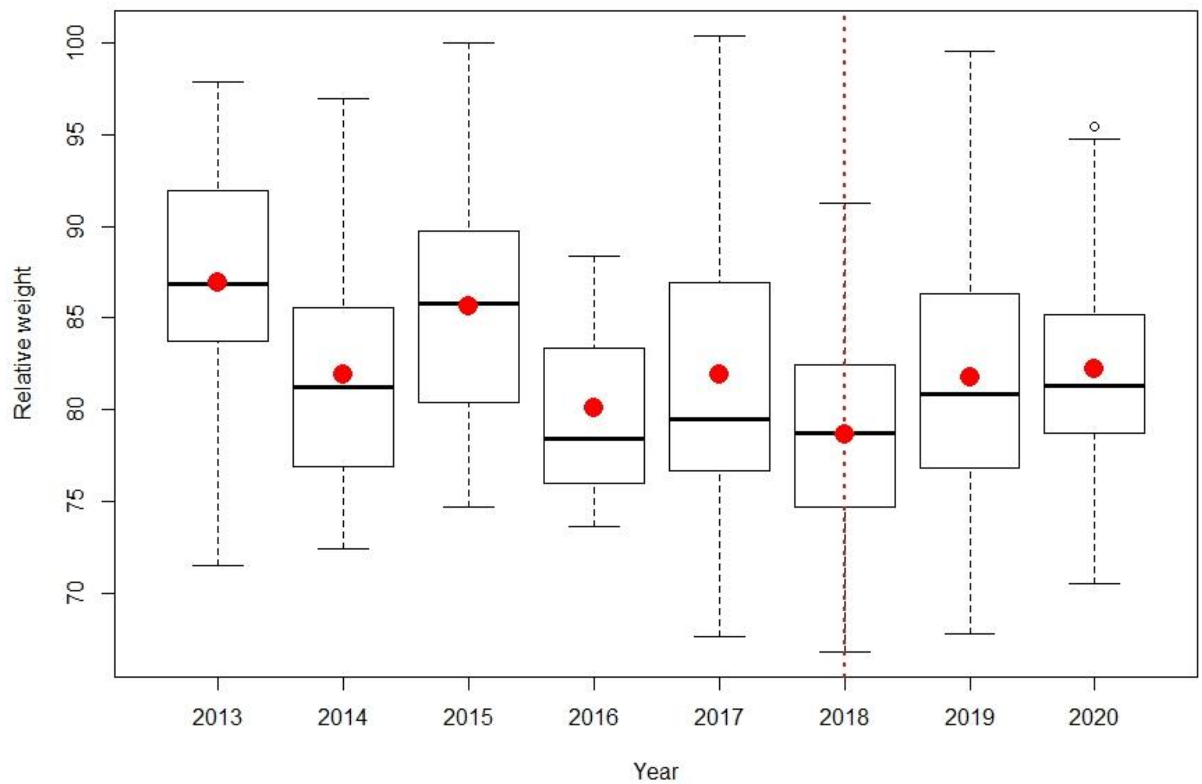

Figure 3-3: Box plot (mean [circles], median, interquartile range, and outliers) of relative weight (Wr) of Lake Sinclair black crappie from 2013 to 2020. The year hydrilla was introduced is represented by the dashed red line.

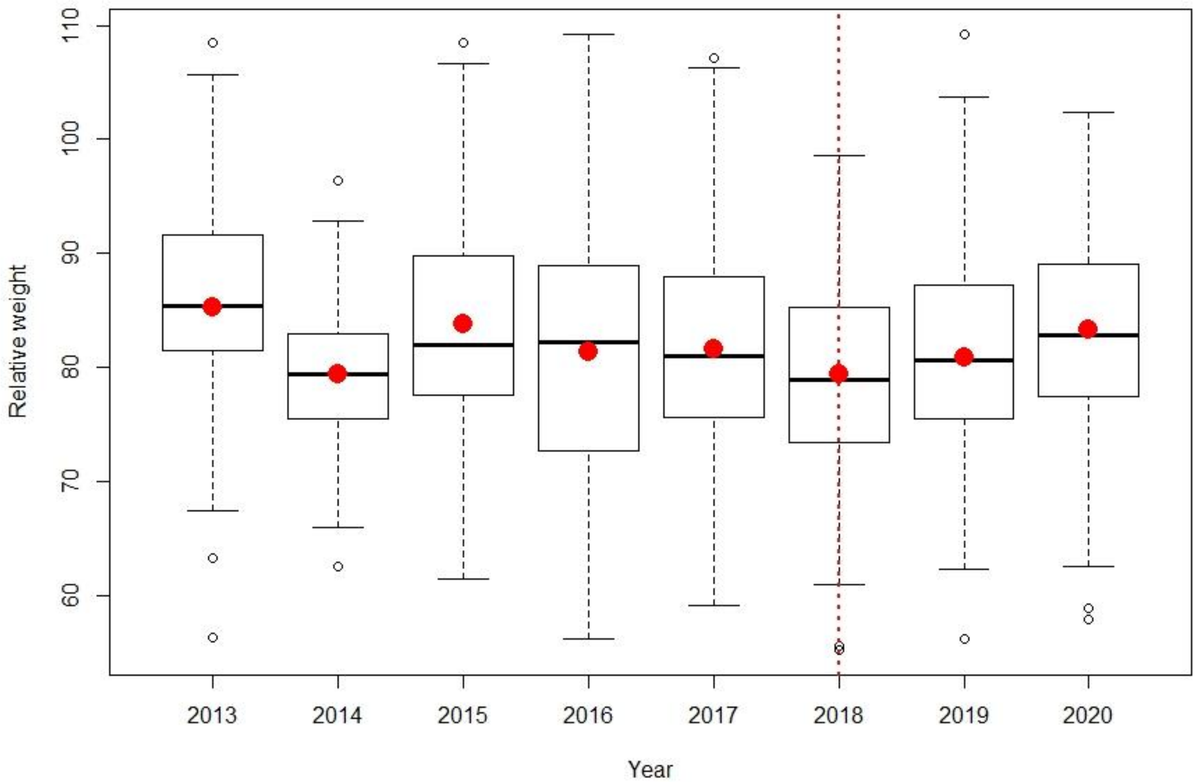

Figure 3-4: Box plot (mean [circles], median, interquartile range, and outliers) of relative weight (Wr) of Lake Sinclair bluegill from 2013 to 2020. The year hydrilla was introduced is represented by the dashed red line.

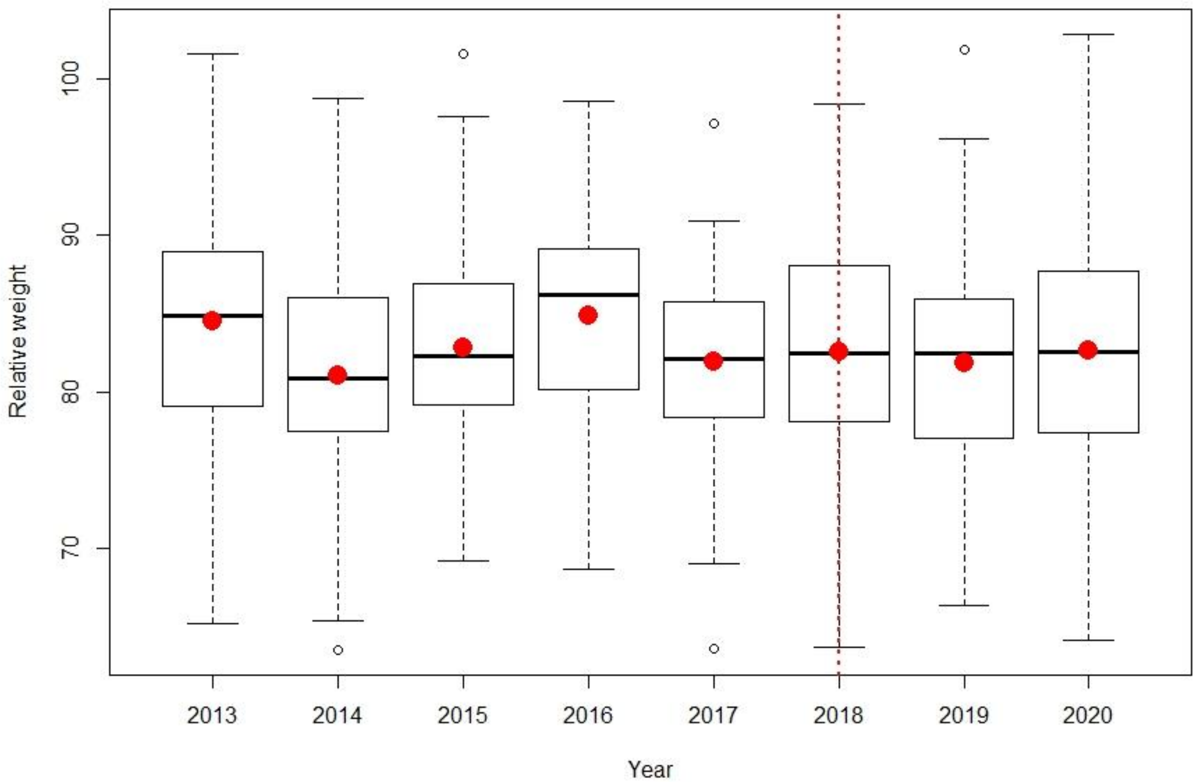

Figure 3-5: Box plot (mean [circles], median, interquartile range, and outliers) of relative weight (Wr) of Lake Sinclair redear sunfish from 2013 to 2020. The year hydrilla was introduced is represented by the dashed red line.

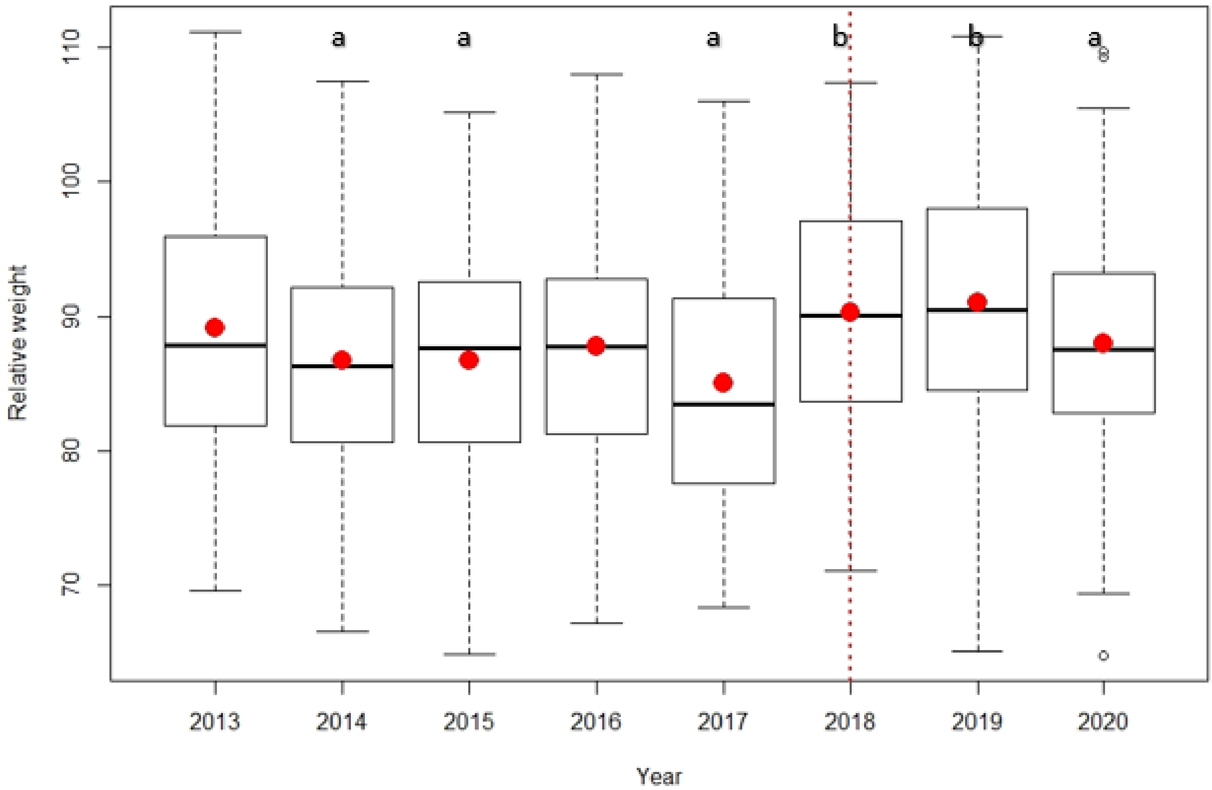

Figure 3-6: Box plot (mean [circles], median, interquartile range, and outliers) of relative weight (Wr) of Lake Sinclair largemouth bass from 2013 to 2020. Statistically significant differences are indicated with lowercase letters, and the year hydrilla was introduced is represented by the dashed red line.

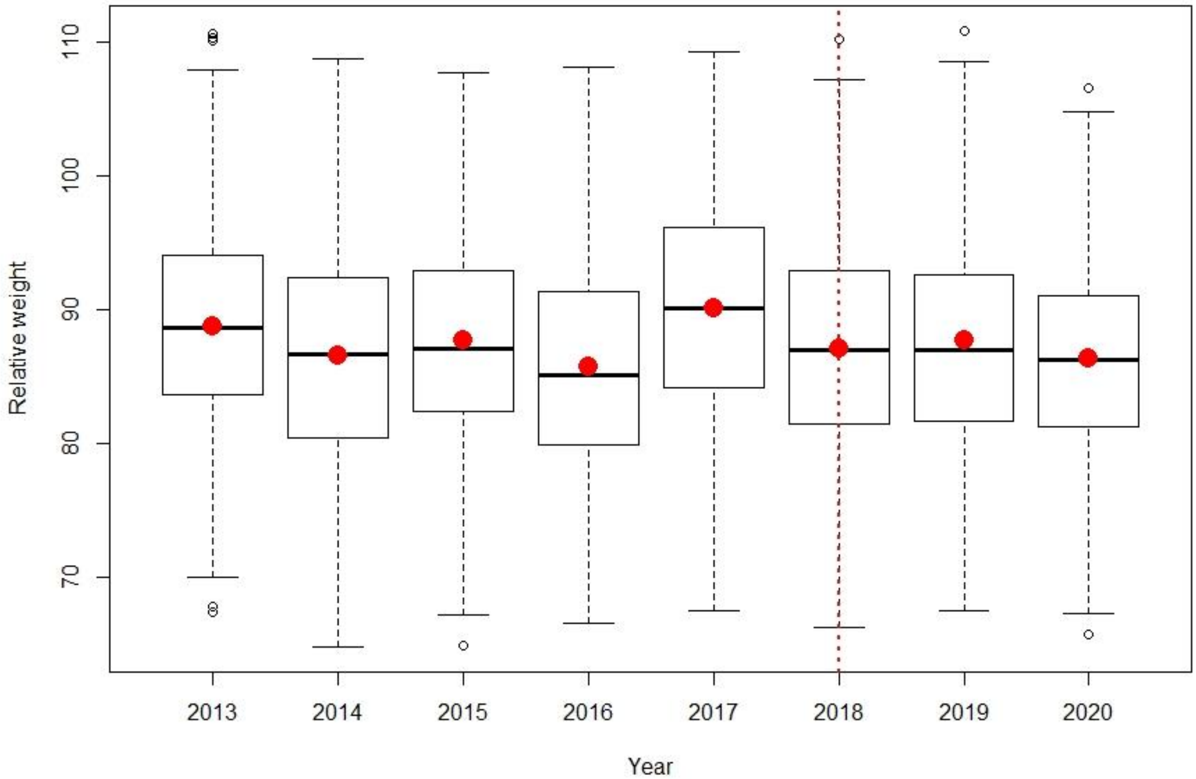

Figure 3-7: Box plot (mean [circles], median, interquartile range, and outliers) of relative weight (Wr) of Lake Oconee largemouth bass from 2013 to 2020. The year hydrilla was introduced is represented by the dashed red line.

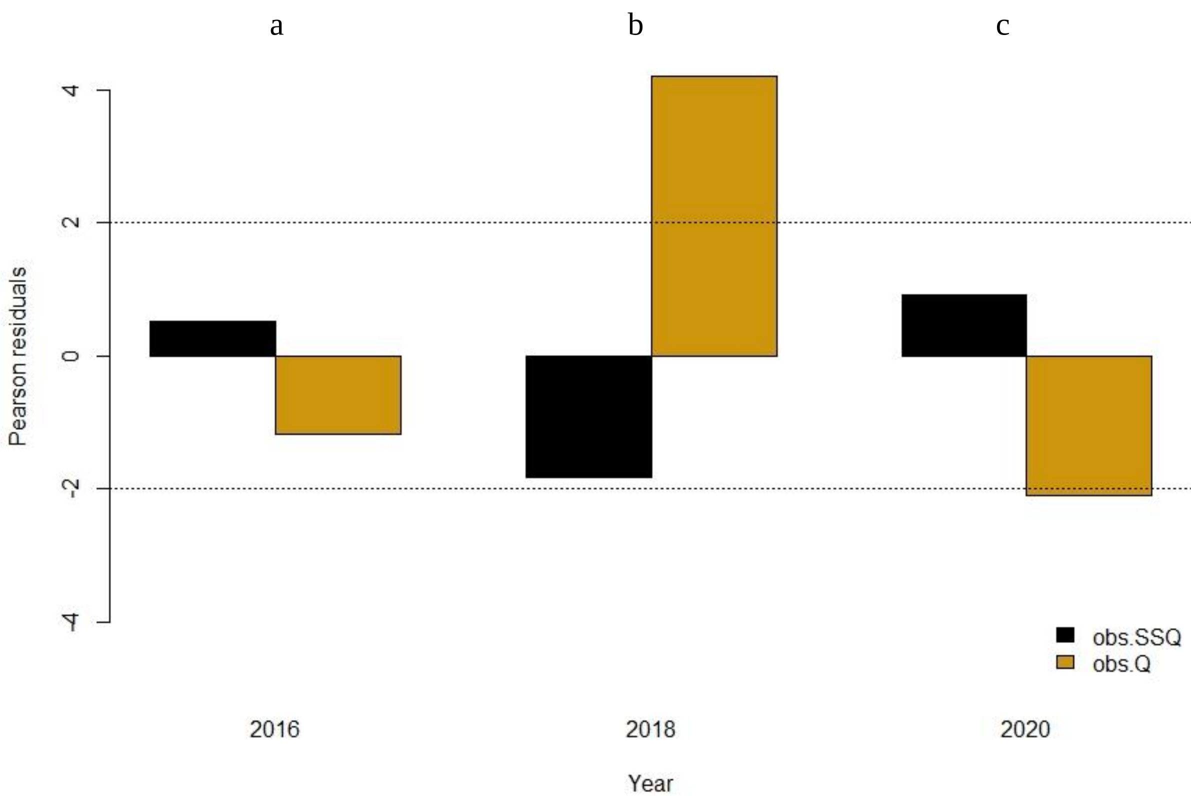

Figure 3-8: Pearson residuals for observed catch of sub-stock to quality (SSQ) and Quality+ (Q) bluegill in Lake Sinclair in 2016, 2018, and 2020. Pearson residuals indicate how much observed values deviate from the expected values generated by a Chi-Squared test, with values approaching +2 or -2 being considered significant. Statistically significant differences are indicated with lowercase letters.

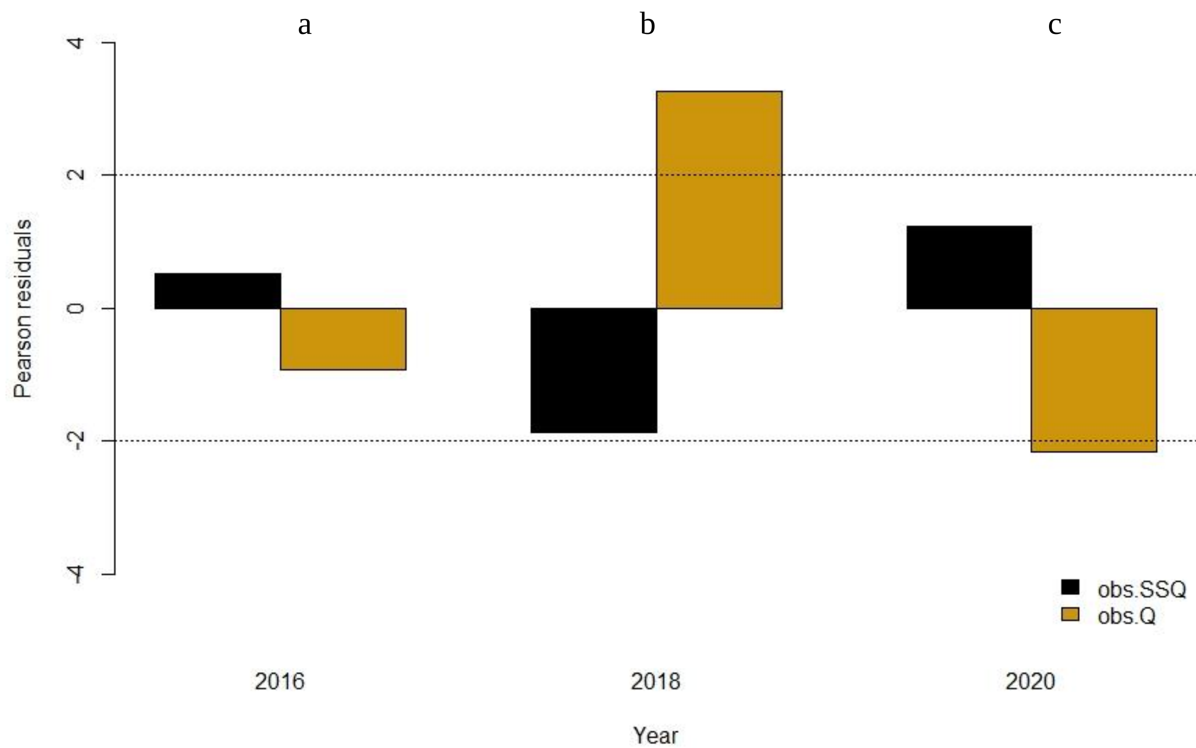

Figure 3-9: Pearson residuals for observed catch of sub-stock to quality (SSQ) and Quality+ (Q) bluegill in Lake Oconee in 2016, 2018, and 2020. Pearson residuals indicate how much observed values deviate from the expected values generated by a Chi-Squared test, with values approaching +2 or -2 being considered significant. Statistically significant differences are indicated with lowercase letters.

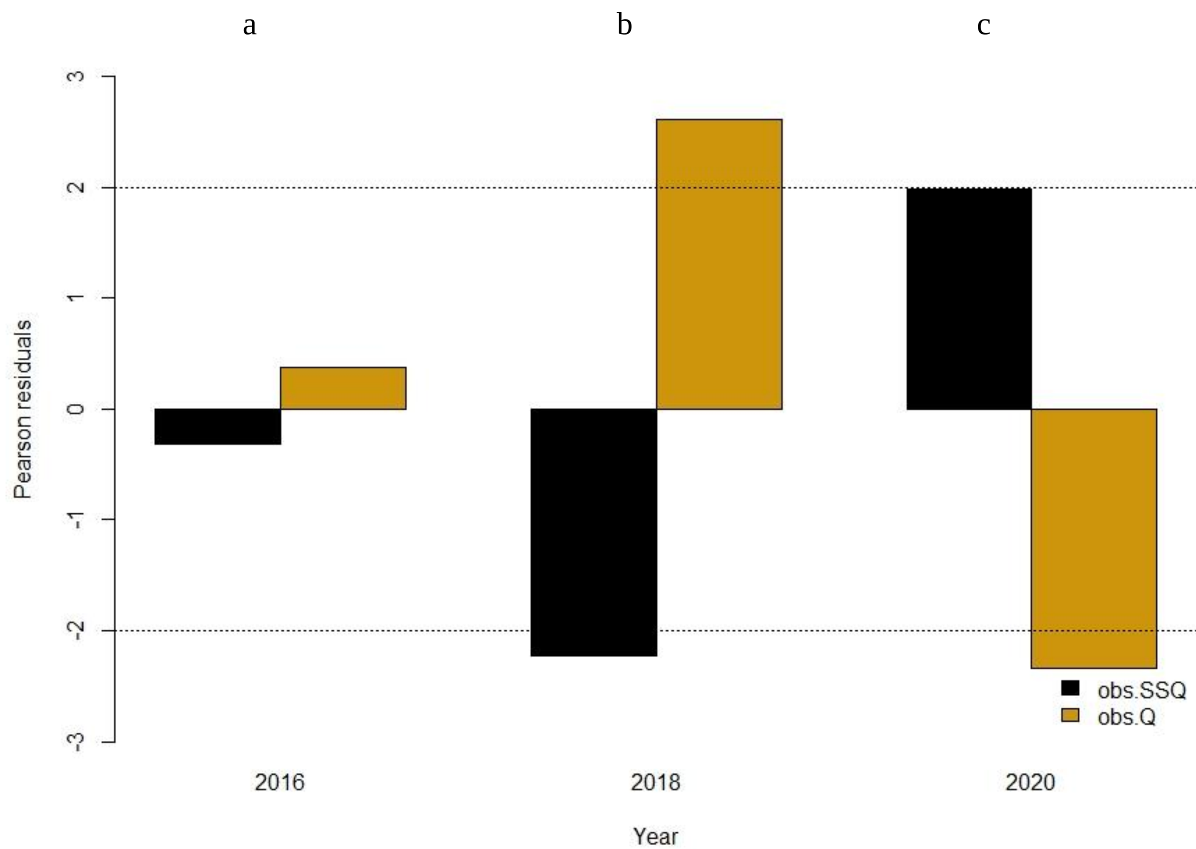

Figure 3-10: Pearson residuals for observed catch of sub-stock to quality (SSQ) and Quality+ (Q) redear sunfish in Lake Sinclair in 2016, 2018, and 2020. Pearson residuals indicate how much observed values deviate from the expected values generated by a Chi-Squared test, with values approaching +2 or -2 being considered significant. Statistically significant differences are indicated with lowercase letters.

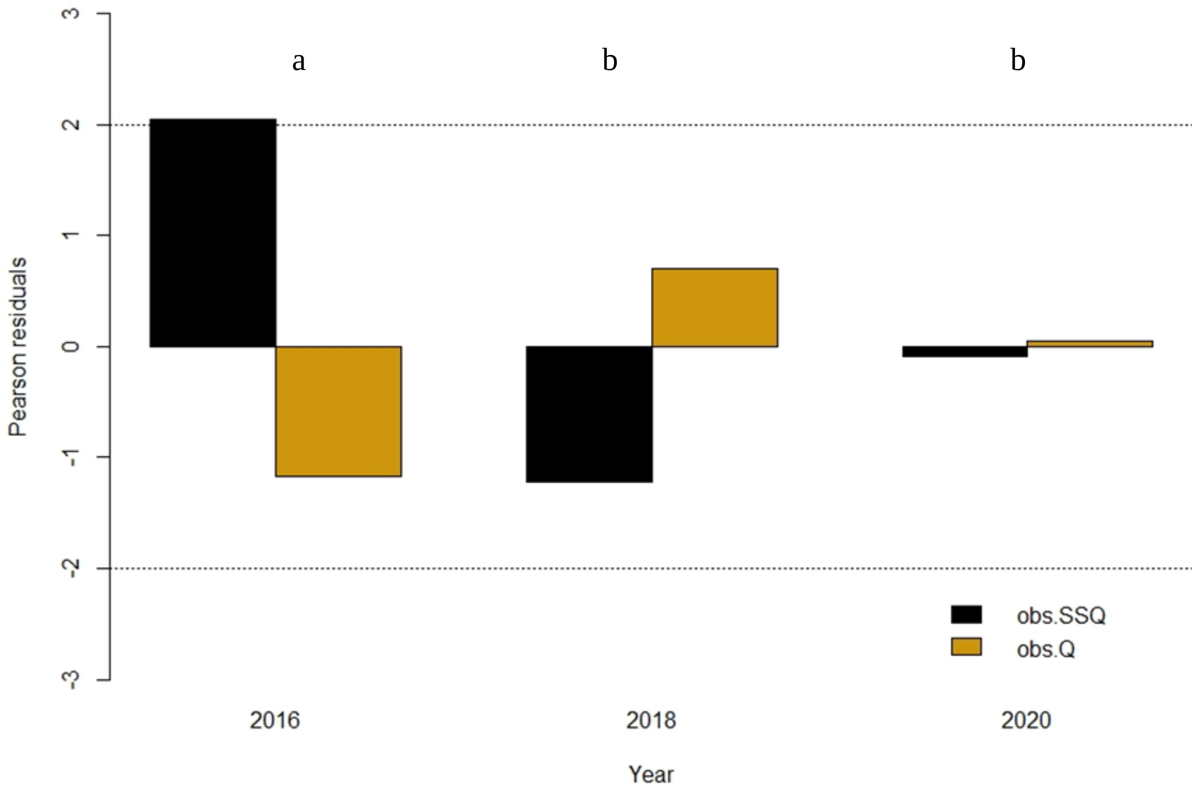

Figure 3-11: Pearson residuals for observed catch of sub-stock to quality (SSQ) and Quality+ (Q) redear sunfish in Lake Oconee in 2016, 2018, and 2020. Pearson residuals indicate how much observed values deviate from the expected values generated by a Chi-Squared test, with values approaching +2 or -2 being considered significant. Statistically significant differences are indicated with lowercase letters.

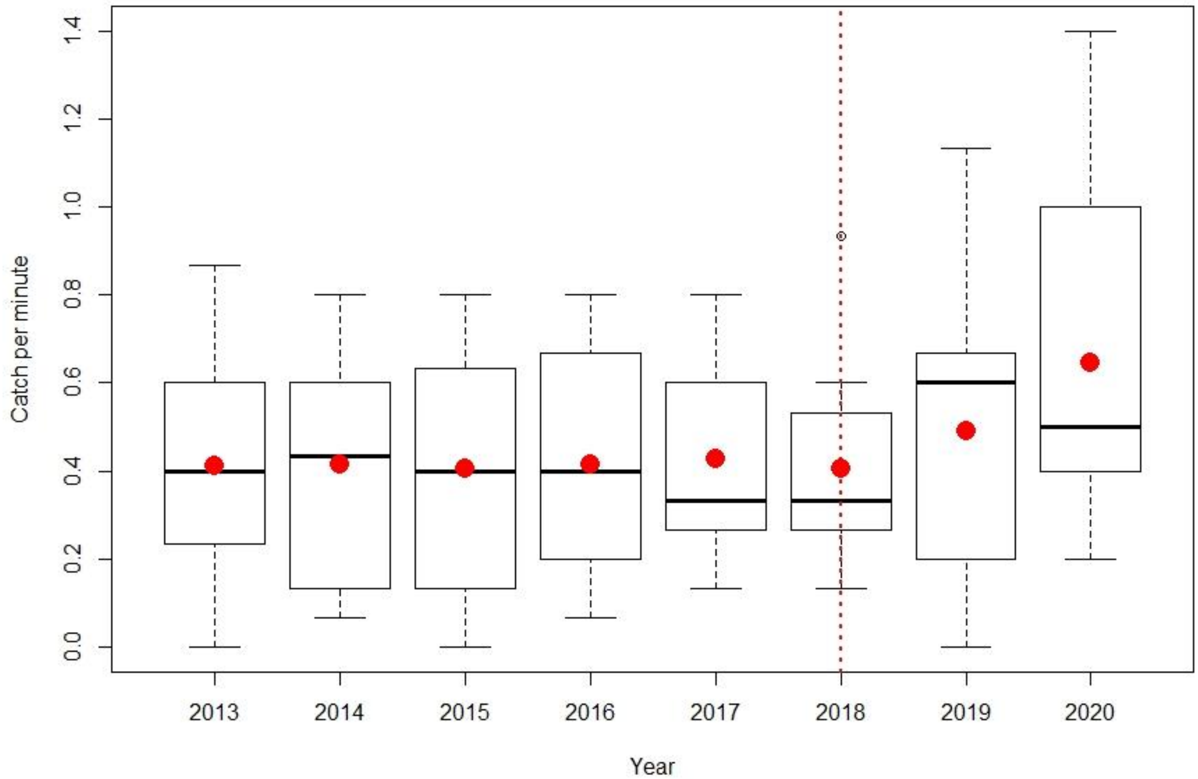

Figure 3-12: Box plot (mean [circles], median, interquartile range, and outliers) of redear sunfish catch per unit effort (CPUE) in Lake Sinclair from 2013 to 2020. The year hydrilla was introduced is represented by the dashed red line.

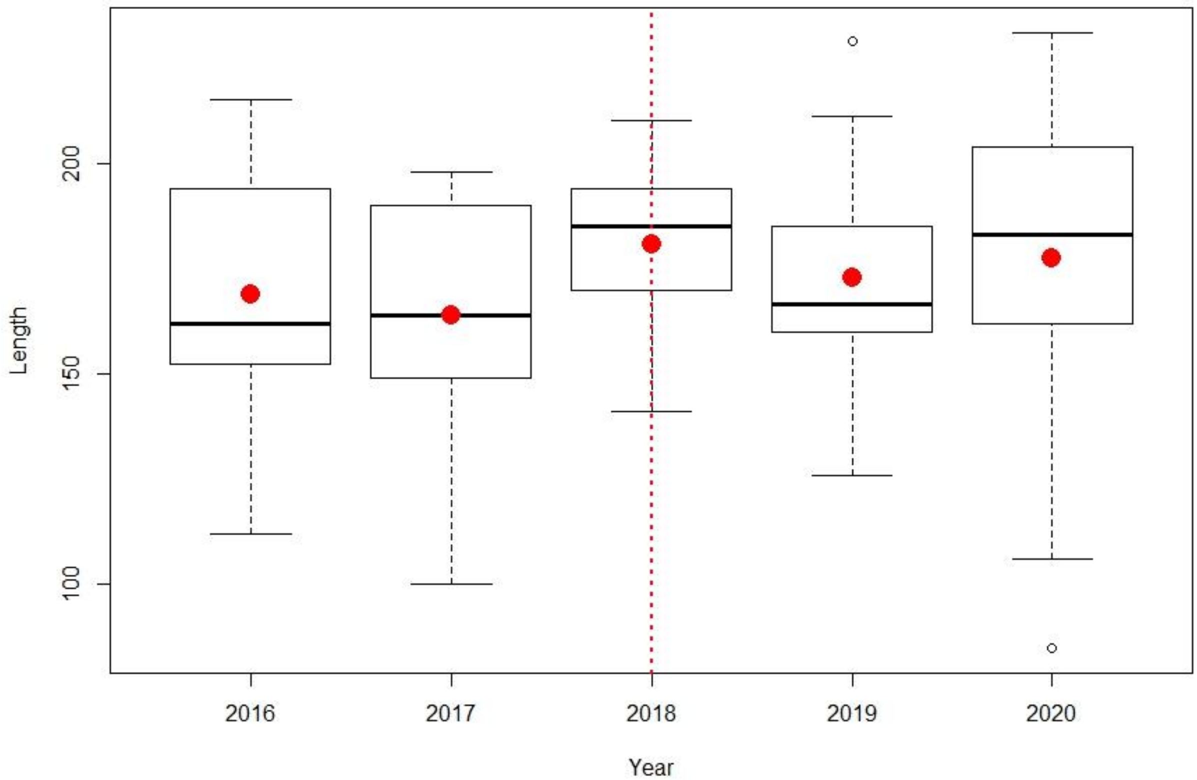

Figure 3-13: Box plot (mean [circles], median, interquartile range, and outliers) back-calculated length of age-1 largemouth bass in Lake Sinclair from 2016 to 2020. The year hydrilla was introduced is represented by the dashed red line.

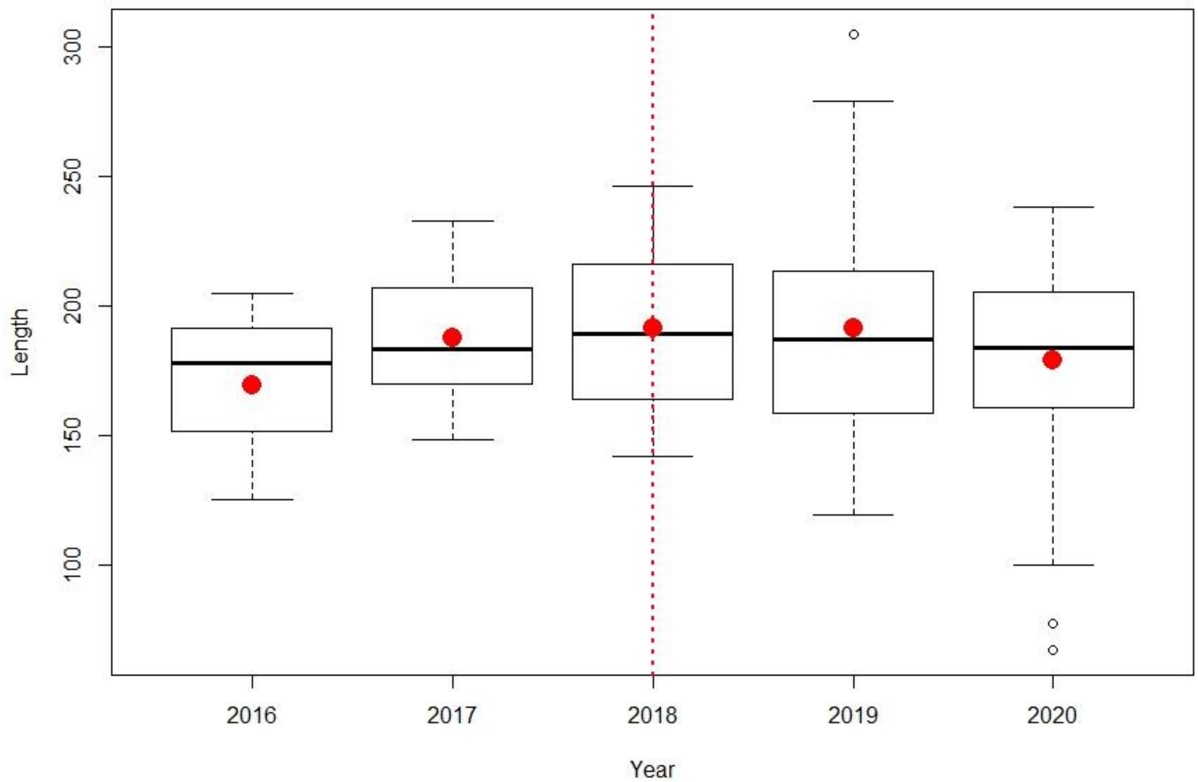

Figure 3-14: Box plot (mean [circles], median, interquartile range, and outliers) back-calculated length (mm) of age-1 largemouth bass in Lake Oconee from 2016 to 2020. The year hydrilla was introduced is represented by the dashed red line.

CHAPTER 4

SUMMARY, MANAGEMENT IMPLICATIONS, AND FUTURE DIRECTIONS

Summary

Hydrilla has spread rapidly through the shallow coves and tributaries of Lake Sinclair since its introduction in 2018, forcing Georgia Power to pursue management strategies and consider different assessment methods. This study sought to test the efficacy of high-resolution satellite and UAV imagery for identifying hydrilla and monitoring its spread, as well as to quantify impacts that hydrilla may be having on the sportfish community in Lake Sinclair.

Pixel-based image classifications generated for hydrilla detection using PLANET satellite imagery were found to be 91% accurate, and UAV imagery was extremely valuable for validating these findings. However, this level of accuracy was only possible late in the growing season when hydrilla was emergent. UAV imagery was unable to differentiate between species of submerged aquatic vegetation but was able to distinguish emergent hydrilla from all submerged species, as well as emergent water willow.

The sportfish community in Lake Sinclair exhibited little change since hydrilla's introduction in 2018. The redear sunfish population saw a shift toward smaller sizes, possibly due to the refuge from predation provided by hydrilla. No significant change was observed in relative weight, catch per unit effort, or proportional size distribution for any other species. In addition, mean length of age-1 largemouth bass did not change following hydrilla's introduction.

Management Implications

Chapter 2: Employing Remote Sensing Techniques for Monitoring Hydrilla Occurrence

Satellite imagery classification and analysis provides a quick and effective way to identify areas of prominent aquatic vegetation. Satellite imagery can effectively act as an early-detection method to identify new areas of vegetation occurrence, allowing managing officials to go directly to the area of concern rather than survey the entire reservoir by boat. UAV imagery is valuable for confirming the results of a satellite image classification, but both methods are only effective when vegetation is emergent. UAV imagery does show promise for differentiating between submerged vegetation species, and with further method development, could conceivably be used in lieu of rake-toss transect surveys – or other physical sampling techniques – in the future. Side-scan sonar assessments may still be required to obtain biovolume estimates for herbicide dosage calculations, but satellite image classification in combination with a bathymetric map could be used to obtain an approximate biovolume estimate.

The costs of equipment and time required for effective implementation of each of these methods is a necessary consideration for applied management. In my study, it took approximately 60 minutes to obtain satellite imagery of Shoulderbone Cove, upload it into GIS, and generate a pixel-based image classification. Comparatively, it took approximately 90 minutes to survey the same cove with a UAV and 240 minutes with side-scan sonar. The processing time for these data is variable depending on the capabilities of the computer. Commercial subscription prices from Planet Labs were not available, but as of 2018, the imagery cost $0.012 per hectare with a minimum area purchase price of $218.00 (Sozzi et al. 2018). When comparing the costs of UAV and side-scan sonar, it is important to consider the hours of labor required for each as well as recurring subscriptions. Startup costs for side-scan sonar are approximately 20% less than UAV, but recurring subscription costs will make side-scan sonar significantly more expensive over time (Table 4-1). All methods used require GIS software, with an ArcGIS Pro license costing approximately $1,500 annually.

Chapter 3: Examining the Initial Response of a Sportfish Community to Hydrilla Invasion

The sportfish community in Lake Sinclair was largely unchanged since hydrilla was first observed in the reservoir. However, given the recency of the introduction, there could be a lag before effects are observed at the fish community level. Furthermore, Lake Sinclair may be resilient to disturbance at the current level of hydrilla coverage, but additional colonization throughout the reservoir could exceed the ecological threshold and move Lake Sinclair into a new alternative state (Scheffer et al. 2001). Continued long-term monitoring is essential to identify changes in sportfish community metrics.

Future Directions

Chapter 2: Employing Remote Sensing Techniques for Monitoring Hydrilla Occurrence

Developing additional methodologies is necessary for UAV technology to be effectively used for submerged aquatic vegetation assessments. Different lens filters may help to cut glare, and sensors with greater spectral capabilities would allow for more robust vegetation indices (e.g., NDVI) to be used in the analysis.

The satellite image classification technique can be problematic when the area of interest cannot be encompassed by a single image. Lake Sinclair covers an area too large to be downloaded as a single surface reflectance image. To capture the full extent of the reservoir, it must be split up into five georeferenced images to preserve resolution. Even when images are taken on the same day, the position of the satellite, the sun, and atmospheric conditions will have changed. These minor changes will result in highly variable NDVI pixel values associated with each cover type and confuses the image classification. This could be overcome by performing ground truthing within the extent of each image and classifying them all separately, but this would be just as labor intensive as traditional methods. Instead, an atmospheric correction called dark-object subtraction could be performed. This technique subtracts the darkest pixel in an image from all other pixels, maintaining the distinction between each cover type, but making the values associated with each cover type more uniform across all images (Ding et al, 2015). The Atmospherically Resistant Vegetation Index 2 (ARVI2) (Adamu et al. 2018), where:

$$ARVI2 = -0.18 + 0.17\left(\frac{NIR - Red}{NIR + Red}\right)$$

could also be employed to achieve a similar result.

Chapter 3: Examining the Initial Response of a Sportfish Community to Hydrilla Invasion

To fully understand the nature of the relationship between hydrilla and sportfish in Lake Sinclair, we need to explore how hydrilla is impacting habitat quality, individual fish, and the broad-scale aquatic community. Hydrilla at high densities in small systems can create hypoxic conditions and reduced habitat quality (Bradshaw et al. 2015). Lake Sinclair is a large system, but it has many small coves off the main channel of the reservoir with dense hydrilla. Deploying dissolved oxygen and temperature loggers in these dense stands of hydrilla during the summer months could be highly informative, and the findings would determine whether aggressive hydrilla management action needs to be pursued. Additionally, analytical techniques such as stable isotope and otolith microchemistry analyses could provide valuable information as to whether hydrilla is causing dietary shifts in adult fish or preventing the shift to piscivory in young fish, as has been observed in previous studies (Valley and Bremigan 2002, Bettoli et al. 1992, Shelton et al. 1979).

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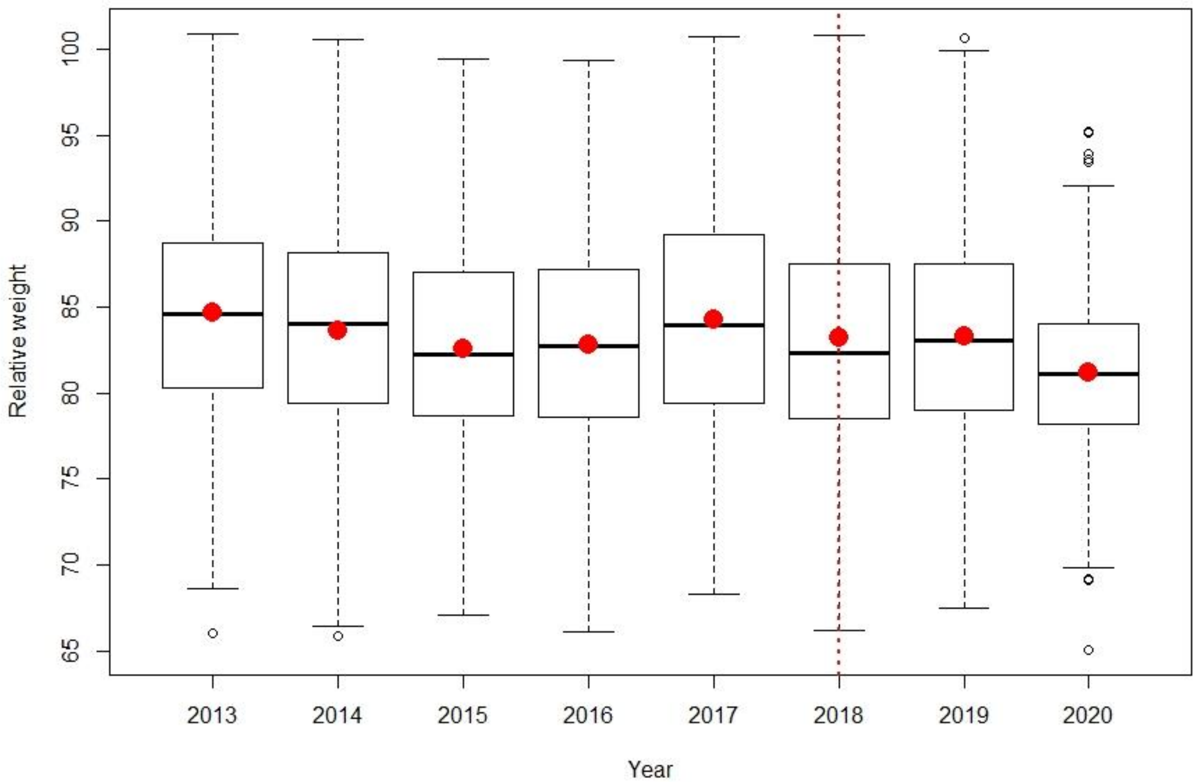
Table 4-1: Cost breakdown of recurring and one-time expenses for UAV and Side-Scan surveying.

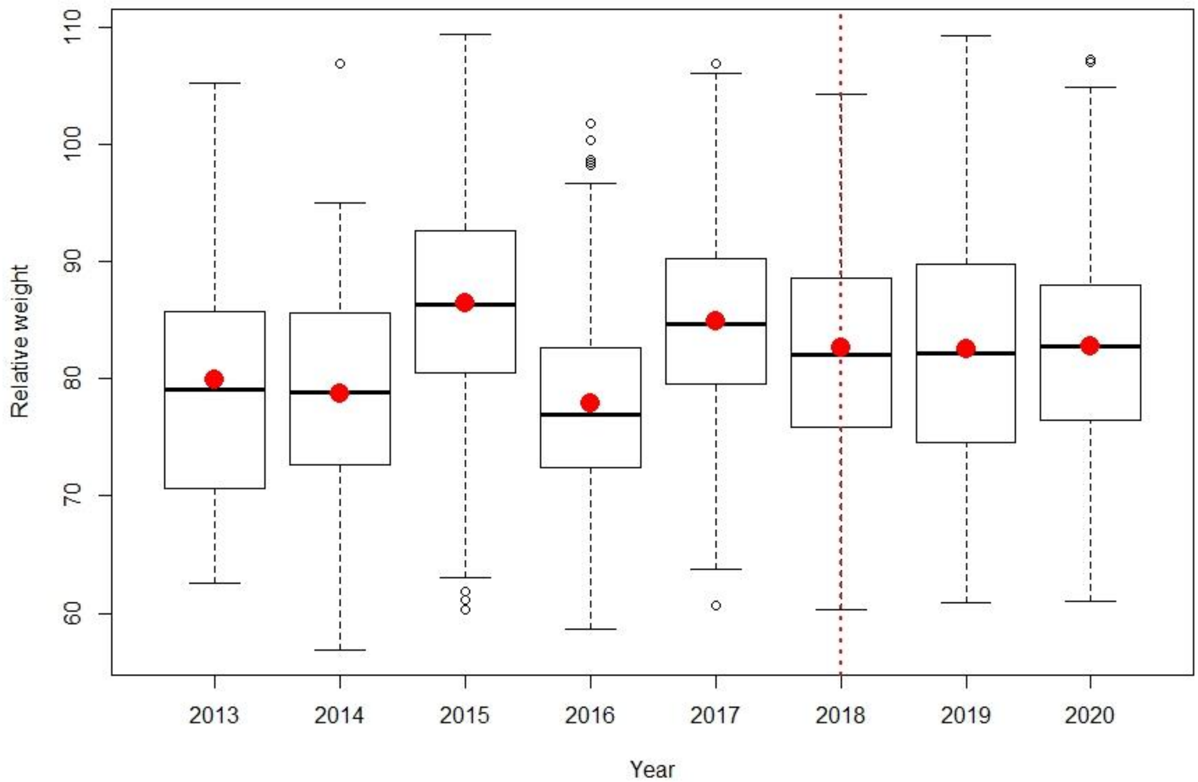
Product	Price	Recurring?
	UAV	
DJI Phantom 4 Pro V2 UAV	$1,600.00	No
4-Cell 15.2V High-Capacity Intelligent Flight Batteries (x3)	$555.00	No
iPad Mini	$500.00	No
Agisoft Metashape Pro License	$3,500.00	No
TOTAL	$6,155.00	
	Side-Scan Sonar	
Lowrance HDS-9 Head Unit and Active Imaging 3-in-1 Transducer	$2,200.00	No
BioBase Habitat+ Annual Subscription	$2,800.00	Yes
TOTAL	$5000.00	

APPENDIX A

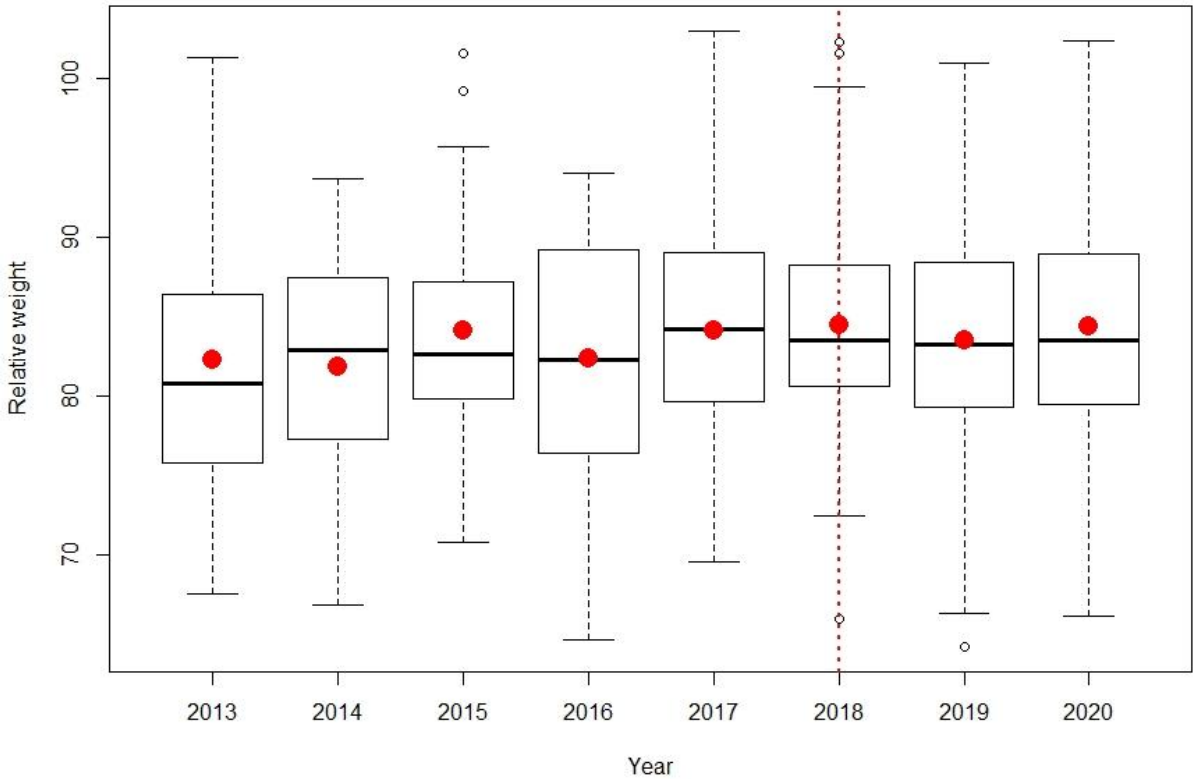
ADDITIONAL FIGURES



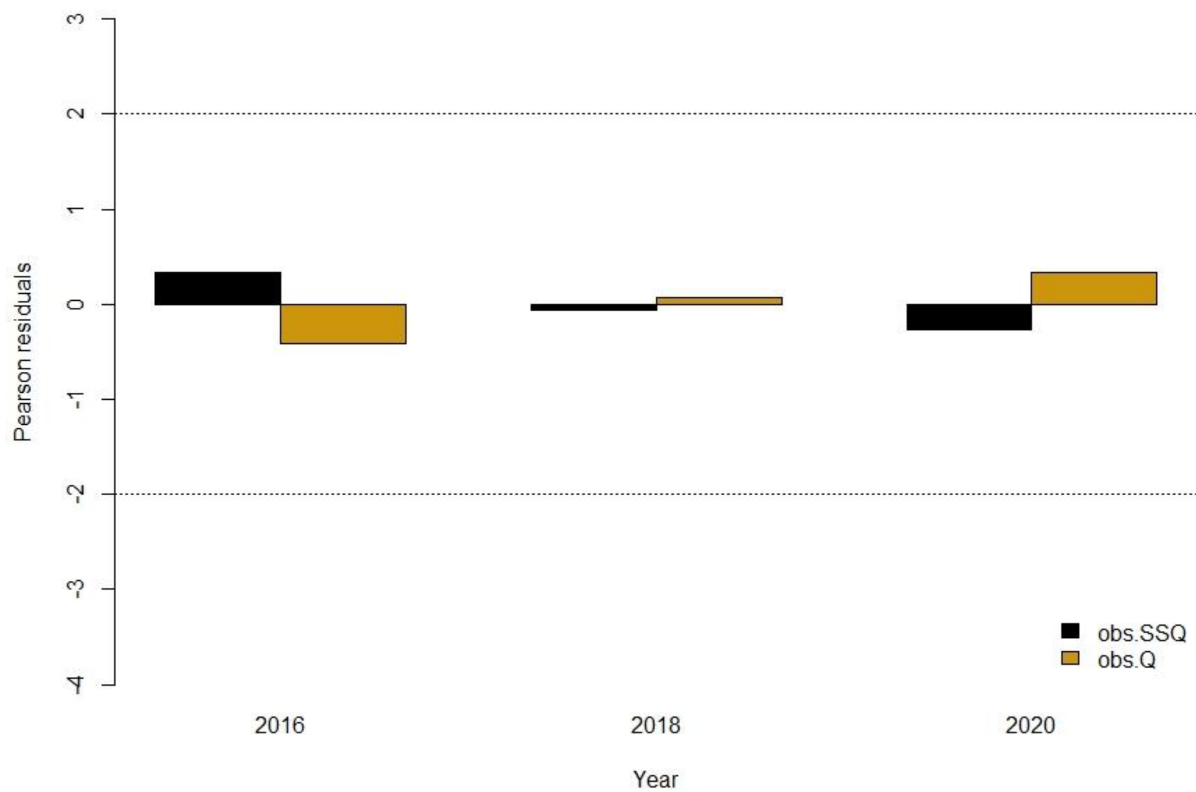
Appendix Figure 1. Box plot (mean [circles], median, quartiles, range, and outliers) of relative weight (Wr) of Lake Oconee black crappie from 2013 to 2020. The year hydrilla was introduced to Lake Sinclair is represented by the dashed red line.



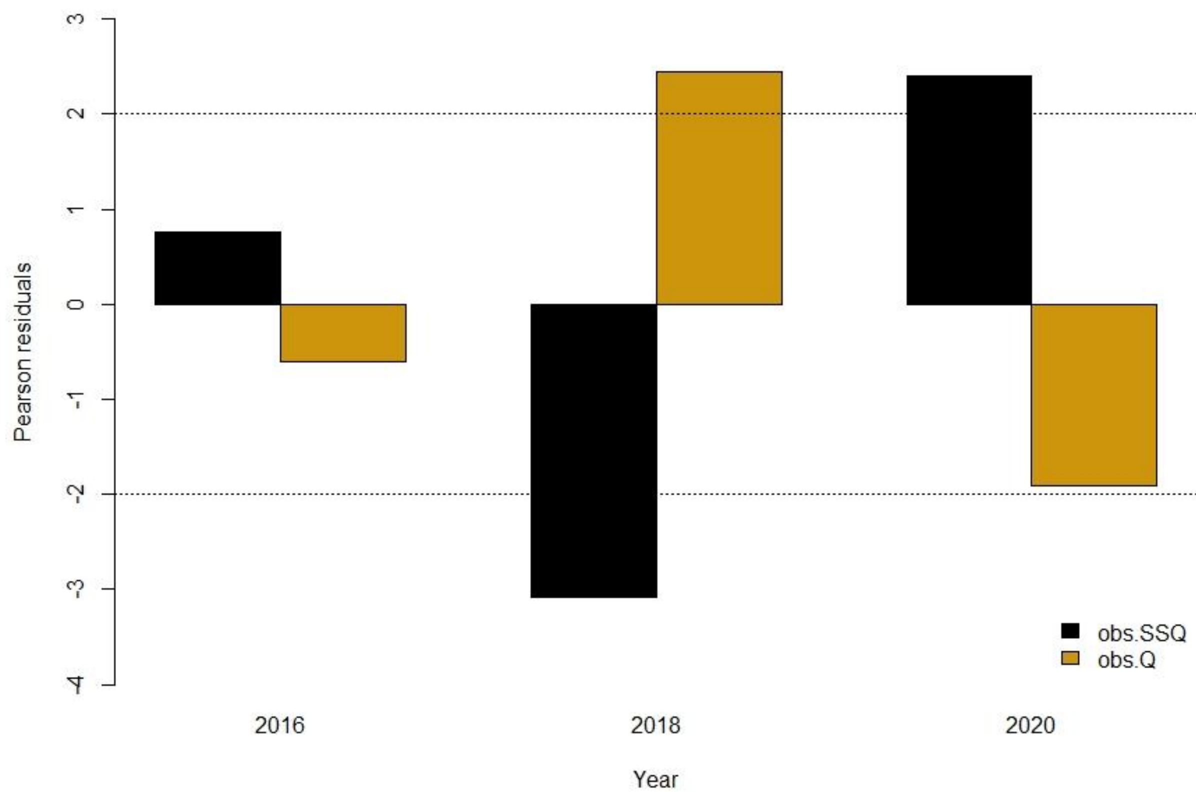
Appendix Figure 2. Box plot (mean [circles], median, quartiles, range, and outliers) of relative weight (Wr) of Lake Oconee bluegill from 2013 to 2020. The year hydrilla was introduced to Lake Sinclair is represented by the dashed red line.



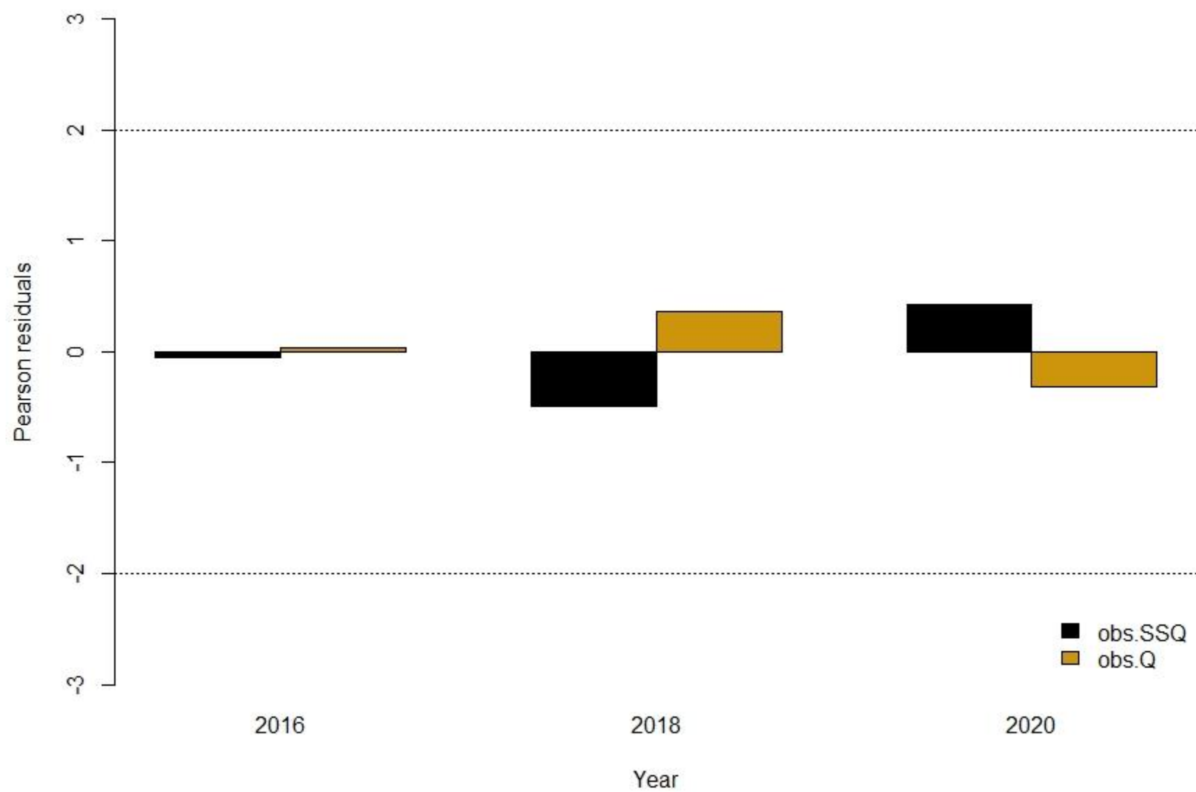
Appendix Figure 3. Box plot (mean [circles], median, quartiles, range, and outliers) of relative weight (Wr) of Lake Oconee redear sunfish from 2013 to 2020. The year hydrilla was introduced to Lake Sinclair is represented by the dashed red line.



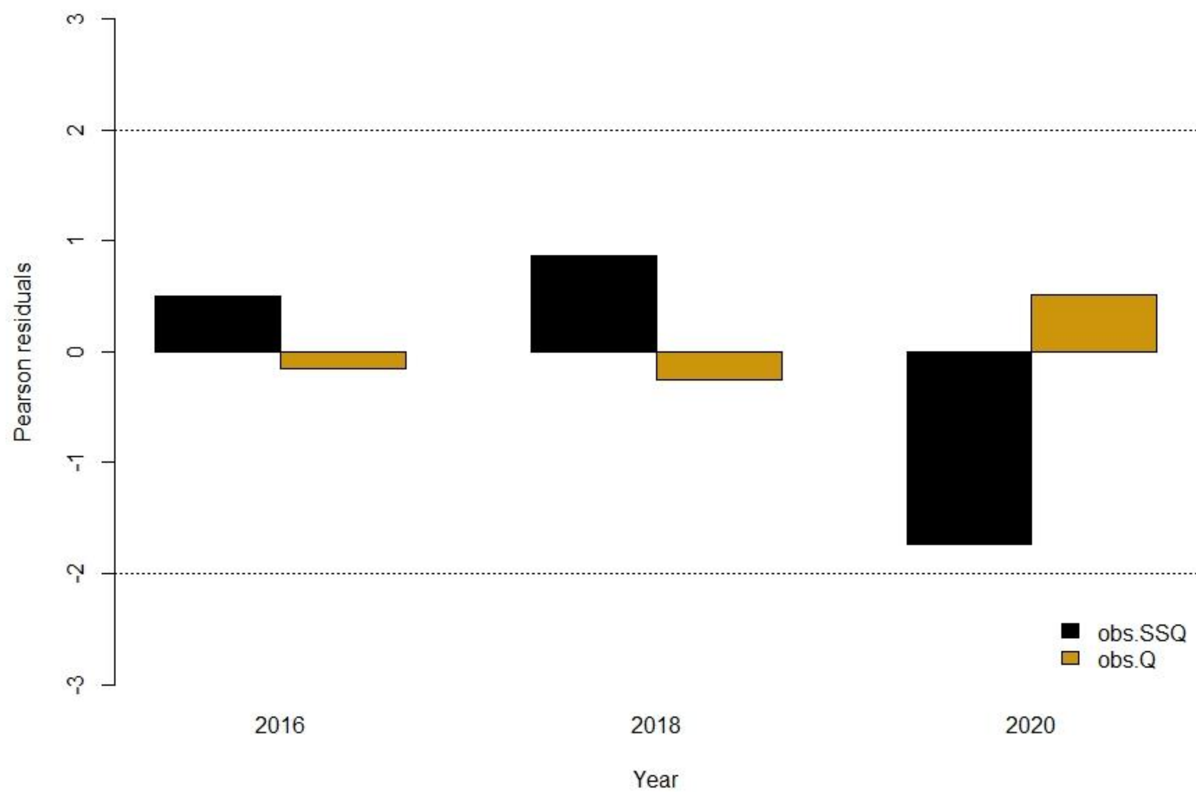
Appendix Figure 4. Pearson residuals for observed catch of sub-stock to quality (SSQ) and Quality+ (Q) largemouth bass in Lake Sinclair in 2016, 2018, and 2020. Pearson residuals indicate how much observed values deviate from the expected values generated by a Chi-Squared test, with values approaching +2 or -2 being considered significant.



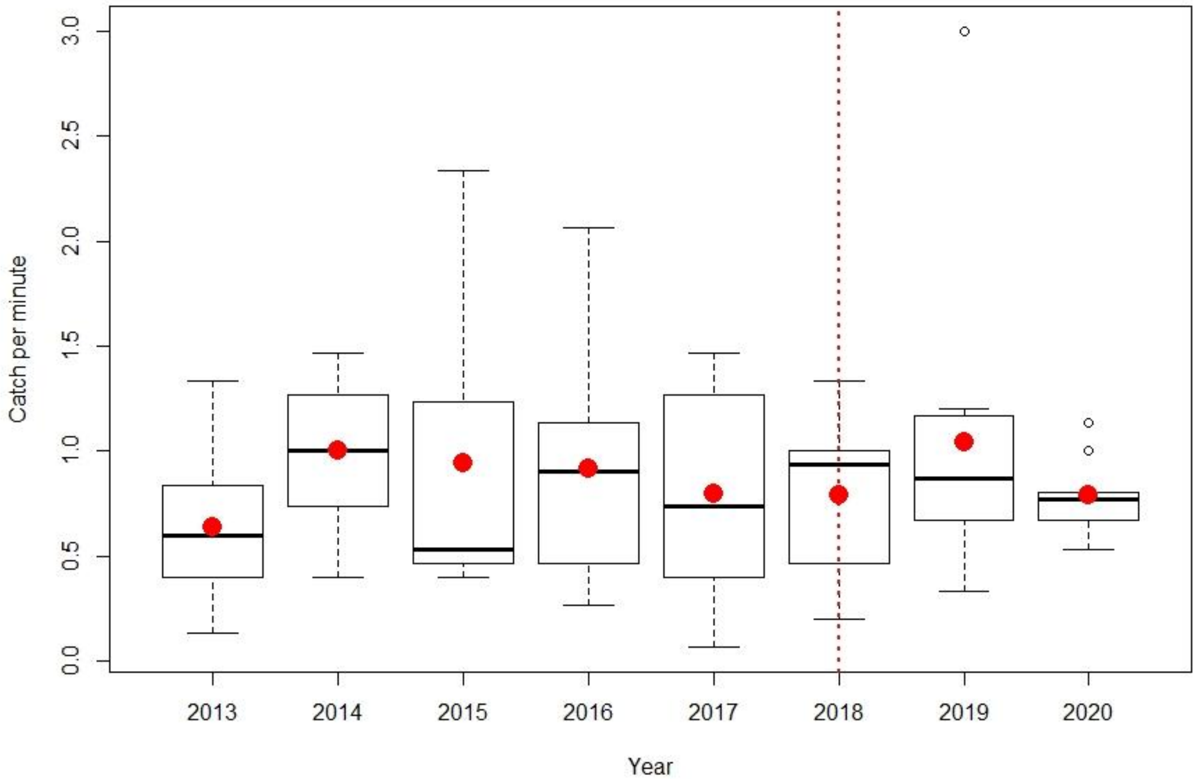
Appendix Figure 5. Pearson residuals for observed catch of sub-stock to quality (SSQ) and Quality+ (Q) largemouth bass in Lake Oconee in 2016, 2018, and 2020. Pearson residuals indicate how much observed values deviate from the expected values generated by a Chi-Squared test, with values approaching +2 or -2 being considered significant.



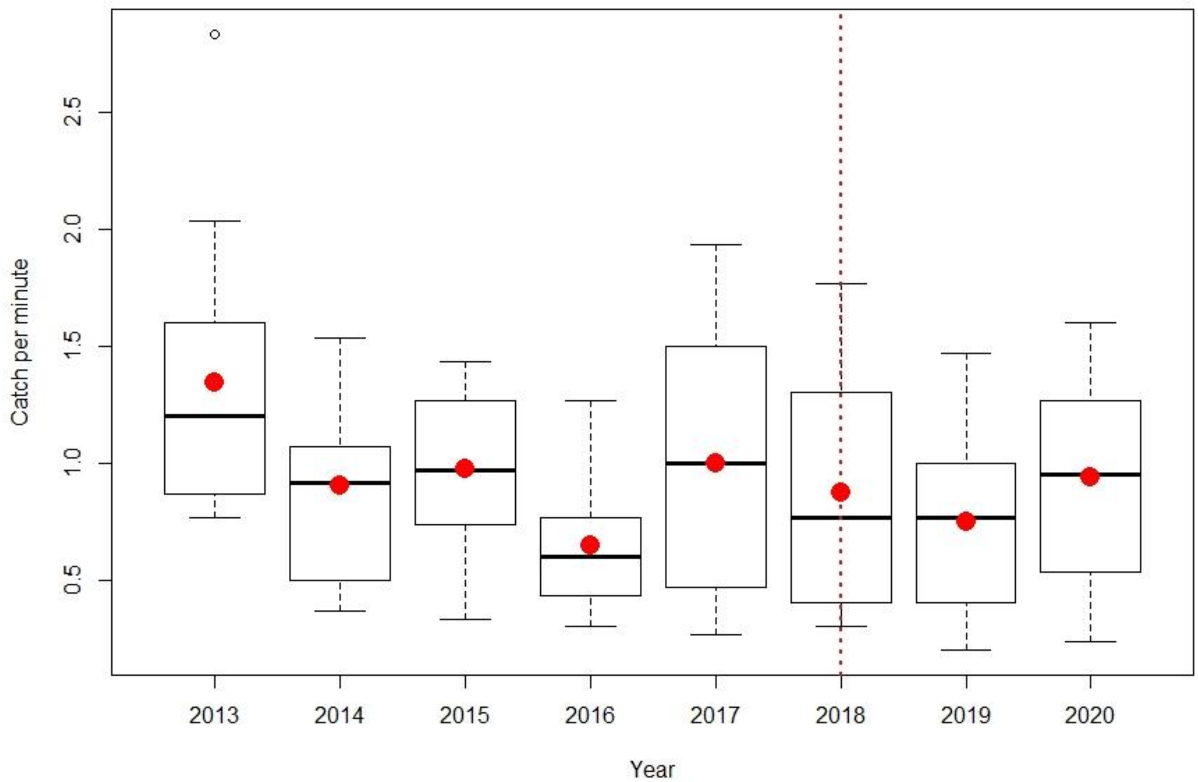
Appendix Figure 6. Pearson residuals for observed catch of sub-stock to quality (SSQ) and Quality+ (Q) black crappie in Lake Sinclair in 2016, 2018, and 2020. Pearson residuals indicate how much observed values deviate from the expected values generated by a Chi-Squared test, with values approaching +2 or -2 being considered significant.



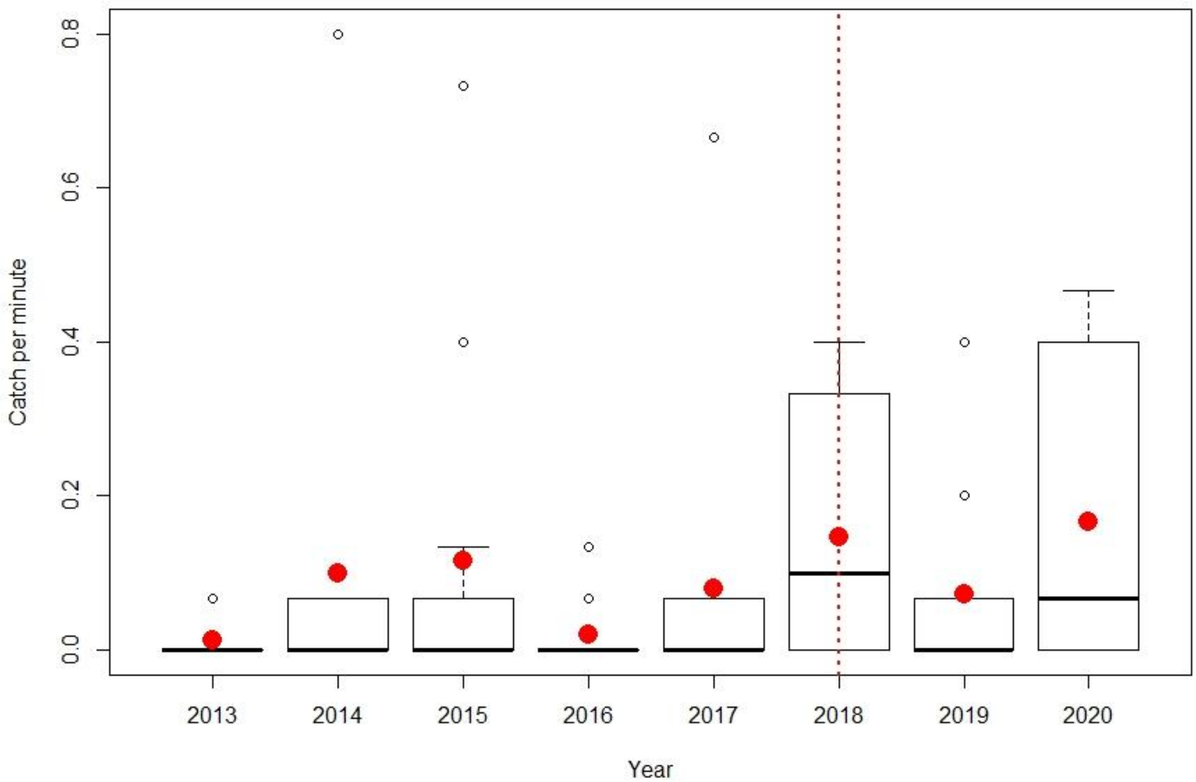
Appendix Figure 7. Pearson residuals for observed catch of sub-stock to quality (SSQ) and Quality+ (Q) black crappie in Lake Oconee in 2016, 2018, and 2020. Pearson residuals indicate how much observed values deviate from the expected values generated by a Chi-Squared test, with values approaching +2 or -2 being considered significant.



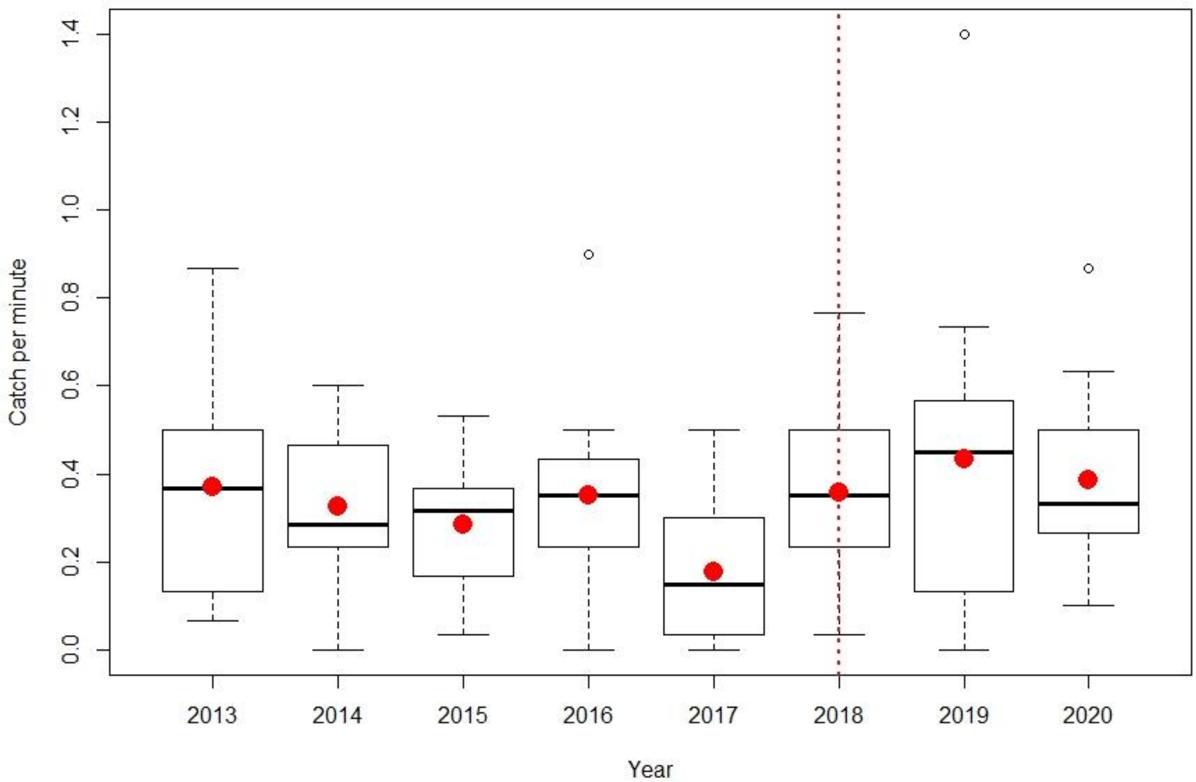
Appendix Figure 8. Box plot (mean [circles], median, interquartile range, and outliers) of catch per unit effort (CPUE) of Lake Sinclair largemouth bass from 2013 to 2020. The year hydrilla was introduced is represented by the dashed red line.



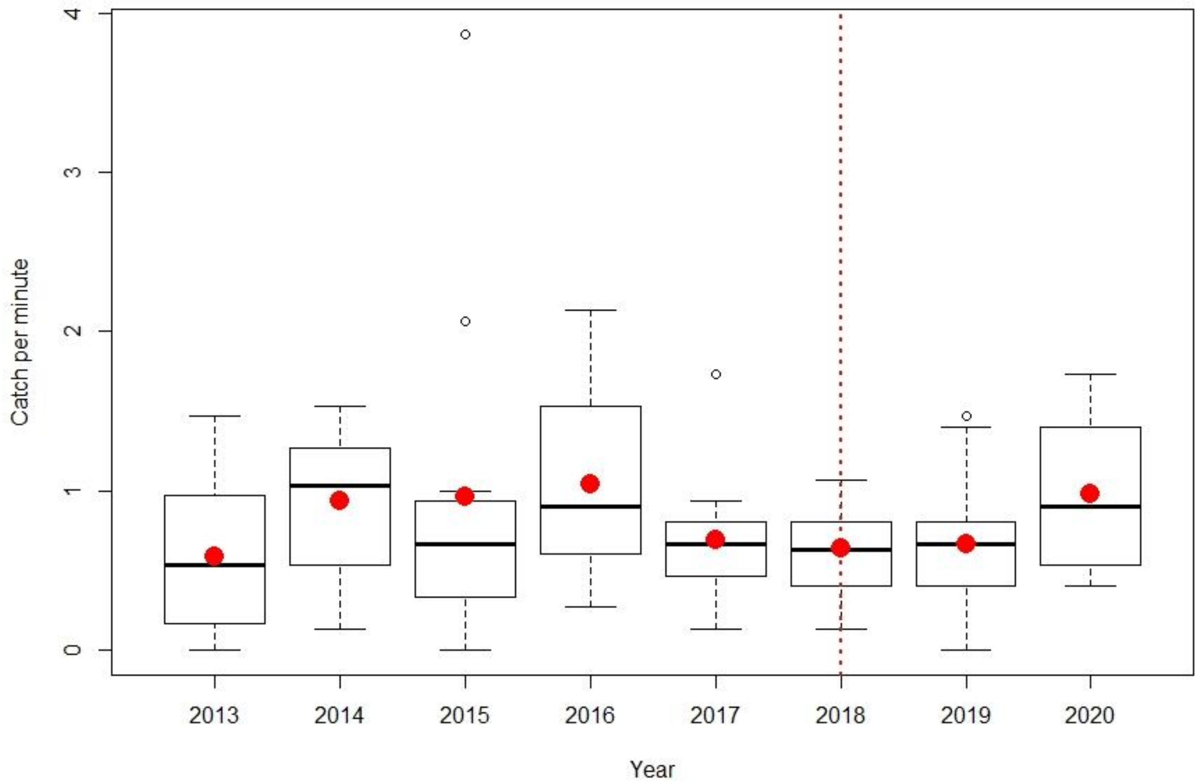
Appendix Figure 9. Box plot (mean [circles], median, interquartile range, and outliers) of catch per unit effort (CPUE) of Lake Oconee largemouth bass from 2013 to 2020. The year hydrilla was introduced is represented by the dashed red line.



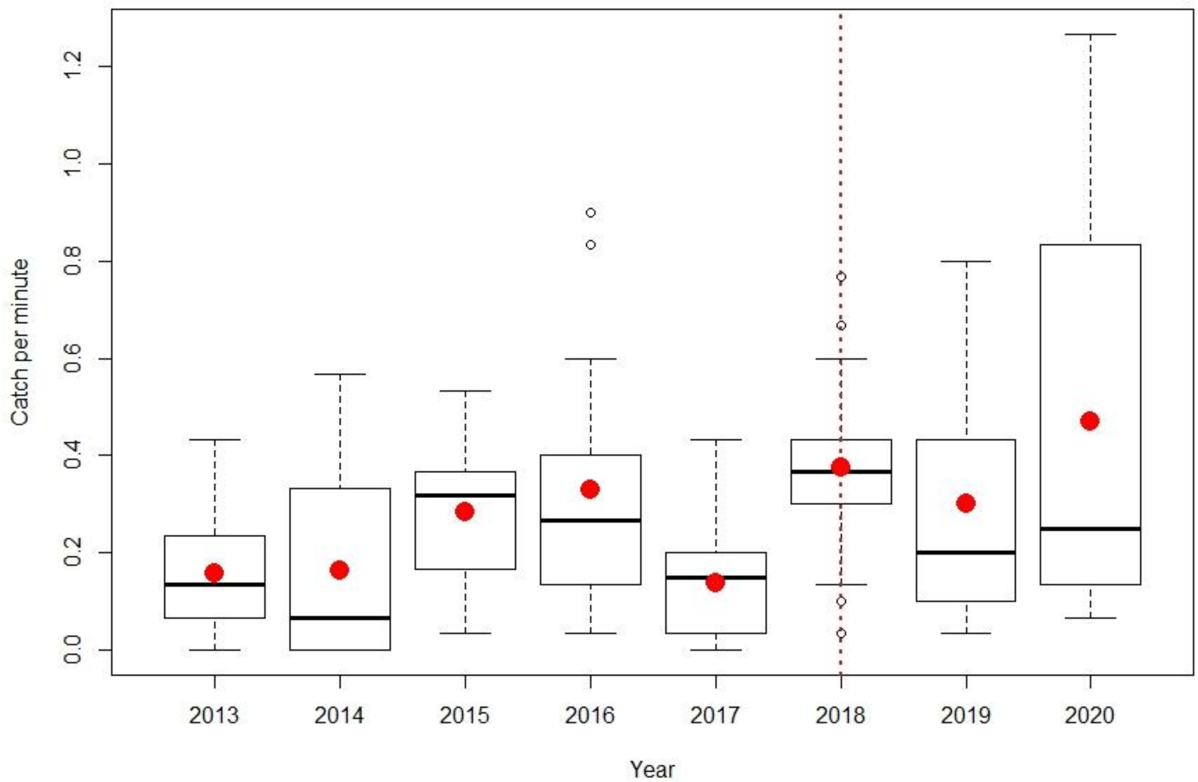
Appendix Figure 10. Box plot (mean [circles], median, interquartile range, and outliers) of catch per unit effort (CPUE) of Lake Sinclair black crappie from 2013 to 2020. The year hydrilla was introduced is represented by the dashed red line.



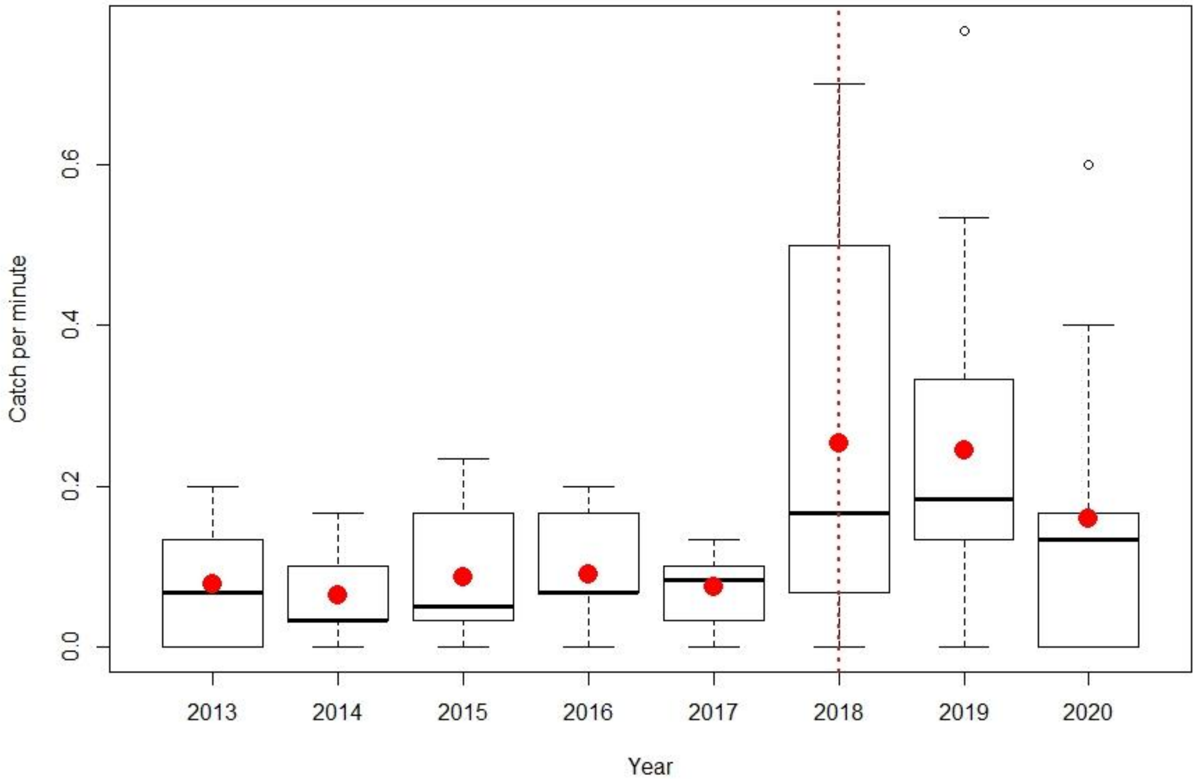
Appendix Figure 11. Box plot (mean [circles], median, interquartile range, and outliers) of catch per unit effort (CPUE) of Lake Oconee black crappie from 2013 to 2020. The year hydrilla was introduced is represented by the dashed red line.



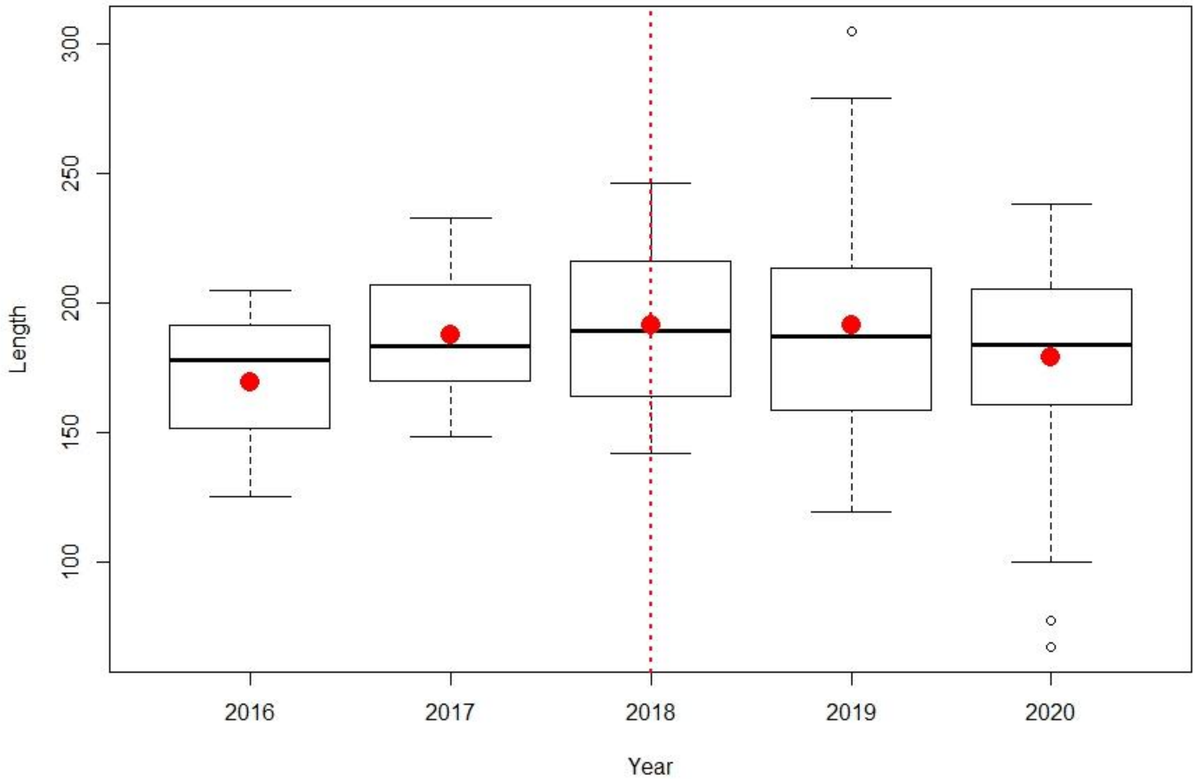
Appendix Figure 12. Box plot (mean [circles], median, interquartile range, and outliers) of catch per unit effort (CPUE) of Lake Sinclair bluegill from 2013 to 2020. The year hydrilla was introduced is represented by the dashed red line.



Appendix Figure 13. Box plot (mean [circles], median, interquartile range, and outliers) of catch per unit effort (CPUE) of Lake Oconee bluegill from 2013 to 2020. The year hydrilla was introduced is represented by the dashed red line.



Appendix Figure 14. Box plot (mean [circles], median, interquartile range, and outliers) of catch per unit effort (CPUE) of Lake Oconee redear sunfish from 2013 to 2020. The year hydrilla was introduced is represented by the dashed red line.



Appendix Figure 15. Box plot (mean [circles], median, interquartile range, and outliers) back-calculated length (mm) of age-1 largemouth bass in Lake Oconee from 2016 to 2020. The year hydrilla was introduced is represented by the dashed red line.