Ecological Analyses of Macroinvertebrate and Fish Species in Six Streams on a Louisiana Military Base from 2001 to 2019

Research Article

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Abstract

An in-depth ecological analysis of how and why the aquatic community changes over time was conducted for 6 streams on the Fort Polk military base in Louisiana using data collected from 2001 to 2019. Fort Polk is a unique location as nineteen first-order streams are located on the premises belonging to three separate drainages. The primary goal was to determine whether temporal or between-drainage variation has a larger effect on community structure. To accomplish this the effects of disturbance on fish and macroinvertebrate assemblages was determined temporally and between drainages. Several hypotheses were drawn from this: 1) temporally, assemblages exhibit fluctuations in diversity around disturbance events, but eventually recover to a base-state; 2) the 2012 drought caused a reduction in both fish and macroinvertebrate assemblage diversity and overall quantity in comparison to the other years; and 3) fish assemblages will vary between drainages more so than macroinvertebrate assemblages. A secondary goal was to determine the unique taxa of the drainages. It was found that 1) there was no recognizable pattern to assemblage diversity fluctuations and recovery; 2) the 2012 drought did not cause a significant reduction in fish or macroinvertebrate assemblages compared to subsequent years; and 3) fish assemblages differed by watershed more than macroinvertebrate assemblages, which often differed significantly by year. These ecological analyses present a more comprehensive picture of the ecosystems in the region.

Keywords: Headwater Streams, Ecology, Fish, Macroinvertebrates

1.0 Introduction

Headwaters provide unique habitats for a diverse array of aquatic organisms, as small streams often differ in chemical, biotic, and physical attributes (Lowe and Likens 2005; Meyer et al. 2007). These headwater species include both permanent residents and migrants who travel depending on the season or their particular life stages (Lowe and Likens 2005; Meyer et al. 2007). Even ephemeral streams can support diverse communities if the dry periods are predictable, allowing for the aquatic organisms to move downstream when the waters become too low (Meyer et al. 2007). Headwater streams can be important areas of refugia for
organisms from flow and temperature extremes, predators, competitors, and larger bodied invasive organisms (Meyer et al. 2007). Other habitat benefits of headwater streams include offering safer spawning and rearing areas than second or third-order streams, serving as a source of colonists and food, and establishing migration corridors in the landscape (Meyer et al. 2007).

Fish and macroinvertebrates are often the most prevalent assemblages in headwater streams, though amphibian and reptile communities may be present as well. North of Mexico, the southern US has more native fish than any other region in North America (Warren et al. 2000). However, many species are threatened or endangered; in Texas alone, 13 species are federally listed and 32 species are state listed (TPWD 2018). Habitat degradation is the primary cause of native fish decline, reducing and fragmenting ranges which isolates fish populations, limiting mobility, food, and gene flow (Angermeier 1995; Warren et al. 2000). Channelization, sedimentation, and flow alteration have contributed to a 125% increase in state and/or federally listed fish species of this region (Warren et al. 2000). In headwater streams, three landscape attributes influence fish population dynamics (Schlosser 1995). The first is the functional interactions at the transitional border between terrestrial and aquatic communities in terms of resource supply and predator-prey interaction, both spatially and temporally. The second is large-scale spatial habitat relationships in terms of resource use and fish movement. The third is presence or absence of refugia in the face of extreme environmental conditions in terms of fish survival, emigration, and immigration rates (Schlosser 1995). To generalize, the interconnected streams and the surrounding watershed habitats are highly important for the survival and proliferation of fish populations.

Macroinvertebrates are among the most diverse and ubiquitous of freshwater organisms (Clarke et al. 2008). In comparison to fish which are often higher-level predators, macroinvertebrates are an important link between energy sources (algae, detritus, etc.) and the top predators in aquatic systems (Allan 1995). They have a wide range of life-history traits, allowing them a broad range of habitat variability; with community dynamics recurrently changing in response to water quality and hydromorphological elements (Alvarez-Cabria et al. 2009). This wide range of habitat variability (i.e., tolerance) is the primary reason benthic macroinvertebrates are utilized as indicators of stream health and community structure.

The objective of this study was to analyze the ecological changes in the aquatic community, over time and geography, for 6 streams on Fort Polk military base in Louisiana using data collected from 2001 to 2019. The primary goal was to determine whether temporal or between-drainage variation had a larger effect on community structure. To accomplish this the effects of disturbance on fish and macroinvertebrate assemblages was determined by geography between drainages and temporal patterns. Several hypotheses were made: 1) assemblages exhibit temporal fluctuations in diversity around disturbance events, but eventually recover to a base-state, as areas prone to disturbance often have taxa with higher tolerance levels (Ross et al. 1985); 2) fish assemblages vary between drainages more so than macroinvertebrate assemblages, because of the differences in availability of movement between the two different assemblages (Williams et al. 2003; 2005); 3) the 2012 drought caused a reduction in both fish and macroinvertebrate assemblages in comparison to the other years, as droughts have been shown to reduce densities especially in macroinvertebrate assemblages (Hakala and Hartman 2004; Iversen et al. 1978). With the climate warming and extreme weather events more likely, it is
important to determine how this will affect aquatic populations, among others (Ummenhofer and Meehl 2017). Adding to the collective data on the effects of climate change locally can assist in decision-making at broader scales (Ummenhofer and Meehl 2017). A secondary goal was to determine the presence of habitat indicator species to acquire a clearer picture of the unique taxa in each stream. These ecological analyses serve to acquire a more comprehensive picture of the drainage basins in the region which can be utilized for future management of the streams on Fort Polk and the basins in general.

Fort Polk is a military base located in western-central Louisiana which has been active since 1939 (Dudley 2017). It consists of the primary base and a satellite base, the Joint Readiness Training Center (JRTC) and the Peason Ridge Training Area (PRTA), respectively. In terms of acreage, the JRTC encompasses approximately 51,000 acres, with over half of the property used continuously for military training and housing, while the PRTA and adjacent Kisatchie National Forest land covers over 200,000 acres. Nineteen first-order streams are located on the premises belonging to three separate drainages: the Calcasieu River, the Red River, and the Sabine River. These first-order streams are prone to flashy conditions which, in general, leads aquatic species to be fairly tolerant of disturbance (Conner and Suttkus 1986). As dictated by a previously established survey rotation, the following six streams were surveyed in 2019 and are the focus of this study: Whiskey Chitto Creek from the Calcasieu, Lyles and Tiger Creeks from the Red River, and Bayou Zourie, Dowden, and Martin Creeks from the Sabine River. Whiskey Chitto and Bayou Zourie are located on the JRTC, while the remaining streams are located on the PRTA; all originate on military land (Figure 1).

The Calcasieu River flows southward approximately 320 km from its origin, passing through the Kisatchie National Forest, until emptying in the Gulf of Mexico (USGS 2019). It is predominantly lowland coastal plain, and passes through areas of intensive petroleum refining (Douglas 1974, USGS 2019). The Sabine River travels approximately 820 km from northeastern Texas, through western Louisiana, to the Gulf of Mexico (USGS 2019). It also passes through areas of intensive petroleum refining activities. The Red River flows eastward at a fairly high gradient for approximately 2,200 km from the headwaters in Texas and New Mexico, through Oklahoma, Arkansas and Louisiana, until its confluence with the Mississippi River (Tyson 1981). Along with length and gradient, watershed area differs between the three rivers, with the Calcasieu, Sabine, and Red River watersheds encompassing approximately 22,000 square km, 25,000 square km, and 83,000 square km, respectively (Table 1).
Figure 1: The 6 Fort Polk tributaries surveyed in 2019.
Table 1: Length of streams and rivers, and area of watersheds affiliated with Fort Polk. Site (km) length is the distance from the beginning of the tributary to the surveyed site; whereas total length (km) is the distance from the beginning of the tributary to its confluence with a larger tributary.

<table>
<thead>
<tr>
<th>Stream Name</th>
<th>Site (km)</th>
<th>Total (km)</th>
<th>Watershed Area Site (sq km)</th>
<th>River Name</th>
<th>Length Miles</th>
<th>Km</th>
<th>Watershed Area Acres</th>
<th>Sq Km</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bayou Zourie</td>
<td>4.97</td>
<td>30.697</td>
<td>12.299 Sabine</td>
<td>510</td>
<td>821</td>
<td>6300911</td>
<td>25498.9</td>
<td></td>
</tr>
<tr>
<td>Dowden</td>
<td>6.263</td>
<td>15.647</td>
<td>19.64 Sabine</td>
<td>510</td>
<td>821</td>
<td>6300911</td>
<td>25498.9</td>
<td></td>
</tr>
<tr>
<td>Lyles</td>
<td>2.838</td>
<td>12.856</td>
<td>9.645 Red</td>
<td>1360</td>
<td>2189</td>
<td>20533316</td>
<td>83095.45</td>
<td></td>
</tr>
<tr>
<td>Martin</td>
<td>8.078</td>
<td>13.843</td>
<td>25.707 Sabine</td>
<td>510</td>
<td>821</td>
<td>6300911</td>
<td>25498.9</td>
<td></td>
</tr>
<tr>
<td>Tiger</td>
<td>2.259</td>
<td>2.958</td>
<td>2.393 Red</td>
<td>1360</td>
<td>2189</td>
<td>20533316</td>
<td>83095.45</td>
<td></td>
</tr>
<tr>
<td>Whiskey Chitto</td>
<td>19.924</td>
<td>139.015</td>
<td>74.179 Calcasieu</td>
<td>200</td>
<td>322</td>
<td>5368981</td>
<td>21727.51</td>
<td></td>
</tr>
</tbody>
</table>

The Calcasieu headwater tributaries on Ft. Polk are slow flowing, black water streams with loamy soil. It is fairly common to encounter pools up to 2 meters deep at high water along with occasional rocky riffles (Felix 2012). Whiskey Chitto is a large tributary compared to the other six headwater streams, extending past the borders of Fort Polk for an approximately 140 km and flowing directly into the Calcasieu River. The watershed area draining to the survey location is approximately 74 square km (Table 1). The Sabine and Red River headwater tributaries on Peason Ridge are shallower in general with loamy to sandy soil (Felix 2012). The length of these tributaries range from approximately 3 km for Tiger Creek to approximately 30 km for Bayou Zourie, with average length for the three remaining streams around 14 km (Table 1). The watershed areas draining to the survey locations range from 2 square km for Tiger Creek to 26 square km for Martin Creek (Table 1). In addition to their relatively shorter length, these five streams feed directly into other tributaries, not the main river. Tiger Creek in particular has the most intermediate tributaries before reaching the Red River. It flows directly into Odom Creek, which empties into Little Sandy Creek, Kisatchie Bayou, Old River, Cane River, until finally reaching the Red River by Colfax, LA. This confluence of tributaries may influence the species found within the streams. Wooded riparian buffers surround the vast majority of stream area, with dominant terrestrial species associated with loblolly and longleaf pine as well as those of bald cypress tree groves (Williams et al. 2005). The prevalence of wood debris in the rivers creates habitat for fish and macroinvertebrates. The few unvegetated stream banks occur predominantly along sandbars (Felix 2012).

Fort Polk has undergone numerous anthropogenic disturbances including extensive logging prior to 1939, localized logging in 2003 around Odom and Tiger Creeks of the Red River, and the construction of roads, bridges, and culverts across the military base (Williams et al. 2005; Dudley 2017). During the logging prior to military ownership, timber bridges were established across many streams of the area for transportation of goods; however, more durable bridges and major surfaced roadways were required for military activities (Grubh, 2006). Culverts became one of the most common crossing types, with round single culverts ranging from 30 inches to 6 feet used because of their ease of placement and high weight capabilities (Dudley 2017). To install these culverts, the bed area is excavated and replaced with washed gravel, then the culvert is placed, backfilled, and paved. However, more recent innovations have tried to limit the impacts to natural hydrology,
incorporating arch culverts and modified placement techniques for box culverts (Grubh 2006). Fort Polk underwent further modification with the construction of a range complex to facilitate combined arms training in association with the Digital Multipurpose Battle Area Course (DMPBAC) in 2003 (Dudley 2017). The area around Tiger and Odom creeks was clear-cut to create firing lanes and further road systems and stream crossings for the transport of large equipment. The riparian zone around these streams was decimated, though it began to recover after construction concluded (Dudley 2017).

Prior to the military presence on Fort Polk, the longleaf pine ‘fire climax’ community was maintained by wildfires started from seasonal lightning strikes (Bridges and Orzell 1989). As wildfires are difficult to control and can be devastating, a prescribed two- to three-year burn cycle was implemented to maintain the forested regions (Williams et al. 2005). Current base activities utilize the roads frequently, and the streams are fed by surface water runoff across the entire base. However, there are still significant amounts of intact stream characteristics, even in areas of high disturbance (Williams et al. 2005). This means that the areas directly adjacent to and above these riparian corridors are relatively intact with respect to soil and vegetation, filtering the majority of pollutants before it enters the stream system. One reason for the prevalence of intact riparian corridors is the U.S. Army is mandated to ensure their activities do not have a negative impact on the aquatic communities and water chemistry on their training bases. Stream health has been monitored using index of biotic integrity (IBI) measurements since 2001. Because of this historic data and the need to ensure the overall health of the streams, the analysis of ecological changes in the aquatic community, temporally and geographically brought greater understanding and potential efficiency to the management of the riparian areas.

2.0 Methods

2.1 Collection of 2019 Samples and Habitat Data

We surveyed six streams in the summer of 2019, in late May, to maximize the potential species collected as both fish and macroinvertebrates peak in diversity during the summer months (Williams et al. 2007). Study streams were determined from the ongoing monitoring rotation in years past on the Fort Polk military base (19 total streams with six streams sampled per year on a three-year rotation). We utilized protocols established during previous surveys of Fort Polk to maintain consistency (Williams et al. 2002, 2005, 2007; Felix 2012; Dudley 2017). Each survey location was approximately 100 meters of stream section in length beginning at a stream crossing for ease of access and for the monitoring of military activities on the stream systems. Each 100-meter section consisted of substrate, woody debris, and multiple mesohabitat characteristics (including riffles, runs, and pools) of the overall stream reach; therefore, each site was representative of conditions within the stream.

We recorded habitat data using the Texas Commission on Environmental Quality Surface Water Quality Monitoring - Habitat Assessment Worksheet B Part III of III (Habitat Quality Index, https://www.tceq.texas.gov/waterquality/monitoring/swqm_procedures.html), as Louisiana did not have a comparable assessment form at the time of the study. We collected fish using backpack electrofisher and fish landing nets starting from downstream and moving upstream for a 100-meter reach. The collected fish were
anesthetized in the field with tricaine methanesulfonate and preserved in a 10% formalin solution. We collected macroinvertebrates using two methods: grab sampling (D-frame kick nets) and substrate samples (Surber). Dip nets were used by 1 to 2 people in approximately 20 subsamples in a variety of mesohabitats per site (Barbour et al. 1992). We took a single Surber sample for five minutes per site, as riffles were often a minor mesohabitat within the reach. The Surber was placed along a riffle, facing upstream, and the substrate was disturbed to dislodge and capture any buried macroinvertebrates. Surber sampling offers the potential of detecting rare or elusive taxa not represented by dip net sampling (Storey et al. 1991). We preserved the macroinvertebrate samples in the field in 70% ethanol.

We rinsed fish samples after 3 days to remove the formalin solution and identified to species. The kick net and Surber samples were processed in the lab by lab technicians following TCEQ protocols and the macroinvertebrates identified to family. This level of identification was determined to be sufficient for the statistical analyses to be performed (Bowman and Bailey 1997). All identified organisms were preserved in 70% ethanol.

2.2 Previous Data

We utilized data from previous survey years (2001-2004, 2012, and 2016) for the statistical analyses. The 2001 fish and macroinvertebrate data were acquired with a 3-man field crew from all sites except Bayou Zourie, which was not surveyed until 2012. Subsequent collection years had field crews of 4 to 6 individuals. The 2002 macroinvertebrate data were collected from all sites, sans Bayou Zourie, and fish data were collected from Whiskey Chitto only. In 2001 and 2002, fish were identified in the field, enumerated, then released (except for voucher specimens). In 2003 and 2004, macroinvertebrate data were collected from Whiskey Chitto only. Fish and macroinvertebrate data from 2012 and 2016 were acquired from all sites following the same procedures used in 2019.

2.3 Statistical analyses

We used all the yearly data in these analyses, unless otherwise specified. To determine if there were habitat indicators for both fish and macroinvertebrate taxa, we used the Indicator Species Analysis (ISA). Habitat factors and geographic levels used include stream site (such as Tiger Creek), watershed or river drainage (such as the Red River drainage), and hydrology (such as discharge using USGS gauges on the Calcasieu and Sabine Rivers). We used Nonmetric Multidimensional Scaling to determine the relative similarity in assemblage structure between sites over time and geographically. The software, PC-ORD (version 6, MjM Software Design, Gleneden Beach, OR), was utilized for the indicator species analysis (ISA) and the nonmetric multidimensional scaling (NMS). When running NMS, we used Sorenson and all other default settings, as there was not a specific need to modify the settings. We graphed the data in the appropriate dimension, utilizing Excel (Office 365, Microsoft Corporation) for 2D graphs and SigmaPlot (Version 10.0, Systat Software Inc., San Jose, CA) for 3D graphs. We drew polygons to delineate sites, watersheds, and years. We utilized Multi-response Permutation Procedures (MRPP) to test for significance between the watersheds and between the years for fish, macroinvertebrate, and the combined fish and macroinvertebrate data. As Sorenson would not allow for negative numbers, we used Euclidean in MRPP. For the combined fish and
macroinvertebrate data, we did not use 2003 and 2004 yearly data for Whiskey Chitto, as well as 2002 data for Dowden, Lyles, Martin, and Tiger Creeks, as there was no fish data for those years at those sites.

3.0 Results

Spanning 18 noncontinuous years, researchers samples 6 streams for fish and macroinvertebrate assemblages on Fort Polk. Over 15,000 individuals were collected in total, with approximately 1,700 of those organisms being fish (Table 2). Over 13,000 macroinvertebrates were collected, with nearly 6,500 from 2019 alone (Table 2).

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</thead>
<tbody>
<tr>
<td>Fish</td>
<td>501</td>
<td>213</td>
<td>366</td>
<td>487</td>
<td>217</td>
<td>731</td>
<td>52</td>
<td>711</td>
<td>4424</td>
<td>6905</td>
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<td></td>
<td></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>Macroinvertebrates</td>
<td>954</td>
<td>4389</td>
<td>365</td>
<td>1163</td>
<td>6481</td>
<td>13352</td>
<td></td>
<td></td>
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<tr>
<td>Chironomidae</td>
<td>549</td>
<td>1169</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>Total Organisms</td>
<td>1455</td>
<td>4602</td>
<td>731</td>
<td>1630</td>
<td>6698</td>
<td>15116</td>
<td></td>
<td></td>
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</tr>
</tbody>
</table>

Table 2: Quantities of total organisms collected per year

Relating to differences in fish species by watershed, we determined that 6 fish species were significant indicators using Indicator Species Analysis (ISA): *Cyprinella venusta* (p = 0.004) and *Percina sciera* (p = 0.029) in the Calcasieu drainage system, and *Etheostoma artesiae* (p = 0.033), *Lepomis cyanellus* (p = 0.002), *Luxilus chrysocephalus* (p = 0.011), and *Noturus phaeus* (p = 0.01) in the Red River drainage system (Table 3). No indicator species were determined for the Sabine drainage system.

<table>
<thead>
<tr>
<th>Species</th>
<th>Max Group</th>
<th>Observed</th>
<th>Mean</th>
<th>St. dev.</th>
<th>p value</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Cyprinella venusta</em></td>
<td>Calcasieu</td>
<td>88.2</td>
<td>36.6</td>
<td>13.6</td>
<td>0.004</td>
</tr>
<tr>
<td><em>Percina sciera</em></td>
<td>Calcasieu</td>
<td>56.5</td>
<td>27.6</td>
<td>11.29</td>
<td>0.029</td>
</tr>
<tr>
<td><em>Etheostoma artesiae</em></td>
<td>Red</td>
<td>51.1</td>
<td>25.2</td>
<td>10.93</td>
<td>0.033</td>
</tr>
<tr>
<td><em>Lepomis cyanellus</em></td>
<td>Red</td>
<td>76.2</td>
<td>33.4</td>
<td>10.87</td>
<td>0.002</td>
</tr>
<tr>
<td><em>Luxilus chrysocephalus</em></td>
<td>Red</td>
<td>60.5</td>
<td>24.1</td>
<td>11.17</td>
<td>0.011</td>
</tr>
<tr>
<td><em>Noturus phaeus</em></td>
<td>Red</td>
<td>62.5</td>
<td>22.2</td>
<td>10.81</td>
<td>0.010</td>
</tr>
</tbody>
</table>

Table 3: Fish Indicator Species Analysis (ISA) by watershed.

We determined 10 macroinvertebrate families to be significant indicators, 9 of which were indicators for the Calcasieu drainage system (Table 4). Hemiptera Corydalidae (p = 0.04) was significant for the Sabine River watershed, while none were significant for the Red River watershed. For the Calcasieu River watershed, Coleoptera Gyrinidae (p = 0.029), Diptera Empididae (p = 0.006), Diptera Ephyridae (p = 0.026), Ephemeroptera Isonychiidae (p = 0.014), Ephemeroptera Tricyrtiidae (p = 0.041), Hemiptera Veliidae (p = 0.047), Odonata Aeshnidae (p = 0.028), Odonata Gomphidae (p = 0.041), and Trichoptera Hydropsychidae (p = 0.002) were all indicator species (Table 4).
Table 4: Macroinvertebrate Indicator Species Analysis (ISA) by watershed.

<table>
<thead>
<tr>
<th>Order Family</th>
<th>Max Group</th>
<th>Observed Mean</th>
<th>St. dev.</th>
<th>p value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Coleoptera Gyrinidae</td>
<td>Calcasieu</td>
<td>52.7</td>
<td>28</td>
<td>10.13</td>
</tr>
<tr>
<td>Diptera Empididae</td>
<td>Calcasieu</td>
<td>46.7</td>
<td>16.7</td>
<td>8.41</td>
</tr>
<tr>
<td>Diptera Ephyridae</td>
<td>Calcasieu</td>
<td>40</td>
<td>12.7</td>
<td>6.81</td>
</tr>
<tr>
<td>Ephemeroptera Isonychiidae</td>
<td>Calcasieu</td>
<td>53.2</td>
<td>22.8</td>
<td>10.59</td>
</tr>
<tr>
<td>Ephemeroptera Tricyrthidae</td>
<td>Calcasieu</td>
<td>39.7</td>
<td>20.1</td>
<td>9.27</td>
</tr>
<tr>
<td>Hemiptera Veliidae</td>
<td>Calcasieu</td>
<td>56.8</td>
<td>33</td>
<td>12.53</td>
</tr>
<tr>
<td>Odonata Aeshnidae</td>
<td>Calcasieu</td>
<td>53.8</td>
<td>31.7</td>
<td>9.11</td>
</tr>
<tr>
<td>Odonata Gomphidae</td>
<td>Calcasieu</td>
<td>60.6</td>
<td>36.1</td>
<td>11.52</td>
</tr>
<tr>
<td>Trichoptera Hydropsychidae</td>
<td>Calcasieu</td>
<td>83.9</td>
<td>32.9</td>
<td>11.89</td>
</tr>
<tr>
<td>Hemiptera Corydalidae</td>
<td>Sabine</td>
<td>50.5</td>
<td>26</td>
<td>10.58</td>
</tr>
</tbody>
</table>

Relating to differences in fish species by year, we determined that 4 fish species were significant indicators using ISA: *Lepomis punctatus* (p = 0.045) and *Lythrurus umbratilis* (p = 0.003) in 2001, *Ameiurus natalis* (p = 0.034) in 2012, and *Lepomis miniatus* (p = 0.049) in 2016 (Table 5). We did not find any indicator species for the year 2019 (Table 8).

Table 5: Fish Indicator Species Analysis (ISA) by year.

<table>
<thead>
<tr>
<th>Species</th>
<th>Max Group</th>
<th>Observed Mean</th>
<th>St. dev.</th>
<th>p value</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Lepomis punctatus</em></td>
<td>2001</td>
<td>40</td>
<td>15.1</td>
<td>10.6</td>
</tr>
<tr>
<td><em>Lythrurus umbratilis</em></td>
<td>2001</td>
<td>68.2</td>
<td>29.8</td>
<td>9.86</td>
</tr>
<tr>
<td><em>Ameiurus natalis</em></td>
<td>2012</td>
<td>47.6</td>
<td>23.8</td>
<td>10.39</td>
</tr>
<tr>
<td><em>Lepomis miniatus</em></td>
<td>2016</td>
<td>47.3</td>
<td>25.7</td>
<td>10.91</td>
</tr>
</tbody>
</table>

We determined 14 macroinvertebrate families to be significant indicators. The year 2002 had 9 families, *Diptera Tabanidae* (p = 0.043), *Ephemeroptera Caenidae* (p = 0.015), *Hemiptera Gerridae* (p = <0.001), *Megaloptera Sialidae* (p = 0.015), *Odonata Coenagrionidae* (p = <0.001), *Odonata Corduliidae* (p = 0.01), *Odonata Gomphidae* (p = 0.008), *Odonata Libellulidae* (p = 0.048), and *Trichoptera Leptoceridae* (p = 0.001; Table 6). *Amphipoda Gammeridae* (p = 0.015) and *Ephemeroptera Ameletidae* (p = 0.024) were indicators of 2012. The remaining 3 families were indicators of the year 2019 (Table 6).
Using NMS, we determined that two dimensional graphs were considered optimal for the combined fish and macroinvertebrates data and the macroinvertebrate data both by watershed and by year (Figures 3-6). Three dimensional graphs were considered optimal for the fish data both by watershed and by year (Figures 7-8). We used MRPP to test for significant differences. For the combined data of fish and macroinvertebrates categorized by watershed and for the macroinvertebrate data categorized by watershed, we found a substantial overlap within and between the sites and watersheds, as shown with the overlapping polygons; the MRPP further supports this with non-significant p-values of 0.473 and 0.239 for the combination data and macroinvertebrate data, respectively, indicating little difference within and between the sites and watersheds (Figures 3 and 5). For the fish data categorized by watershed, there is little overlap of polygons in three-dimensional space, with a p-value of 0.014 (Figure 7). For all the data categorized by year, fish, macroinvertebrate, and the combination, there is very little overlap; with p-values of 0.001, <0.001, and <0.001, respectively (Figures 4, 6, and 8).

### Table 6: Macroinvertebrate Species Analysis (ISA) by year.

<table>
<thead>
<tr>
<th>Order Family</th>
<th>Max Group</th>
<th>Observed</th>
<th>Mean</th>
<th>St. dev.</th>
<th>p value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Diptera Tabanidae</td>
<td>2002</td>
<td>47.7</td>
<td>22.9</td>
<td>11.45</td>
<td>0.043</td>
</tr>
<tr>
<td>Ephemeroptera Caenidae</td>
<td>2002</td>
<td>70.1</td>
<td>37.1</td>
<td>11.71</td>
<td>0.015</td>
</tr>
<tr>
<td>Hemiptera Gerridae</td>
<td>2002</td>
<td>71.4</td>
<td>23.7</td>
<td>9.55</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Megaloptera Sialidae</td>
<td>2002</td>
<td>58.5</td>
<td>25</td>
<td>11.08</td>
<td>0.015</td>
</tr>
<tr>
<td>Odonata Coenagrionidae</td>
<td>2002</td>
<td>92.1</td>
<td>33.3</td>
<td>13.74</td>
<td>0.001</td>
</tr>
<tr>
<td>Odonata Corduliidae</td>
<td>2002</td>
<td>62.1</td>
<td>24.6</td>
<td>10.87</td>
<td>0.010</td>
</tr>
<tr>
<td>Odonata Gomphidae</td>
<td>2002</td>
<td>66</td>
<td>32.1</td>
<td>11.2</td>
<td>0.008</td>
</tr>
<tr>
<td>Odonata Libellulidae</td>
<td>2002</td>
<td>51</td>
<td>26.3</td>
<td>11.59</td>
<td>0.048</td>
</tr>
<tr>
<td>Trichoptera Leptoceridae</td>
<td>2002</td>
<td>73.8</td>
<td>23.4</td>
<td>11.76</td>
<td>0.001</td>
</tr>
<tr>
<td>Amphipoda Gammeridae</td>
<td>2012</td>
<td>51.7</td>
<td>23.5</td>
<td>9.49</td>
<td>0.015</td>
</tr>
<tr>
<td>Ephemeroptera Ameletidae</td>
<td>2012</td>
<td>50</td>
<td>17.1</td>
<td>9.71</td>
<td>0.024</td>
</tr>
<tr>
<td>Diptera Chironomidae</td>
<td>2019</td>
<td>61</td>
<td>38.4</td>
<td>7.78</td>
<td>0.010</td>
</tr>
<tr>
<td>Diptera Simulidae</td>
<td>2019</td>
<td>78.1</td>
<td>27.9</td>
<td>12.37</td>
<td>0.002</td>
</tr>
<tr>
<td>Odonata Aeshnidae</td>
<td>2019</td>
<td>50.4</td>
<td>25.8</td>
<td>7.76</td>
<td>0.007</td>
</tr>
</tbody>
</table>
Figure 3: Nonmetric Multidimensional Scaling (NMS) of fish and macroinvertebrate data by watershed. There is not a significant separation of polygons (p = 0.473). Each polygon is a site; Calcasieu tributaries are green, Red River tributaries are red, and Sabine tributaries are blue.
Figure 4: Nonmetric Multidimensional Scaling (NMS) of fish and macroinvertebrate data by year. There is less overlap of polygons ($p < 0.001$). Each polygon is a year; 2001 is yellow, 2012 is red, 2016 is purple, and 2019 is green.

Figure 5: Nonmetric Multidimensional Scaling (NMS) of macroinvertebrate data by watershed. There is not a significant separation of polygons ($p = 0.239$). Each polygon is a site; Calcasieu tributaries are green, Red River tributaries are red, and Sabine tributaries are blue.
Figure 6: Nonmetric Multidimensional Scaling (NMS) of macroinvertebrate data by year. There is a clear separation of polygons (p = <0.001). Each polygon is a year; 2001 is yellow, 2012 is red, 2016 is purple, and 2019 is green.
Figure 7: Nonmetric Multidimensional Scaling (NMS) of fish data by watershed. There is a significant separation of polygons in three-dimensional space \((p = 0.014)\). Each polygon is a watershed; Calcasieu tributaries are green, Red River tributaries are red, and Sabine tributaries are blue.

Figure 8: Nonmetric Multidimensional Scaling (NMS) of fish and macroinvertebrate data by year. There is a significant separation of polygons in three dimensional space \((p = 0.027)\). Each polygon is a year; 2001 is yellow, 2012 is red, 2016 is purple, and 2019 is green.

### 4.0 Discussion and Conclusion

The relationship between fish, macroinvertebrates and their environment depends on both spatial and temporal scales (Wiens et al. 1986; Lohr and Fausch 1997; Vinson and Hawkins 1998; Lammert and Allan 1999). Zoogeographic history, climate, and geomorphology impact regional species pools at large scales (Vinson and Hawkins 1998; Willams et al. 2002), while local species assemblages are formed by the influence of smaller scale biotic and abiotic factors on regional pools (Williams et al. 2005). Lotic systems, in particular, have a shifting mosaic of biotic and abiotic conditions formed by spatial and temporal changes (Resh et al. 1988). Factors such as rapid water volume changes, movement of substrate, and anthropogenic disturbances can cause this shifting mosaic (Resh et al. 1988). Our ecological analyses demonstrated variation in fish and macroinvertebrate assemblages between years and between watersheds on Fort Polk military base, suggesting a shifting mosaic of aquatic factors.

When examining our secondary goal of the unique taxa of the streams, some conclusions can be made. Whiskey Chitto tributary is the longest and most riverine of the study streams on Fort Polk, as such it offers the
most stable mesohabitats as it is less likely to dry up during droughts (Ward et al. 2002). This type of riverine habitat is the preference of the two significant fish indicator species for Whiskey Chitto Creek (i.e. the Calcasieu drainage basin), C. venusta and P. sciera (Table 3). The four other species which we determined to be significant indicators of the Red River drainage system using ISA, E. artesiae, L. cyanellus, L. chrysocephalus, and N. phaeus, were also supported by the species’ habitat preferences. The Red River tributaries are the smallest in length, which corresponds to L. cyanallus’ tolerance of small streams in comparison to other species. Finally, L. chrysocephalus and N. phaeus are found predominately in the Red River basin of Western Louisiana in general; N. phaeus in particular is considered endemic to the basin (Thomas et al. 2007; Douglas 1974). We determined a total of 10 macroinvertebrate families to be significant indicators of watersheds, 9 of which were indicators for the Calcasieu drainage system, and 1 Sabine drainage system (Table 4). The vast dichotomy between the quantity of indicator species for the Calcasieu and the other two basins is most likely because of the habitats present in each, as Whiskey Chitto in the Calcasieu is one of the most pristine of the stream sites, offering better and more varied habitat than the other tributaries (Dudley 2017).

With the current data, both total organism quantities and ISA, the hypothesis that the 2012 drought caused a reduction in both fish and macroinvertebrate assemblages in comparison to the other years was not entirely supported. Although 2012 had the least quantity of macroinvertebrates, the year had neither the lowest nor the highest quantity of individual fish nor indicator taxa. This may be explained by the dynamic regime of the area, with taxa adapted to frequent changes in their environment (Williams et al. 2005; 2007).

The findings from the NMS suggest an interesting contrast between fish and macroinvertebrate assemblages for year and drainage basin. For the combination of fish and macroinvertebrate data, and macroinvertebrate data separately, we found there is not much difference among the watersheds (i.e., the polygons overlap), but there is a significant difference among the years (i.e., there is limited polygon overlap). Fish assemblages, on the other hand, have significant differences among years and among watersheds (Figures 7-8). This likely related to the fact that fish in an area may be separated by many stream kilometers if they are in different tributaries, while macroinvertebrates with terrestrial or aerial adult life stages may only be separated by a few kilometers or less. This allows for greater gene flow and more uniform species distribution within the macroinvertebrate assemblages than within the fish assemblages (Anderson and Wallace 1995). Macroinvertebrates, therefore, generally respond to more local environmental conditions while fish are influenced by occurrences within drainages (Lammert and Allan 1999; Williams 2003b). This decisively supports our hypothesis that fish assemblages vary between drainages more so than macroinvertebrate assemblages. The significance of year in both assemblages illustrates that the communities are changing or fluctuating through the years.

In conclusion, the hypothesis that fish assemblages vary between drainages more so than macroinvertebrate assemblages was supported, the hypothesis that the 2012 drought caused a reduction in both fish and macroinvertebrate assemblages in comparison to the other years was rejected, and the hypothesis that assemblages exhibit temporal fluctuations in diversity around the disturbance events, but will eventually recover to a base-state was inconclusive. These ecological analyses serve to provide a more comprehensive picture of the drainage ecosystems in the region.
References


