THE FLORIDA AGRICULTURAL AND MECHANICAL UNIVERSITY

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EFFECTS OF STREAM IMPOUNDMENT AND DAM REMOVAL ON AQUATIC INSECT COMMUNITIES IN STEEPHEAD RAVINES OF THE APALACHICOLA RIVER BASIN, FLORIDA

By

AUBREY M. HEUPEL

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The members of the Committee approved the thesis entitled, “Effects of Stream Impoundment and Dam Removal on Aquatic Insect Communities in Steephead Ravines of the Apalachicola River Basin, Florida”, by Aubrey M. Heupel, defended on March 28, 2014.

Manuel L. Pescador, Ph.D.
Major Professor and Committee Chair

Andrew K. Rasmussen, Ph.D.
Committee Member

Katherine A. Milla, Ph.D.
Committee Member

Oghenekome U. Onokpise, Ph.D.
Committee Member

Michael H. Thomas, Ph.D.
Committee Member

Approved:

Lambert H. B. Kanga, Ph.D., Graduate Programs Coordinator, College of Agriculture and Food Sciences

Lee E. Anderson, Ph.D., Interim Director, Division of Agricultural Sciences, College of Agriculture and Food Sciences

Robert W. Taylor, Ph.D., Dean and Director, College of Agriculture and Food Sciences

Verian D. Thomas, Ph.D., Interim Dean, School of Graduate Studies and Research

This thesis is dedicated to

Joseph T. Collins and Matt Greene,

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ABSTRACT

The Florida Panhandle, one of five biological hotspots in the continental United States, has a high concentration of endemic flora and fauna. Much of this diversity is found within the unique drainage networks called steephead ravines that carve into the sandhills across the Florida Panhandle. Steephead ravines harbor rich aquatic insect communities, but decades of anthropogenic disturbances, including impoundments, have impacted many of North Florida's steephead habitats and communities. The impacts of stream impoundment and dam removal on aquatic insect communities in low-order streams, including steephead ravines, are unknown.

The questions addressed in this investigation were: 1) How are aquatic insect communities in Florida's steephead ravine streams affected by small man-made dams and impoundments; and 2) How much do aquatic insect communities recover five years after dam removal? To answer these questions, a comparative study was undertaken in which aquatic insects and their terrestrial adults were collected, identified, and categorized into functional feeding groups (FFG) from three steephead ravine sites within the Apalachicola River Basin: below a small impoundment, below a removed dam, and from a reference stream.

Relative abundances, taxonomic richness, and Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa measures from the impounded stream and the removed dam stream were compared to those of the reference stream to analyze structural differences. The impounded stream community was characterized by low EPT and overall taxonomic

richness and high abundances of tolerant taxa. Increased summer water temperatures, resulting from impoundment, is one of the primary factors limiting the aquatic insect community below the impoundment. The removed dam community exhibited similar EPT and overall taxonomic richness with a similar abundance distribution as that of the reference stream. Summer water temperatures were elevated but improved over those observed below the intact dam.

FFG percentages from the impounded stream and the removed dam stream were compared to those of the reference stream to analyze differences in community functionality. Outside of gathering-collectors, the impounded stream community was dominated by the filtering-collector taxa *Hydropsyche* and *Cheumatopsyche*, while FFG at the reference stream were more evenly distributed outside of gathering-collectors. Increased amounts of drifting food material from the impoundment and minimal upstream inputs of coarse woody debris due to dam presence have likely contributed to this shift in community functionality. Below the removed dam community functionality was still in the process of recovery. Overall shredder percentages were improved over what was observed below the impoundment, but there was an increased number of filtering-collectors, including *Hydropsyche* and *Chimarra*, compared to the reference stream. The reference condition of FFG cannot be restored below the removed dam until the old impoundment bed is sufficiently reforested to shade out aquatic vegetation and provide allochthonous input.

The information attained by these investigations suggests that stream impoundment has impacted the structural and functional attributes of the downstream aquatic insect community. Furthermore, five years after dam removal, aquatic insect community structure resembles that of the reference stream, but community function remains impacted by the old impoundment bed upstream.

CHAPTER ONE

INTRODUCTION AND GENERAL LITERATURE REVIEW

1.1 Steephead Ravines

The Florida Panhandle is considered one of five biological hotspots in the continental United States due to the high concentration of endemic flora and fauna found within the region (Stein et al., 2000). A large portion of this diversity can be found within the unique drainage networks known as steephead ravines that carve into the deep permeable sandhills in the Citronelle Formation across panhandle Florida. Steephead formations are unique to north Florida and were first described by Sellards and Gunter (1918). Steephead formation was originally believed to be driven by the presence of an impermeable layer in the soil profile (i.e., clay), but recent work suggests that formation is driven by variable water table levels around the tips of ravines (Abrams et al., 2009; Schumm et al., 1995). These varying water tables cause water to seep out at the surface and down the slope. Gradually the water carries sand downstream and a characteristic U-shaped valley is developed (Holt, 2008). Over time, seeps undercut the hillside as erosion occurs until part of the sandhill sloughs down, and the water continues to seep through, starting the process over again and creating a migrating valley (Means, 2000). Ultimately, ravines form natural amphitheaters with the head sloping down at approximately 45 degrees and can be up to 35 m deep.

Steephead ravines harbor rich communities of flora and fauna, composed of many species closely related to or disjunct from populations of northern lineages in the Appalachian Mountains (Delcourt & Delcourt, 1984; James, 1961; Neill, 1957; Platt & Schwartz, 1990). It is believed that during glacial advances of the Pleistocene, many organisms migrated south along river corridors like the Apalachicola-Chattahoochee-Flint river basin (Delcourt & Delcourt, 1975; Neill, 1957). During interglacial periods, northern elements found refuge on the mesic slopes of steepheads and other ravines in the Apalachicola River basin where favorable microclimates existed for these cold-adapted species. The aquatic insects found in steephead ravine streams were likely no exception. Hamilton and Morse (1990) discovered that many of the endemic Coastal Plain Trichoptera (caddisflies) have lineages closely aligned with northern species. Rogers (1933) described a similar phenomenon with the craneflies (Tipulidae) of North Florida. Furthermore, in a study exploring the diversity of Trichoptera and Plecoptera (stoneflies) species within steephead ravines, Rasmussen (2004) observed disjunct populations of several species typically found farther north. He also discovered at least 12 new Trichoptera species, indicating how little is known about these unique communities. Additional research within steephead ravine communities can help develop a better understanding of these newly discovered endemics and disjunct populations.

1.2 Stream Impoundment

America has a long history (over 200 years) of altering its river systems through dam construction, with a large portion of the production occurring between 1950 and 1970 (AASHTO, 2005). According to the Army Corps of Engineers database, there are over 76,000 dams greater than 2m high within the United States (Heinz Center, 2002). However, small dams are not monitored by the Corps and it is believed there are actually well over 2.5 million dams throughout the nation (Johnston Associates, 1989). Today, more than 600,000 miles of our nation’s waterways are impounded behind dams (Feldman, 2010).

An impoundment is the resulting water body or reservoir created by the damming of a waterway. Impounded waters of all sizes can be of economic importance and their uses may include municipal drinking water, industry, agricultural and municipal irrigation, flood control, and recreational activities such as fishing and swimming. When dam construction, maintenance, or removal is considered, these economic needs frequently take precedence over long-term, often irreversible, ecological impacts such as migration obstruction (Li et al., 2012; Penczak et al., 2012; Ziewitz, 2005), habitat fragmentation and population isolation (Watters, 1996; Zhao et al., 2012), altered hydrological regimes (Costigan & Daniels, 2012; Zhao et al., 2012), loss of biodiversity (Li et al., 2013; Ogbeibu & Oribhabor, 2002), changes in water chemistry and temperature (Humborg et al., 1997), and changes in the vegetative communities (Benjankar et al., 2012; Su et al., 2013). The literature available on the impacts of large dams is extensive (e.g., Gray & Ward, 1982; Humborg et al., 1997; Inverarity et al., 1983; Li et al., 2013; Penczak et al., 2012; Skalak et al., 2013; Zhao et al., 2012); however, the ecological impacts of small dams are largely understudied (Hart et al., 2002). All dams disrupt the natural flow and connectivity of a river system, and it is important to understand the extent and trends of disturbance caused by these smaller structures, especially considering the number of small dams in existence today.

1.3 Dam Removal

In the past few decades, environmental and societal interests have turned towards undamming our nation's rivers (AASHTO, 2005; Bednarek, 2001; Hart et al., 2002; Heinz Center, 2002). This push has largely come about as a result of the nation's large number of aging, failing, and unused dams and their continued impact on our river systems. For instance, many small dams, such as mill dams, no longer serve a purpose, so as they age and deteriorate, dam owners are deciding to have them removed instead of repaired. Over the last several decades, the dam removal decision has been made by many land-owners across the country, and more than 600 dams have been removed (AASHTO, 2005). This option is supported economically (AASHTO, 2005; American Rivers et al., 1999; Born et al., 1998), but its usefulness in ecologically restoring our free-flowing river systems is undetermined (Hart et al., 2002). Only a small fraction of dam removals have been accompanied by scientific study and, for the most part, studies have provided only short-term post-removal information. Dam removal can be just as ecologically damaging as the dam itself without an adequate knowledge of the best removal strategies for ecosystem recovery, and there have been recent concerns over this knowledge gap and the need to close it as the ecological benefits of dam removal remain uncertain (AASHTO 2005, American Rivers et al., 1999; Bednarek, 2001; Hart et al., 2002; Heinz Center, 2002).

In 2002, Hart et al. provided a comprehensive summary of the dam removal studies available and their findings. Although there is more information available on the ecological impacts of large dams, the cost of removal and the economic need for these structures often prohibits their removal. Thus, the majority of dam removal projects involve dams less than 7 meters high, which are often privately owned (AASHTO, 2005; Heinz Center, 2002). Some of the beneficial findings of these small dam removals have included improvements in sediment transport and decreased water temperatures (e.g., Born et al., 1998; Bushaw-Newton et al., 2002). The biotic impacts of small dam removal, however, have been more varied. For instance, Born et al. (1998) reported improved fish passage and trout spawning after the removal of a small dam in Wisconsin, but Sethi et al. (2004) found that mussel abundances and diversity decreased downstream after the removal of small dams in Minnesota and Wisconsin. The few existing studies on small dam removal have had variable results, but more long-term recovery information is necessary before conclusions should be drawn regarding the effects of dam removal on aquatic ecosystems.

1.4 Aquatic Insects and Stream Impoundment

Stream impoundments have been shown to impact many aquatic-linked taxa, including fishes (Li et al., 2013; Penczak et al., 2012; Yi et al., 2010), mussels (Dean et al., 2002; Gangloff et al., 2011; Singer & Gangloff, 2011;Watters, 1996), herpetofauna (Hunt et al., 2013), birds (Graf et al., 2002; Pandey, 1993), mammals, and plants (Benjankar et al., 2012; Su et al., 2013). Over the last several decades, many studies have also examined the impacts of impoundments on aquatic insects (Inverarity et al., 1983; Katano et al., 2009; Munn & Brusven, 1991; Ogbeibu & Oribhabor, 2002; Spence & Hynes, 1971; Valentin et al., 1995; Ziser, 1985). The aquatic insect communities below stream impoundments have been shown to significantly differ in composition from those in undammed streams or above-dam reaches (Inverarity et al., 1983; Katano et al., 2009; Lessard & Hayes, 2003; Ogbeibu & Oribhabor, 2002). In addition, abundances and densities of aquatic organisms are often higher below impoundments (Katano et al., 2009; Zhang et al., 2010; Ziser, 1985), while diversity has been found to be significantly lower in some cases (Lessard & Hayes, 2003; Ogbeibu & Oribhabor, 2002; Ziser, 1985). Increases in water temperature and drifting food resources are some of the factors contributing to community changes (Katano et al., 2009; Lessard & Hayes, 2003). These trends have been observed in many high-order (orders 4-5) streams with large dams (>7 m), but few studies have focused on small-order streams (orders 2-3) with small dams like those found in this study. Hart et al. (2002) suggested that some insight could be gained from knowledge of natural analogs of small dams, such as beaver dams, landslides, or waterfalls, but expressed the need for further research on small dam impacts. It is essential to understand how aquatic insect communities are affected by small dams since these communities play such a vital role in the food web and overall stream ecosystem function. In addition, small-order streams are an important source of stream system biodiversity (Meyer et al., 2007) and greater efforts are needed to understand and minimize disturbance impacts**.**

1.5 Ephemeroptera, Plecoptera, and Trichoptera and Functional Feeding Groups

Aquatic insects can be particularly susceptible to anthropogenic disturbances and are widely used as measures of stream ecosystem health (Karr, 1999). In particular, sensitivity to changes in water temperature, chemistry, and flow (Berner & Pescador, 1988; Lessard & Hayes, 2003; Punzo & Thompson, 1990; Zivic et al., 2013) make aquatic insect communities useful candidates for monitoring our impacted waterways for potentially harmful fluctuations. In addition to basic structural measures of community such as taxonomic richness (number of taxa), relative abundance, and diversity, many ecologists also use richness of Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies), often referred to as EPT, and functional feeding groups (FFG) when studying aquatic insect communities (Anderson, 1992; Katano et al., 2009; Poepperl, 1999).

EPT richness is a variation of taxonomic richness frequently used in aquatic insect and stream assessment studies (Rosenberg et al., 2008). The concept of measuring only EPT taxa was originally based on the thought that these orders are the least tolerant groups to pollution and/or disturbance (Resh & Jackson, 1993). This is not always the case as some taxa are better generalists, thriving in a variety of habitats with varying degrees of disturbance. For instance, increased densities after disturbance are often attributed to increased abundances of chironomids (midges) (Brown et al., 1997; Nislow & Lowe, 2006), but abundances of some Ephemeroptera taxa (e.g., Baetidae) have also been known to increase following stream disturbance (Carlson et al., 1990; Gurtz & Wallace, 1984; Noel et al., 1986). That being said, most EPT taxa are still negatively affected by pollution and disturbance (Compin & Cereghino, 2003; Ort et al., 1995; Woodcock & Huryn, 2005). Together, these orders often make up a large portion of the macroinvertebrate communities of stream ecosystems and are important in terms of ecosystem functioning (Berner & Pescador, 1988; Stewart & Stark, 2008; Wallace et al., 1982).

In addition to structural metrics, many studies of stream health have also used FFG to examine differences or changes in the functioning of an ecosystem following disturbance (Fuchs et al., 2003; Kedzierski & Smock, 2001; Liljaniemi et al., 2002; Miserendino & Masi, 2010; Nislow & Lowe, 2006; Quinn et al., 2004; Whiles & Wallace, 1997). FFG are categories used to differentiate aquatic insects based on morpho-behavioral mechanisms used for food acquisition (Rosenberg et al., 2008). This method separates organisms based on their methods of gathering food rather than the type of food they eat (e.g., detritus). Cummins et al. (2008) defines five main FFG categories:

shredders, piercers, scrapers, predators, and collectors, which are frequently broken down into two subcategories, filtering and gathering. This method was originally developed, in part, to aid in classification when taxonomic identifications were unavailable (Cummins, 1973). Today, using FFG can aid in explaining why structural changes in a community exist. For instance, Nislow and Lowe (2006) found higher overall abundances with greater amounts of disturbance. By using FFG, they were able to observe a functional shift from shredder-dominated communities (species that shred and chew decomposing plant material) in less disturbed sites, towards high scraper-dominated communities (species that feed on materials like algae that are attached to underwater surfaces) in more recently disturbed sites. In using structural and functional metrics, Nislow and Lowe (2006) were able to explain that the observed increases in overall density at recently disturbed sites were due to increases in scrapers, which is often an indication of a switch from allochthonous inputs (i.e., leaf litter) typically used by shredders, to autochthonous inputs (i.e., periphyton and algae), often used by scrapers. An overall shift from allochthonous to autochthonous inputs after disturbance (e.g., logging) has been reported by many (e.g., Gurtz & Wallace, 1984; Reid et al., 2010; Stone & Wallace, 1998).

1.6 Aquatic Insects and Dam Removal

Only a few studies (Pollard & Reed, 2004; Stanley et al., 2002; Thomson et al., 2005) have examined the ecological impacts of dam removal on aquatic insect communities. Pollard and Reed (2004) and Thomson et al. (2005) reported that one year after dam removal, macroinvertebrate assemblages were more similar at impacted downstream sites and at upstream reference sites than they had been prior to removal. Thomson et al. (2005) reported significantly lower macroinvertebrate densities after dam removal possibly due to increased sedimentation. Both studies provide insight into aquatic insect community recovery after dam removal, but both were conducted on high order ($\geq 4^{th}$ order) streams. Little is known about how aquatic insect communities respond to dam removal in low-order streams and how quickly they can recover.

1.7 Purpose of Study

Stream impoundments affect rivers and streams of all shapes and sizes across the country, and the steephead ravine streams of North Florida are no exception. The impacts of stream impoundment have been studied for decades. However, small dams with small impoundments on low-order streams, like Florida's steephead streams, are understudied, and it is largely unknown how the aquatic communities, such as aquatic insects, are affected by these structures. Furthermore, few studies have examined aquatic insect community recovery following small dam removal on low-order streams and have mostly provided only short-term or immediate community recovery data.

Given the lack of knowledge regarding aquatic insect communities and small dams on low-order streams, small dam removal, and steephead ravines, this study aims to answer the following questions:

1. How are aquatic insect community structure and function of a steephead ravine stream affected by the presence of a small dam and impoundment?
2. To what extent has the aquatic insect community structure and function recovered five years after the removal of a small dam on a low-order steephead ravine stream, when compared to the aquatic insect community of an undisturbed steephead ravine stream?

3. To what extent have the Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies; EPT) recovered, structurally and functionally, five years after the removal of a small dam on a low-order steephead ravine stream, when compared to pre-removal and post-removal data?

CHAPTER TWO

EFFECTS OF STREAM IMPOUNDMENT ON THE AQUATIC INSECT COMMUNITY OF A STEEPHEAD RAVINE WITHIN THE APALACHICOLA RIVER BASIN, FLORIDA

2.1 Introduction

Steephead ravine streams have received little attention since the formations were first described in 1918 (Sellards & Gunter), and surprisingly little has been done to address how land use patterns affect the faunal communities found within these unique habitats. Like many other natural communities, steephead communities are faced with a variety of anthropogenic disturbances, including stream impoundment, but the ecological impacts of such disturbances on steephead ravine faunal communities are unknown. Furthermore, the aquatic insect communities found in these stream systems contain a variety of cold-adapted, northern relics and narrow-range endemics (Rasmussen, 2004; Rogers, 1933), which may be particularly sensitive to habitat disturbances like stream impoundments that alter flow and thermal regimes. In order to protect and properly manage for these species, it is important that we understand how long-term disturbance by stream impoundment has altered steephead ravine stream communities.

The goals of this study were to investigate the aquatic insect community of an impounded steephead ravine stream and determine: 1) the effects of impoundment on aquatic insect community structure, and 2) the effects of impoundment on aquatic insect community function. A comparative study was undertaken to compare the aquatic insect community of an impounded steephead stream with that of an intact steephead stream that served as the reference model. Aquatic insects and their terrestrial adults were collected below the impoundment on a steephead ravine stream and at the undisturbed reference stream. Relative abundances, taxonomic richness, and EPT taxa measures were determined and compared between the two sites to identify structural differences between the communities. Aquatic insects were also categorized into functional feeding groups (FFG) and percentages were calculated and compared to identify differences in functionality between the two communities. The null hypothesis for the investigation was that the aquatic insect community below the impoundment would not differ structurally nor functionally from the reference stream. More specifically, the two communities would not differ in relative abundances, EPT and total taxonomic richness, nor in their respective FFG percentages.

2.1.1 Study Area

The two steephead ravine streams selected for study lie on the eastern side of the Apalachicola River in the central Florida panhandle (Fig. 1). The first site, referred to as Spring Canyon, is located on private property in Gadsden County, Florida, and is part of the Crooked Creek watershed (Fig. 2). The property contains a third-order stream, based on the Strahler stream order system (Strahler, 1957). The stream has been impounded by a small earthen dam since the mid-1900's, and the 9 acre impoundment has been primarily used for recreational purposes (Fig. 3). The surrounding uplands are being restored to native longleaf pine-wiregrass. Native beech-magnolia dominates the ravine canopy and Florida anise (*Illicium floridanum*) and remnant mountain laurel (*Kalmia latifolia*) are dominant understory species in the ravines above and below the impoundment. The southern and northeastern sides of the impoundment display drastic elevation change with native vegetation still intact, while the northwestern side has been disturbed by vegetative clearing.

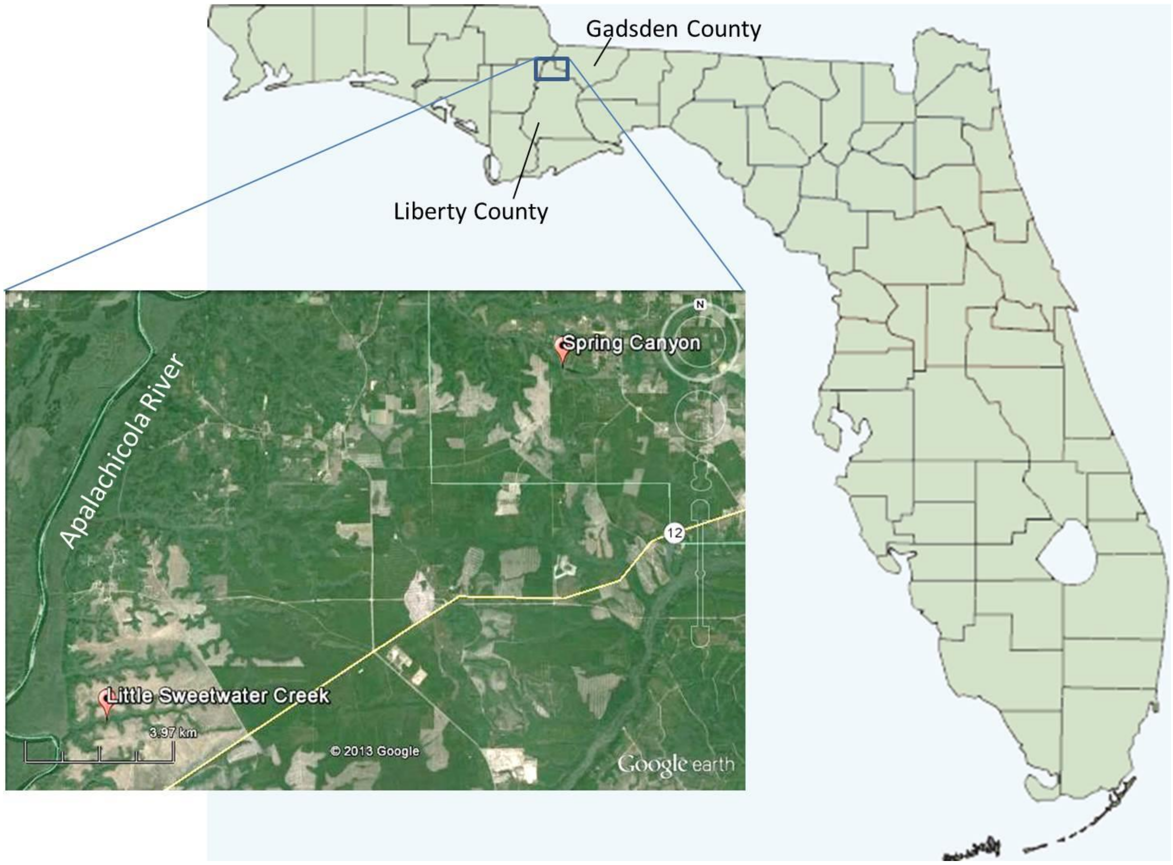

Figure 1. The steephead ravine systems studied in Gadsden County (Spring Canyon) and Liberty County (Little Sweetwater Creek), Florida. Aerial image courtesy of Google Earth.

The second study site, Little Sweetwater Creek, flows directly into the Apalachicola River and is located in Liberty County on the Apalachicola Bluffs and Ravines Preserve, managed by The Nature Conservancy (Fig. 4). Little Sweetwater Creek was selected as the reference stream because it is an intact, steephead ravine stream. The ravine slopes are largely intact with native vegetation. The uplands were cleared before the land was purchased by TNC in the 1980's, but the agency has successfully restored much of the surrounding upland habitat over the past 30 years. Although the lands surrounding the steephead ravines in the aerial image in Figure 1 appear to be clear-cut, these lands are forested but at a lower basal area compared to the pine plantations surrounding the preserve. Since the land was purchased by TNC there has been little to no human activity in or around the reference stream system outside of periodic prescribed burning conducted by The Nature Conservancy.

Figure 2. The Spring Canyon property in Gadsden County, FL. The studied steephead stream is part of the Crooked Creek watershed in Gadsden County, FL. Image courtesy of Google Earth.

Figure 3. A) The earthen dam and culvert at Spring Canyon and B) the resulting impoundment, located on private property in Gadsden County, FL. Images taken by Aubrey M. Heupel.

Figure 4. The Nature Conservancy's Apalachicola Bluffs and Ravines Preserve in Liberty County, FL. Image courtesy of Google Earth.

A reference stream was used instead of an upstream/downstream approach for several reasons. First of all, the accessible portion of the stream above the impoundment was still clearly impacted by the presence of the impoundment. Although there was an obvious stream channel, there was a lot of surrounding marsh-like habitat with several smaller stream channels cutting through herbaceous vegetation that is not typical of steephead ravine habitats. Secondly, stream size upstream would have been an issue in terms of habitat and community comparisons with the site below the dam.

2.2 Materials and Methods

In the spring of 2012, one 50-meter aquatic reach and one light-trapping station were established at both stream systems. The aquatic reaches were measured using a 50-m tape, starting downstream (00 m) and running upstream (50 m) (Fig. 5). The Spring Canyon aquatic sampling reach was located approximately 25 m below the dam (SCB; Fig. 6) with the light-trapping station located along the bank at 25 m (N 30°33'40.3", W 084°50'43.4"). The Little Sweetwater Creek aquatic sampling reach (LSC; Fig. 7) was established in the lower reaches of the system to obtain a reach similar in size and order to the Spring Canyon study site. The corresponding light-trapping station was located along the bank at 25 m (N 30°28'34.1", W 084°58'22.8").

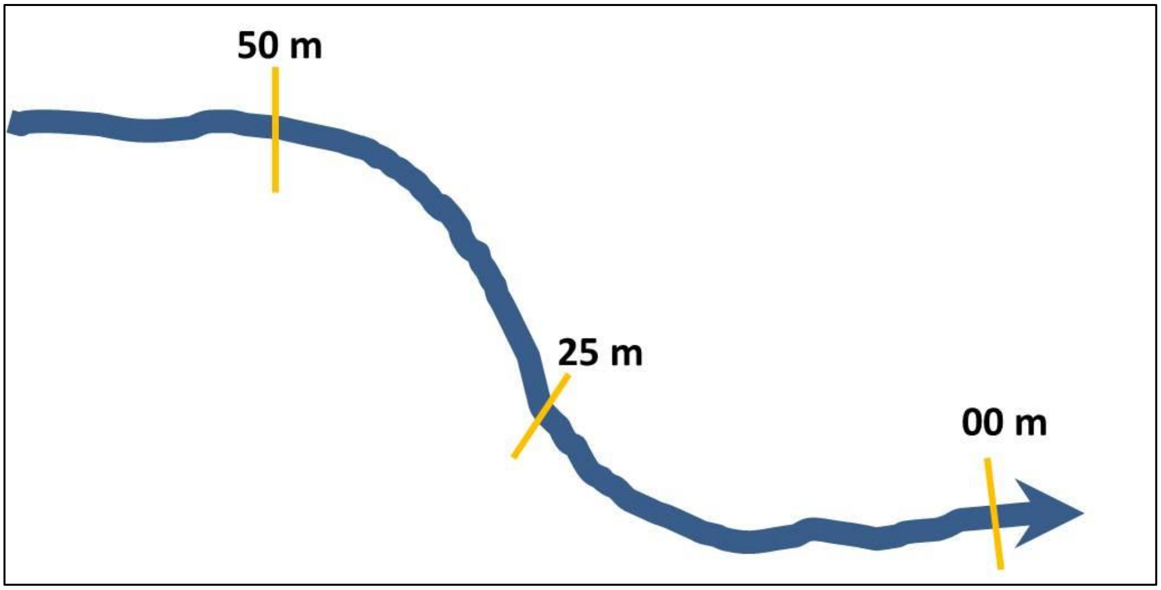

Figure 5. Aquatic reach diagram showing placement of 00 m and 50 m points in relation to stream flow.

Figure 6. The Spring Canyon impoundment and the aquatic sampling site below the dam (SCB) in Gadsden County, FL. The 00 m (downstream) coordinates for the aquatic sampling reach are as follows: N 30°33’40.6”, W 084°50’43.6”. Image courtesy of Google Earth.

Figure 7. Little Sweetwater Creek aquatic sampling site (LSC) on Apalachicola Bluffs and Ravines Preserve in Liberty Co., FL. The 00 m (downstream) coordinates for the aquatic sampling reach are as follows: N 30°28’33.5”, W 084°58’23.0”. Image courtesy of Google Earth.

2.2.1 Abiotic and Habitat Parameters

Several abiotic parameters were measured over the course of the study. Air temperature, wind speed, and relative humidity were collected during each site visit using a Kestrel Pocket Wind Meter (Nielsen-Kellerman Model 3000), and general weather condition was recorded using a weather code (1=clear [<5% cloud cover], 2=partly cloudy [5-90%], 3=cloudy [>90%], 4=rain, 5=other). Water temperature was collected continuously on a one hour interval by Onset Tidbit v2 Data Loggers (accuracy: 0.2°C over 0° to 50°C), attached to steel rebar near the midway point of each aquatic sampling reach. Data were uploaded from the loggers each month, excluding December 2012 and April 2013 when schedules prevented data collection. Conductivity, pH, and dissolved oxygen (DO) were measured using digital sensors (YSI-556 MPS; Yellow Springs

Instruments, Yellow Springs, Ohio). These parameters were collected seasonally from the midway point prior to aquatic sampling to avoid inaccurate readings from upstream disturbance. Flow velocity was estimated seasonally by measuring the length of time, in seconds, it would take for a piece of floating debris (e.g., a leaf) to travel downstream one meter. This is the method employed by the Florida Department of Environmental Protection (FDEP) when conducting in-stream habitat assessments (FDEP, 2012). Stream width was measured quarterly at 0 m, 25 m, and 50 m. A convex spherical densiometer (Robert E. Lemmon, Forest Densiometers Model-A) was used to measure canopy cover monthly (excluding December 2012 and April 2013) at 0 m, 25 m, and 50 m (Fig. 5). Water depths were measured monthly (excluding December 2012 and April 2013) at 0 m, 25 m, and 50 m, with three measurements taken within the stream channel (left, center, and right).

Habitat assessments were conducted at both sites by FDEP personnel in June 2012 and January 2013 (FDEP, 2012). These were done to provide standardized characterization and ranking of the primary (in-stream) and secondary habitat parameters. Streams were given a ranking of optimal (120-160), suboptimal (80-119), marginal (40-79), or poor (11-39), based on eight habitat component scores. Primary habitat scores were for substrate diversity, substrate availability, water velocity, and habitat smothering. Secondary habitat components were artificial channelization, bank stability, riparian buffer zone width, and riparian zone vegetation quality.

2.2.2 Aquatic Insect Sampling

Aquatic insects were sampled seasonally from May 2012 – February 2013. Spring Canyon samples were collected 18 May, 17 August, 16 November, and 15 February, and

Little Sweetwater Creek samples were collected 19 May, 18 August, 17 November, and 16 February. Qualitative data were collected using a D-frame dip-net (0.3 m wide; 1000 µm mesh; 1.5 m handle; Fig. 8). One 0.5 m sweep was collected for each major habitat type within each 50 m reach. The habitat categories considered were root, snag, sand, and leaf pack. Root habitat is classified as woody debris less than thumb diameter and snag habitat is woody debris larger than thumb diameter (FDEP, 2012). Leaf packs are areas where leaf debris has gathered within the stream channel. Many sampling protocols, including that used by the FDEP, also consider aquatic plants as a possible major habitat; however, the study reaches contained little to no aquatic plants throughout the sampling period. A habitat type was considered major when the amount of habitat available was greater than the required 0.5 m sweep area. Because the majority of the substrate in the streams was sand, a random number generator (range of 0-50) was used to determine one location from which to collect the sand sample within each reach during each sampling session. Samples from each habitat type were placed in separate collecting jars with 80% ethyl alcohol and brought back to the laboratory for sorting and identification.

Figure 8. A D-frame dip-net, used to qualitatively sample benthic communities in a variety of habitats. Image taken by Aubrey M. Heupel.

2.2.3 Light-Trap Sampling

Light-trapping was conducted to collect the terrestrial adult stages of Ephemeroptera, Plecoptera, and Trichoptera (EPT) in order to obtain species-level data. EPT taxa were targeted because they are widely used as indicator taxa in bioassessment protocols (Environmental Protection Agency, 1999; FDEP, 2011; North Carolina Department of Environment and Natural Resources, 2011). A large percentage of these taxa cannot be identified to species in the larval stages and some larva can be difficult to collect, so light-trapping was used to supplement aquatic sampling efforts. Trapping was conducted seasonally from May 2012 – April 2013. Spring Canyon samples were collected 1 May, 7 August, 3 November, and 1 April, and Little Sweetwater samples were collected 2 May, 8 August, 2 November, and 19 March. Traps were set near the water's edge around dusk and were deployed for 1-3.5 hours after sunset. Traps consisted of a 15-watt UV-blacklight (BioQuip 2805 DC Light) placed over a white collecting pan (Photoquip HDPE Tray, 810T) containing 80% ethyl alcohol (Fig. 9). The lights were powered by 12-volt rechargeable batteries (Power Patrol SEC 1075 Battery). After trapping, the contents of the pan were poured into a half gallon plastic container and brought back to the laboratory for sorting and identification.

Figure 9. A light-trap set for the collection of the terrestrial adult stages of aquatic insects below the dam at Spring Canyon on 1 April 2013. Image taken by Aubrey M. Heupel.

2.2.4 Specimen Identification

The May and August aquatic samples were sorted completely, with no additional sample preparation or subsampling, using a Leica S6E stereomicroscope (6.3-40X) and fiber optic light source (Techniquip FOI-150). Due to time constraints, the November and February samples were prepared and representative subsamples were analyzed. The subsample preparation process used was adapted from that utilized by the FDEP for Stream Condition Index (SCI) samples (FDEP, 2011). To prepare the samples, the alcohol was first drained from the sample using a U.S.A. Standard Testing Sieve #35 (500 μm mesh). The sample was then placed into a bucket of water to achieve homogenization and poured through a U.S.A. Standard Testing Sieve #10 (2 mm mesh) on top of a #35 sieve to separate the larger floating debris such as leaves and twigs. The larger materials and any sand left in the bucket were placed into white collection trays (Photoquip HDPE Tray, 810T) and examined under a magnifying glass to remove any remaining organisms. Any organisms found were added to the material in the #35 sieve. The material was thoroughly rinsed to remove any materials smaller than the 500 µm mesh. The material was then transferred to a white collection tray that was gridded into 5x5 cm sections. The sample material was spread evenly into a number of sections divisible by four, and half of the sections were randomly selected as subsamples for sorting. Most samples could be evenly distributed into four sections; however, some large samples required eight sections. One of the samples was too small to evenly cover four sections. Instead, it was placed into 2 sections which were subdivided into 8 so four subsamples could be selected to better randomize sample selection. Subsamples were placed into separate vials of alcohol for sorting. The leftover material was combined and stored in alcohol for future studies.

Aquatic insects were sorted out of the subsamples and identified using a stereomicroscope and fiber optic light source. For all aquatic samples, Dipterans were identified to family, EPT were identified to species when possible, and all others were identified to genus. Due to large sample sizes of *Oecetis* at SCB, samples of this genus were examined to determine an approximate number of represented species for taxonomic richness totals but counts were left at genus for relative abundance. Aquatic insects were identified using Merritt et al. (2008) and a number of taxonomic keys developed for Florida (Epler, 2006; Epler, 2010; Pescador et al., 2000; Pescador et al., 2004; Pescador & Richard, 2004; Rasmussen & Pescador, 2002; Richardson, 2003; Richardson, 2010). Specimens housed at FAMU also served as reference material.

Specimen identifications were verified by Dr. Andrew Rasmussen, Dr. Manuel Pescador, or Barton Richard. When possible, specimens were also categorized into one of six FFG: gathering-collectors (GC), filtering-collectors (FC), predators (PR), scrapers (SC), shredders (SH), or vegetative piercers (PC) using FFG classifications given by Cummins et al. (2008) and Environmental Protection Agency (1999).

Light-trapped samples required pre-sorting to remove any non-target taxa such as beetles, moths, midges, etc., from the desired EPT taxa. Pre-sorting and EPT identification were accomplished using a stereomicroscope and fiber optic light source. Individuals were identified to the lowest taxonomic level possible using an extensive amount of taxonomic literature as well as the insect collections available at FAMU. In many cases, only the adult male of a species has been described, so many of the collected females could only be identified to genus. Most samples were small enough that all individuals were identified and tallied. However, the microcaddisfly (Hydroptilidae) males from the November 2012 samples were subsampled due to large sample size. The remainder of the sample was tallied and left at the family level. Similarly, in the case of the Spring Canyon May 2012 sample, the hydropsychid females were counted but not identified past family due to a large sample size and time constraints. Dr. Steven Harris identified all adult microcaddisfly samples, and Dr. Manuel Pescador identified all adult mayflies. All other specimen identifications were verified by Dr. Andrew Rasmussen. When possible, the adult EPT specimens were also categorized into one of six FFG (GC, FC, PR, SC, SH, or PC) using FFG classifications provided by Cummins et al. (2008) and Environmental Protection Agency (1999).

2.2.5 Statistical Analysis

Stream depth (*n*=33), stream width (*n*=12), and percent canopy cover (*n*=33) means were calculated and used for statistical comparisons between SCB and LSC using 2-sample t-tests with α=0.05. Hourly water temperature values were averaged for each day and average daily temperatures (*n*=365) were subjected to a 2-sample t-test to test for differences between the two sites. Average daily temperatures were also compared between the two sites for summer (May 2012 – October 2012) and winter (November 2012 – April 2013) months. Velocity, pH, dissolved oxygen, and conductivity values (*n*=4 for each) were also compared between the sites using 2-sample t-tests with α=0.05. All statistical tests were conducted using MINITAB 16 (Minitab, 2013).

Taxon databases were developed separately for dip-net and light-trap collections in Microsoft Access to query and compile basic descriptive statistics, including relative abundances, unique and common taxa, taxonomic richness, EPT taxa, and FFG percentages for each site and each trapping technique. Early instar specimens that could not be identified to the target taxonomic rank of family or genus and were not unique were omitted from relative abundance calculations. For example, if a specimen of Elmidae could not be identified to genus and one or more elmid genera were already identified, then the unidentified elmid was omitted from the calculations since it did not represent a unique taxon. SCB and LSC dip-net taxonomic richness values were statistically compared by date and for each major habitat type, and SCB and LSC light-trap taxonomic richness values were compared. EPT richness values by date and by habitat type were compared between SCB and LSC dip-net samples. An arcsin data transformation (arcsin[$\sqrt{y}$]) was used to transform EPT and FFG percentages before conducting statistical comparisons for dip-net and light-trap data between SCB and LSC. Light-trap and dip-net FFG were compared separately. All statistical comparisons were done using 2-sample t-tests with α=0.05 in MINITAB 16 (Minitab, 2013).

2.3 Results and Discussion

2.3.1 Physicochemical Parameters

The physicochemical characteristics of both streams are summarized in Table 1. Average stream width at LSC was significantly wider than that at SCB (t_{11}=6.38, $p<0.001$). However, SCB naturally may have been a wider stream, more similar to LSC, but channelization due to dam presence has since altered the stream bed. Statistically, canopy cover was significantly different between SCB and LSC (t_{36}=2.37, p=0.02), but biologically, the difference was negligible as both streams had over 95% canopy cover. Steephead ravine streams are typically heavily forested with a closed canopy, but occasional gaps in the canopy are natural occurrences and part of succession in any forest habitat. In this case, it is the lack of canopy cover over the impoundment that has a greater impact on the stream below the dam. The highest hourly water temperatures recorded throughout the study were 33.2°C and 25.7°C for Spring Canyon and Little Sweetwater Creek respectively, and the minimum hourly temperatures recorded were 10.2°C and 8.9°C for SCB and LSC respectively. Average daily water temperatures over the course of the year were significantly higher (n=365; t_{598}= -6.85, $p<0.001$) below the dam at Spring Canyon than at the reference stream. As illustrated in Figure 10, temperatures were substantially higher at SCB compared to LSC during the summer months and were, on average, nearly 5°C higher below the dam (Table 1) as compared to LSC summer mean temperatures. While average daily water temperatures in the summer period were significantly different (n=182; t_{322}= -20.73, $p<0.001$) between SCB and LSC, there was no significant difference in mean water temperatures in the winter (n=183; t_{351}= -0.46, p=0.65).

Warmer waters were expected below the impoundment because the water within the impoundment warms as it collects more solar radiation and the warmer waters then flow downstream via the top-flow standpipe. Since the mid-point of the study reach was less than 50 m downstream of the culvert, the water had minimal time to change via ambient cooling. In contrast, the reference stream is less impacted by direct solar radiation due to a closed canopy. During the cooler winter months, average daily temperatures were similar between the sites, but fluctuation below the dam was less extreme (Fig. 10). This is likely due to the impoundment, as a large body of water, serving as a buffer against wide fluctuations in temperature over short periods of time.

Aquatic insects can be very susceptible to changes in water temperature, especially cold-adapted, northern relics such as those found in the steephead ravine streams of North Florida. Increases in water temperature can be especially dangerous, as the maximum thermal limit for most freshwater aquatic insects is between 30°C and 40°C (Pennak, 1978). The maximum recorded temperature below the dam falls within this temperature limit, so loss of relics and endemics, and an overall decline in diversity would be expected. Compounding this stress is the fact that water temperature and dissolved oxygen (DO) solubility are inversely related; therefore, as water temperatures rise DO levels typically decrease (Merritt et al., 2008). However, the mean DO levels were not significantly different between SCB and LSC (t_5=1.15, p=0.30;

Table 1. Summary of the physical and chemical properties of the impounded study stream, Spring Canyon, and the reference stream, Little Sweetwater Creek, from May 2012 – May 2013 (*n*= number of samples).

Properties	Spring Canyon (Impounded)				Little Sweetwater Creek (Reference)				*t*-test
	n	Mean ±SE	Min.	Max.	*n*	Mean ±SE	Min.	Max.	p-value
Physical									
Stream Width (m)	12	2.4 ±0.04	2.2	2.6	12	3.9 ±0.2	3.0	5.1	<0.001*
Water Depth (m)	33	0.19 ±0.01	0.06	0.31	33	0.18 ±0.01	0.12	0.31	0.30
Canopy Cover (%)	33	98.2 ±0.6	87.5	100	33	99.6 ±0.1	96.9	100	0.02*
Avg. Daily Water Temp.: May ’12 – May ‘13 (°C)	365	21.6 ±0.3	11.2	30.7	365	19.1 ±0.2	10.7	24.3	<0.01*
Avg. Daily Water Temp.: May ’12 – Oct ‘12 (°C)	182	26.9 ±0.2	16.2	30.7	182	22.0 ±0.1	13.8	24.2	<0.01*
Avg. Daily Water Temp.: Nov ’12 – May ‘13 (°C)	183	16.3 ±0.2	11.2	23.8	183	16.2 ±0.2	10.7	21.1	0.65
Current Velocity (m/s)	4	0.3 ±0.03	0.2	0.3	4	0.3 ±0	0.3	0.3	0.13
Chemical									
pH	4	5.1 ±0.3	4.5	5.7	4	3.7 ±0.4	2.8	4.9	0.05*
Dissolved Oxygen (mg/L)	4	9.0 ±0.9	7.4	11.2	4	10.3 ±0.8	8.8	12.0	0.30
Conductivity (umhos/cm)	4	12.3 ±0.6	11.0	14.0	4	14.5 ±0.3	14.0	15.0	0.03*

The statistical values from the 2-sample t-tests are indicated. *Indicates significant difference for α=0.05.

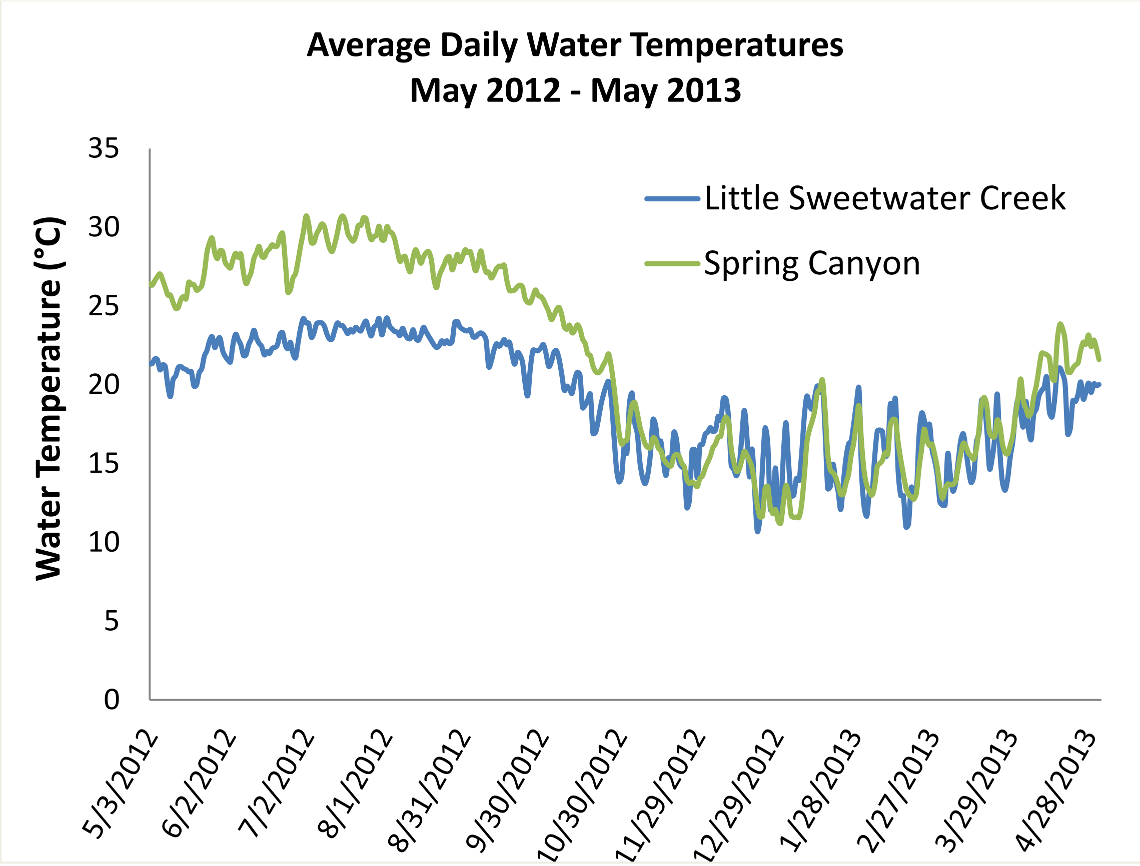

Figure 10. Average daily water temperatures below the dam of the impounded stream, Spring Canyon, and at the reference stream, Little Sweetwater Creek, from May 2012 – May 2013, in Gadsden and Liberty Counties, FL.

Table 1), and were near saturation levels, sometimes. The high DO levels at SCB are possibly due to an aeration effect caused by water cascading through the standpipe and discharging in a perched position downstream of the dam, whereas the high DO levels at LSC are influenced by natural turbulence achieved as the stream flows over natural obstructions.

Average pH values were significantly different between SCB and LSC (t_4= -2.76, p=0.05), with LSC being considerably more acidic. The sandy-bottomed streams of the Panhandle are naturally acidic and the hills through which steephead ravines form contain especially sandy, nutrient-poor, and acidic soils (Barbour et al., 1996; Florida Fish and Wildlife Conservation Commission, 2005; Rasmussen, 2004). The higher pH at SCB may be a result of the impoundment. Previous studies have proposed that increased pH could be a secondary result of high autochthonous production within impoundments (Neel, 1963; Ziser, 1985).

Conductivity levels were low at both sites. The mean conductivity levels were slightly higher in LSC than in SCB; however, the slight difference (2.2 umhos/cm) is probably not biologically meaningful. Average stream depth and current velocity were not significantly different ($p>0.05$) between SCB and LSC (Table 1). The chemical data collected provides some understanding of the properties of SCB and LSC, but further data collection of the chemical properties of both streams is necessary before reliable comparisons can be made regarding these parameters.

2.3.2 In-stream Habitat

Snags and sand were major in-stream habitat components below the dam and at the reference site throughout the study period. However, there were fewer large “open” patches of sand below the dam in February due to obvious silt accumulation. Roots were also a major in-stream habitat component below the dam during all four sampling periods, and, for the reference stream, roots were a major habitat component except during the August sampling session. SCB had notably higher quantities of exposed root habitat which is likely a result of the scouring caused by the dam outfall. Water naturally picks up energy as it falls down the standpipe and flows downstream, which over time can cause scouring of the natural stream channel and increase channelization downstream. Leaf pack habitat was more abundant and consistently available at LSC.

Leaf packs were only a major habitat component at SCB during the November sampling session when leaf fall was near its peak in the surrounding riparian area. In general, there is less leaf litter and other organic matter input at SCB because there is minimal stream length before and within the study stretch where input can occur to create leaf pack habitat.

For the habitat assessments (HA) conducted by the FDEP, SCB had an overall optimal ranking with a score of 121/160 in June 2012, but received an overall suboptimal ranking with a score of 119/160 for the January 2013 HA. The reference stream had an overall optimal score of 141/160 for both HA. Based on the overall HA scores, SCB is in boarder-line optimal condition in terms of overall stream health, but when the scores are broken down into primary and secondary habitat parameters, it is clear that the in-stream habitat at SCB is far from optimal due to the presence of the impoundment. For primary habitat components, SCB only scored 49/80 in June and 47/80 in January, with marginal scores in substrate diversity and habitat smothering during both HA. SCB had optimal scores for the other primary habitat components, substrate availability and water velocity. As discussed above, the presence of the impoundment has an impact on the amount of organic matter input such as leaf litter which helps explain the low scores in substrate diversity. Habitat smothering was determined by the number of available pools within an area and by the amount of silt accumulation. In this case, silt accumulation was a particular issue likely caused by the impoundment. Over time, silt and sediment build up within an impoundment, causing the basin to become shallower. The Spring Canyon impoundment contained notably large amounts of silt, and during rain events silt can easily be stirred up within the impoundment and flushed over the top-flow pipe. In the case of large dams, high velocities below an impoundment effectively flush this fine sediment further downstream. However, water velocity at SCB was similar to that of the reference stream which could allow for silt settling below the dam. Accumulated silt is then less likely to be periodically flushed out by rain events due to the regulated flow regime, leading to heavy silt loads.

LSC scored 63/80 for primary habitat components, with marginal scores in substrate availability and suboptimal scores in substrate diversity during both HA. These results were unexpected given that intact steephead ravines streams usually have an abundance of leaf packs, snags, and roots within undercut banks (Rasmussen, 2004). Undercutting was clear at LSC, but root habitat within these areas was less abundant than might be expected which possibly led to the suboptimal ranking in substrate diversity. A marginal ranking in habitat availability is defined by the FDEP as having 6-15% productive habitat and LSC was scored at the high end of this range. This ranking was likely due to the large amounts of open sandy substrate although patches of productive habitat were still present throughout the stretch.

For secondary habitat components, SCB received a score of 72/80, with suboptimal scores for artificial channelization on both HA. Bank stability, riparian buffer zone width, and riparian zone vegetation quality were all scored as optimal on both HA for SCB. Decreased sinuosity due to the dam outfall is what warranted the suboptimal scores for artificial channelization. The reference stream scored 78/80 with optimal scores for all secondary habitat components on both HA.

2.3.3 Aquatic Samples

Thirteen dip-net samples were collected from Spring Canyon and 15 samples from Little Sweetwater Creek over the entire sampling period (Table 2). Identified from these samples were a total of 15,265 specimens representing 60 distinct taxa. The total number of specimens collected from Spring Canyon and Little Sweetwater Creek were 13,485 (34 taxa) and 1,780 (45 taxa) respectively. Of these specimens, 12,526 from SCB and 1,712 from LSC were identified to the target taxon level (family or genus) and were used for community relative abundance (%) calculations (Appendix A) and taxon abundance plots (Fig. 11).

Table 2. Aquatic insect samples collected by month and habitat type from below the dam at the impounded stream, Spring Canyon (Gadsden Co., FL), and at the reference stream, Little Sweetwater Creek (Liberty Co., FL).

	Spring Canyon (Impounded)					Little Sweetwater Creek (Reference)				
	Habitat Type				Total	Habitat Type				Total
Sampling Month	Root	Snag	Sand	Leaf Pack		Root	Snag	Sand	Leaf Pack	
May 2012	X	X	X		3	X	X	X	X	4
August 2012	X	X	X		3		X	X	X	3
November 2012	X	X	X	X	4	X	X	X	X	4
February 2013	X	X	X		3	X	X	X	X	4
Total	4	4	4	1	13	3	4	4	4	15

The abundance plot illustrates that the community below the dam is characterized by an uneven abundance distribution with low taxonomic richness (number of taxa) compared to the reference stream. The overall five dominant taxa collected at SCB were

Chironomidae (7,902, 63.08%), *Hydropsyche* (1,320, 10.54%), *Cheumatopsyche* (696, 5.56%), *Oecetis* (670, 5.35%), and *Microcylloepus* (668, 5.33%; Appendix A). Chironomidae, *Hydropsyche*, and *Cheumatopsyche* were in the five dominant taxa during each sampling session. Chironomids are a widespread and diverse group which commonly occurs in high abundances, and many species are also tolerant of increased water temperatures and can thrive in a variety of aquatic habitats. *Hydropsyche* and *Cheumatopsyche* are net-spinning caddisflies commonly encountered in a variety of lotic habitats throughout Florida. Both genera were able to thrive below the dam using their nets to catch the visibly high amounts of drifting food resources available in the water column coming from the impoundment. *Cheumatopsyche* tend to dominate in warmer stream habitats and are capable of surviving in poor water quality situations, which helps explain why this genus was present in such great abundances at SCB but was not recorded in the aquatic samples from LSC.

Oecetis was one of the five dominant taxa during each sampling session at SCB except May 2012. As a genus, *Oecetis* is a very diverse group capable of utilizing a wide range of habitat types including sand which is often a taxa-poor substrate. Below the dam, *Oecetis* was represented in the top five taxa for all four major habitat types. The genus was even the second most abundant taxon collected within sand habitat at SCB (behind Chironomidae), comprising 24% of all sand samples (n=645). *Oecetis* species were recorded from LSC, but they were not present in large numbers in any of the

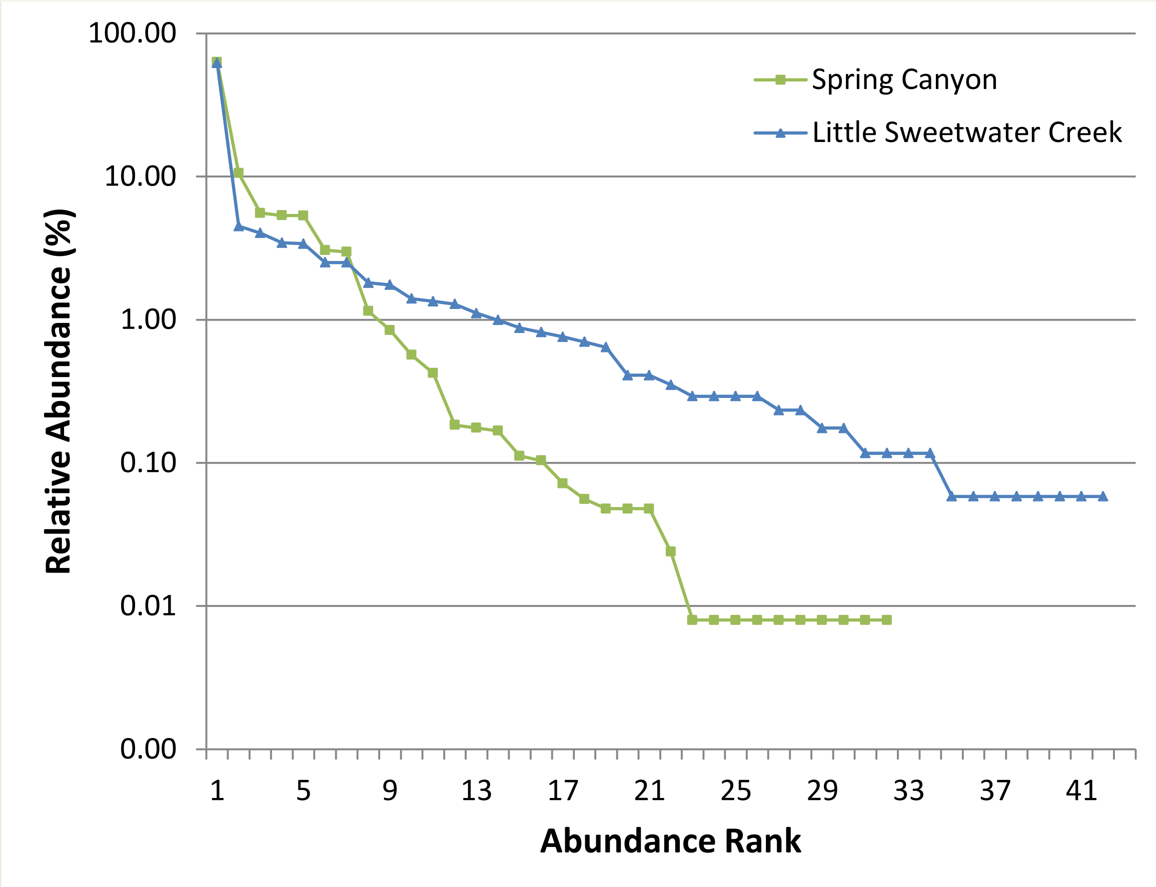

Figure 11. Aquatic insect taxon abundance plots from below the dam of the impounded stream, Spring Canyon, and the reference stream, Little Sweetwater Creek. Taxon abundance ranks according to those listed in Appendix A.

aquatic samples. In addition, they were absent from sand habitat and, as a genus, this group represented less than 2% of the aquatic samples from the reference stream (Appendix A). *Oecetis* are predators so their dominance at SCB could be explained by the sheer number of aquatic insect prey available below the dam. Densities were not calculated for this study but observation and the number of specimens collected indicated that, when compared to the reference site, aquatic insects in general were far more abundant at SCB throughout the study.

Microcylloepus was dominant below the dam except in the February 2013 sample. This beetle genus, represented by only one species in Florida, *M. pusillus*, is one of the most common elmids found in the state. Many elmids cannot tolerate water pollutants and are often considered good indicators of water quality, but *M. pusillus* is known to tolerate silt accumulation like that observed below the dam. However, it is unclear why this taxon was so prolific below the dam but had a low relative abundance (0.18%) at LSC. However, the lack of biological information about this and several other aquatic taxa (e.g., Empididae) was prohibitive when attempting to draw conclusions regarding presence/absence or abundances.

Stenelmis and Simuliidae were the only other taxa represented in the five dominant taxa at SCB throughout the study period. *Stenelmis* is a diverse and common elmid genus found within many of Florida's lotic and lentic habitats. Simuliids, also known as black flies, are often found in high abundances below lake outlets, but at SCB they were only recorded from two samples and their relative abundance was never more than 10% of a sample. Most of the black fly captures at SCB occurred in February, but further production was likely prohibited by intolerably high water temperatures during the summer months.

The LSC community was characterized by a more even abundance distribution with higher taxonomic richness (Fig. 11). The five dominant taxa at LSC for all months combined were Chironomidae (1,062, 59.66%), Empididae (77, 4.33%), *Anisocentropus* (69, 3.88%), *Hydropsyche* (59, 3.31%), and *Acroneuria* (58, 3.26%). Chironomidae was consistently the most dominant taxa for each sample, as was expected. *Anisocentropus* was in the five dominant taxa for each sample except February 2013. This genus, represented by one species in North America, is commonly encountered in Florida's steephead ravine streams. However, *Anisocentropus* along with the other calamoceratid genus, *Heteroplectron*, were not collected below the dam likely due to the increased water temperatures and a lack of leaf pack habitat. *Acroneuria*, a stonefly genus, was recorded in the five dominant taxa in November 2012 and February 2013, but was collected during all sampling sessions at LSC. The only species identifiable from the dip-net samples was *A. lycorias*, a northern relict commonly encountered in steephead ravines, particularly those within the Apalachicola River Basin. *Acroneuria*, like many stoneflies, are warm-water intolerant, which explains their absence from the SCB aquatic collection. In addition, *Acroneuria* are frequently encountered in leaf pack habitats which were plentiful at LSC but limited at SCB, making this site even less suitable for *Acroneuria* survival.

Triaenodes, *Stenelmis*, *Diplectrona*, *Chimarra*, and *Oecetis* were other taxa represented in the five dominant taxa at some point throughout the study at LSC. Of these, the two caddisfly genera, *Diplectrona* and *Chimarra*, were not collected at SCB. *Diplectrona*, represented by one species in Florida, is a ravine specialist in Florida that is often the dominant hydropsychid found in leaf-pack habitats of steephead ravine streams. The lack of substantial leaf pack habitat below the dam, coupled with high water temperatures probably accounts for the absence of this species in the SCB aquatic samples. *Chimarra* are commonly encountered in clear and clean sandy-bottomed streams within Florida. They are filter-feeders which typically benefit below a dam, but some taxa are reportedly intolerant of temperatures just above the maximum temperatures recorded below the dam which could explain their absence at SCB (Moulton et al., 1993).

Taxonomic richness means were not significantly different (p>0.05) between the two sites for root, snag, or sand habitats, and richness values for leaf pack habitats could not be compared due to small sample size (Table 3). Roots were the most taxon rich habitat at SCB and LSC with 32 and 33 taxa respectively. The two sites had at least 12 taxa in common within root habitat. The main differences between the two root communities were a greater taxonomic richness of hydropsychids and Odonates in the root habitat below the dam and the presence of Ephemeroptera taxa at LSC. Snag habitat at LSC was also productive with 25 distinct taxa present, while SCB snag habitat samples contained 20 distinct taxa during the study. The two sites shared at least 11 taxa in common within snag habitats, but the main difference between the communities was the presence of Ephemeroptera and a greater diversity of Trichoptera at the reference site.

Table 3. Summary of the aquatic insect taxonomic richness values from below the dam at the impounded stream, Spring Canyon, and at the reference stream, Little Sweetwater Creek, during four dip-net sampling sessions (May 2012 – February 2013) and in four different habitat types (root, snag, sand, and leaf pack).

Taxon Measure	Spring Canyon (Impounded)					Little Sweetwater Creek (Reference)					t-test
	n	Mean ±SE	Min.	Max.	Total	*n*	Mean ±SE	Min.	Max.	Total	p-value
Sample	4	20.0±3.2	14	26	34	4	28.5±4.3	22	41	45	0.17
Root	4	17.8±3.7	10	24	32	3	18.0±4.7	11	27	33	0.97
Snag	4	12.3±1.9	9	16	20	4	13.5±2.3	7	17	25	0.69
Sand	4	9.0±1.8	5	13	15	4	3.8±1.0	2	6	8	0.07
Leaf Pack	1	8	8	8	8	4	15.3±2.8	8	21	28	**

The statistical values from the 2-sample t-tests of the means are indicated. **Sample size too small for comparison.

Although sand taxonomic richness means were not significantly different between SCB and LSC, overall, the sand habitat below the dam contained more taxa (15) compared to only eight at the reference stream. The difference may be attributed in part to the increased amounts of fine particulate food resources settling on the sands below the dam, potentially increasing the production value of the sand habitat in terms of supporting greater numbers of both prey (e.g., chironomids) and predators such as *Oecetis*. In contrast, sand habitat at LSC, and most other sandy-bottomed streams, is limited in habitat value because it is an unstable substrate with limited food availability.

Leaf pack taxonomic richness means could not be statistically compared, but only eight taxa were recorded in leaf pack habitat at SCB during the November sampling session, whereas 28 taxa were recorded in leaf packs throughout the study at LSC. The low leaf pack taxonomic richness can largely be attributed to the lack of leaf pack habitat throughout the year below the dam.

Taxonomic richness means did not differ between the two streams (t_5=1.59, p=0.17), although the total taxonomic richness at SCB was lower than that at LSC. Overall, 22 families and 22 genera were identified from the SCB dip-net samples, and 30 families and 34 genera were identified from the LSC. During the study, seventeen taxa were recorded at both sites, representing 50% of SCB taxa and 38% of LSC taxa. The two sites shared several Diptera families and elmid genera in common, but their EPT assemblages were drastically different.

EPT richness means were significantly different between the two sites (t_5=3.31, p=0.02). Overall, 13 EPT taxa (including those identified to species) were collected below the dam and 26 taxa were recorded from the reference stream (Table 4). At least 12 Trichoptera taxa were represented in the SCB aquatic samples, but Ephemeroptera were absent and only one Plecoptera taxon was recorded below the dam. In contrast, six

Ephemeroptera taxa and two Plecoptera taxa were represented in the reference stream samples along with 18 Trichoptera taxa. Of the SCB Trichoptera taxa, two families, Hydropsychidae and Leptoceridae, dominated the aquatic EPT samples, containing 98.9% of all EPT specimens (Fig. 12). The Trichoptera at the reference stream were less dominated by hydropsychids and leptocerids, with these two taxa accounting for only 46% of the aquatic EPT specimens collected (Fig. 12). Rasmussen (2004) found similar dominance patterns of hydropsychids and leptocerids in his study of Trichoptera and Plecoptera within Apalachicola steephead streams. Calamoceratidae was also well represented (>10%) in the LSC samples, and the remaining eight Trichoptera families collected from LSC accounted for 14% of the total abundance of EPT, whereas below the dam there were only three other families which represented <1% of the aquatic EPT family composition below the dam (Fig. 12).

Table 4. Summary of the Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness values below the dam at the impounded stream, Spring Canyon, and at the reference stream, Little Sweetwater Creek, during four dip-net sampling sessions (May 2012 – February 2013) and in four different habitat types (root, snag, sand, and leaf pack).

EPT Richness Measure	Spring Canyon (Impounded)					Little Sweetwater Creek (Reference)					t-test
	n	Mean ±SE	Min.	Max.	Total	*n*	Mean ±SE	Min.	Max.	Total	p-value
Sample	4	7.0 ±1.8	3	10	13	4	16.5 ±2.3	11	22	26	0.02*
Root	4	6.8 ±1.7	3	10	11	3	9.7 ±1.7	8	13	18	0.28
Snag	4	5.3 ±1.1	3	8	8	4	7.5 ±1.5	3	9	14	0.28
Sand	4	4.0 ±0.9	2	6	7	4	0.8 ±0.5	0	2	2	0.03*
Leaf Pack	1	3	3	3	3	4	8.8 ±2.1	4	14	16	**

The statistical values from the 2-sample t-tests of the means are indicated. *Indicates significant difference for α=0.05. **Sample size too small for comparison.

Mean EPT richness was significantly higher in SCB sand samples as compared to LSC (t_4= -3.15, p=0.03; Table 4). As discussed above, *Oecetis* was a major component of the SCB sand samples during the study, but was absent from LSC sand habitat. Three additional Trichoptera genera from the SCB sand samples were not recorded from the reference stream sand habitat, but only *Oecetis* represented more than 2% of SCB sand sample specimens. EPT richness means were not significantly different ($p>0.05$) in the root or snag habitats of the two sites, and leaf pack EPT richness means could not be compared.

Overall, the two sites shared only four Trichoptera taxa and one Plecoptera taxon in common, representing 38% of the SCB EPT taxa and 14% of the total taxa. At LSC, this represented only 19% of EPT taxa and 11% of the total taxa. Ephemeroptera and Trichoptera shredder taxa were a large component of the EPT taxa present at LSC but missing from the community below the dam. Percent Ephemeroptera (%E) means were significantly different (t_3=9.48, $p<0.01$; Table 5) between the two sites due to Ephemeroptera absence below the dam. Many Ephemeroptera taxa are intolerant of increased water temperatures (Berner & Pescador, 1988), so the conditions during the summer at SCB could preclude the presence of many Ephemeroptera taxa. In addition, the abundance of *Oecetis* predators in silt-laden sand habitats at SCB could explain the absence of widespread and tolerant mayfly taxa.

EPT Family Composition

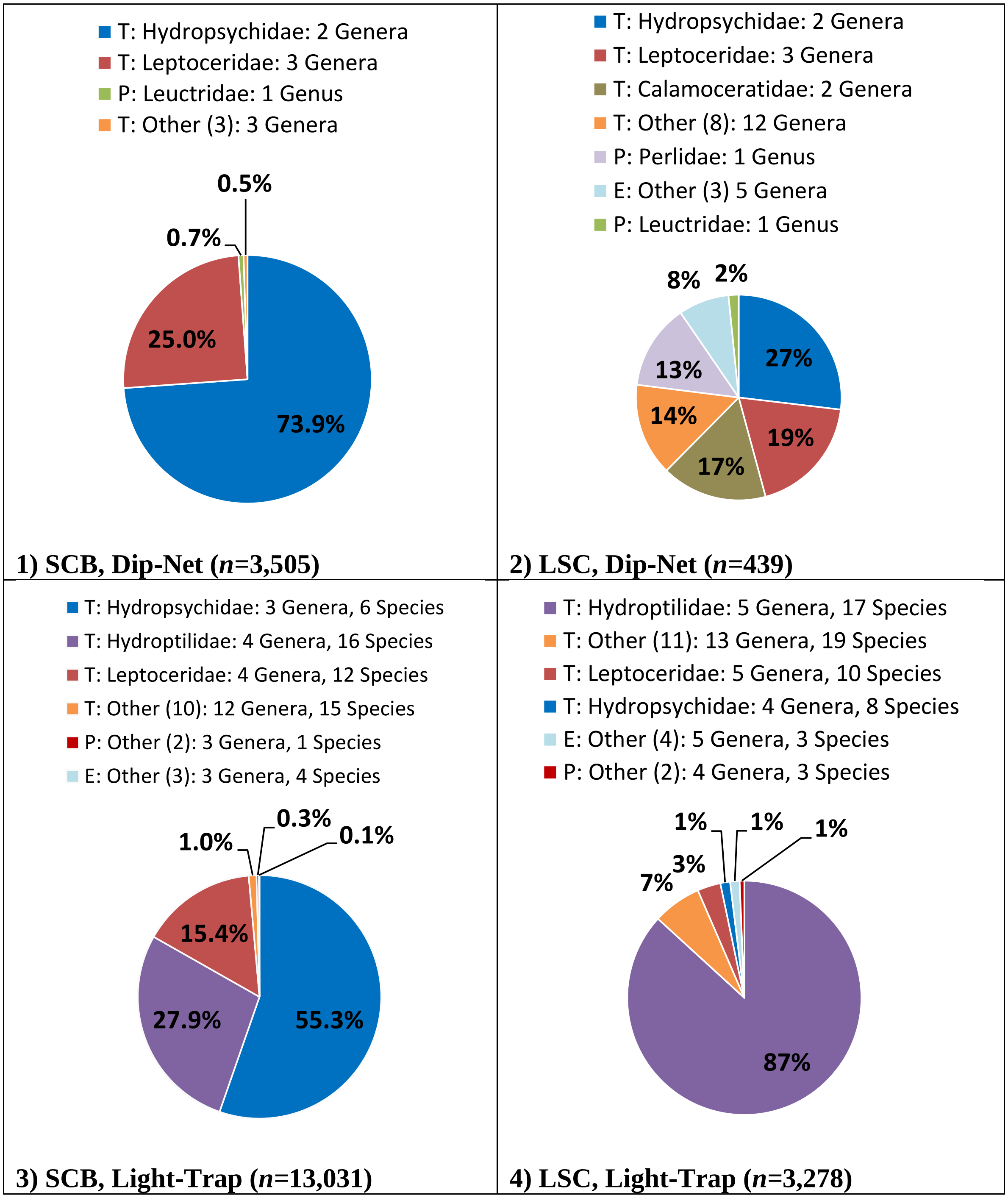

Figure 12. Percent composition (total individuals per family/total number of individuals collected from the site) of Ephemeroptera, Plecoptera, and Trichoptera (EPT) families from 1), 2) dip-net samples (May 2012 – February 2013) and 3), 4) light-trap samples (May 2012 – April 2013) from below the dam of the impounded stream, Spring Canyon (Gadsden Co., FL), and from the reference stream, Little Sweetwater Creek (Liberty Co., FL).

Average percent Plecoptera (%P) was also significantly lower below the dam (t_3=4.40, p=0.02, Table 5). Many Plecoptera taxa found in steephead communities are cold-water adapted so increased summer water temperatures below the dam likely prohibit the survival of many of these taxa. In addition, leaf pack habitats are preferred by several Plecoptera taxa found in Florida. The lack of leaf pack habitat below the dam during much of the year probably contributed to the low representation of this group at SCB. Percent Trichoptera means were not significantly different (t_4= -1.07, p>0.05) between SCB and LSC (Table 5).

Table 5. Summary of the Ephemeroptera, Plecoptera, and Trichoptera (EPT) percentages below the dam at the impounded stream, Spring Canyon, and at the reference stream, Little Sweetwater Creek, during four dip-net sampling sessions (May 2012 – February 2013).

EPT % Measure	Spring Canyon (Impounded)					Little Sweetwater Creek (Reference)					t-test
	n	Mean ±SE	Min.	Max.	Total (%)	*n*	Mean ±SE	Min.	Max.	Total (%)	p-value
EPT	4	0.55 ±0.1	0.44	0.70	27.7	4	0.57 ±0.05	0.45	0.68	26.1	0.87
E	4	0	0	0	0	4	0.14 ±0.01	0.11	0.17	2.1	<0.01*
P	4	0.02 ±0.02	0	0.07	0.2	4	0.22 ±0.04	0.16	0.35	3.9	0.02*
T	4	0.55 ±0.06	0.44	0.70	27.6	4	0.48 ±0.03	0.40	0.53	20.1	0.34

Individual order percentages were calculated by dividing the number of captures from an order by the total number of captures. Percentage data were transformed using an arcsin transformation (arcsin[√%]). Actual percentages are displayed in Total columns, all other data shown are transformed. The statistical values from the 2-sample t-tests of the means are indicated. *Indicates significant difference for α=0.05.

From the aquatic samples, 13,129 SCB specimens and 1,720 LSC specimens were categorized into one of six FFG. Of these six categories, average filtering-collector (FC) and shredder (SH) percentages were significantly different between the two sites (p≤0.05;

Table 6). Filtering-collectors made up an overall larger percentage of the community below the dam (20.8%) than at the reference stream (9.5%). The observed difference was a result of the high abundances of the hydropsychid caddisflies, *Hydropsyche* and *Cheumatopsyche*, below the dam. Combined, these genera represented 16% of the total samples at SCB, but at LSC, only *Hydropsyche* was recorded and represented just 3% of the specimens. LSC is a clear stream with limited amounts of food resources available in the water column for FC. Also, higher amounts of drifting food resources, like that observed below the impoundment, can support larger numbers of FC than other FFG because they are stationary feeders that do not compete as directly for food or space resources. For instance, net-spinners like the hydropsychids need only enough space to build their nets to capture food particles floating downstream, whereas SH have to disperse to find and compete for a limited food source (e.g., leaf litter).

As mentioned above, SH taxa were poorly represented below the dam with only 43 of the 13,129 categorized specimens belonging to the SH FFG. *Leuctra*, a commonly encountered stonefly genus, represented more than half of SH specimens below the dam. Calamoceratids, important shredder taxa often found in steephead ravines, were absent from the SCB samples, but represented more than half of the SH collected at the reference stream. Shredders feed on various types of detritus, usually materials like leaf litter and other coarse particulate organic matter (CPOM), but these food sources have been severely limited by the dam. This decrease in detritus and the increase in drifting food resources from the impoundment have shifted the functionality of the SCB aquatic insect community to a GC/FC dominated community as opposed to the GC dominated reference stream.

Gathering-collector (GC) percentages were not significantly different between the two sites and were the numerically dominant FFG comprising more than 65% of the aquatic samples at SCB and LSC. The dominant taxon, Chironomidae, was categorized as GC and accounts for the majority of the GC reported in the aquatic samples.

Table 6. Summary of percentages of aquatic insects assigned to functional feeding groups (FFG). Samples collected below the dam at the impounded stream, Spring Canyon (N=13,129), and at the reference stream, Little Sweetwater Creek (N=1,720), during four dip-net sampling sessions (May 2012 – February 2013).

FFG	Spring Canyon (Impounded)					Little Sweetwater Creek (Reference)					t -test
	n	Mean ±SE	Min.	Max.	Total (%)	*n*	Mean ±SE	Min.	Max.	Total (%)	p-value
GC	4	0.96 ±0.04	0.85	1.04	69.1	4	0.89 ±0.05	0.77	1.02	65.8	0.36
FC	4	0.48 ±0.05	0.36	0.62	20.8	4	0.30 ±0.05	0.21	0.43	9.5	0.05*
PR	4	0.27 ±0.01	0.25	0.29	6.8	4	0.38 ±0.05	0.24	0.51	11.9	0.14
SC	4	0.12 ±0.04	0.03	0.20	2.9	4	0.23 ±0.01	0.20	0.27	5.1	0.08
SH	4	0.03 ±0.02	0	0.08	0.3	4	0.27 ±0.04	0.18	0.35	7.6	<0.01*
PC	4	0.03 ±0.01	0	0.05	0.1	4	0.02 ±0.01	0	0.05	0.1	0.61

Percentage data were transformed using an arcsin transformation (arcsin[√%]). Actual percentages are displayed in Total columns, all other data shown are transformed. The statistical values from the 2-sample t-tests of the means are indicated. GC=gathering-collectors; FC=filtering-collectors; PR=predators; SC=scrapers; SH=shredders; PC=vegetative-piercers. *Indicates significant difference for α=0.05.

2.3.4 Light-Trap Samples

From the eight light-trap samples, a total of 16,309 specimens representing 83 distinct EPT taxa were identified. The total number of specimens and taxa from SCB and LSC were 13,031 (56) and 3,278 (63) respectively. Of the specimens collected, 3,371 from SCB and 1,394 from LSC were identified to species and used to calculate relative abundance (%) (Appendix B) and species abundance plots (Fig. 13). The abundance plot illustrates that EPT species richness was slightly lower and the abundance distribution was less even at SCB as compared to the reference stream. The five dominant species collected from SCB were *Cheumatopsyche analis* (753, 22.34%), *Oecetis inconspicua* (527, 15.63%), *Ceraclea maculata* (522, 15.49%), *Hydropsyche decalda* (311, 9.23%), and *Hydroptila armata* (259, 7.68%; Appendix B). The abundance of *Cheumatopsyche* and *Hydropsyche* in aquatic samples corresponded with high numbers of adults collected during all light-trapping sessions (including unidentifiable females). Some conditions, like those observed below the dam, including warm water temperatures and high food availability are known to increase aquatic insect growth and production rates (Huryn et al., 2008) which could explain the high number of adults collected in all seasons in both *Cheumatopsyche* and *Hydropsyche*. At LSC, only three specimens of *Cheumatopsyche analis* were collected in the light-trap samples. *Cheumatopsyche analis* is highly tolerant of water pollution and habitat degradation and while it is occasionally found in unspoiled habitats like LSC, it is more often encountered within impacted stream systems, such as SCB, where few other caddisflies occur. *Hydropsyche decalda* is common throughout Florida but not typically a steephead ravine species as was demonstrated by its absence from the LSC light-trap collection. Its high abundances below the dam demonstrate that it, like *C. analis*, is a warm-water tolerant species capable of thriving in disturbed habitats.

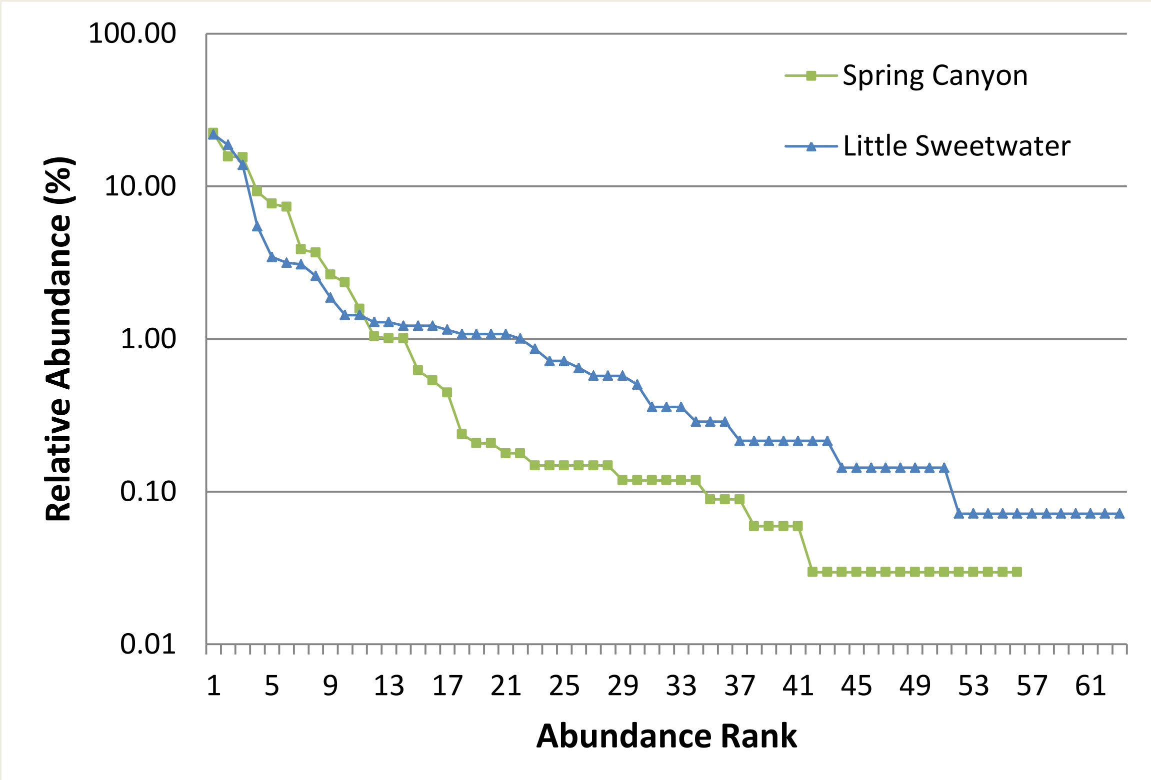

Figure 13. Adult Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxon abundance plots from below the dam of the study stream, Spring Canyon, and the reference stream, Little Sweetwater Creek. Taxon abundance ranks according to those listed in Appendix B.

Oecetis inconspicua is probably the most common *Oecetis* species in Florida and during this study was consistently one of the three most dominant species collected below the dam. *Ceraclea maculata* is also a widespread and commonly occurring species in Florida which was consistently in the top five at SCB throughout the sampling period. In contrast, only one specimen was collected in the LSC light-trap samples and *Ceraclea* was not recorded in the LSC aquatic samples. It is unclear why this genus was not better represented at the reference stream. *Hydroptila armata* was only dominant in the November SCB light-trap sample, but it was the most abundant species collected in that sampling session (256, 36.36%). Other species represented in the top five throughout the sampling period at SCB included *Oecetis osteni* and *Cheumatopsyche virginica,* neither of which were collected from the reference stream. *Oecetis osteni* is primarily associated with aquatic vegetation in lentic and slow-moving lotic habitats (Floyd, 1995; Pescador et al., 2004), and *C. virginica* is a commonly occurring species in Florida's lentic and lotic habitats.

Both sites experienced exceptionally high hydroptilid numbers during November sampling. The abundances were so significant that numerically four of the five overall dominant taxa at LSC were hydroptilids. The only reason any other taxa occurred in the top five at SCB was because hydropsychids and leptocerids were so consistently abundant throughout the study. Even though some of the light-trap samples were dominated by this family, relatively few hydroptilid specimens were collected in the aquatic samples at either site (SCB=16; LSC=3). It is unclear what may have caused this disconnect in numbers outside of adult dispersion from neighboring habitats.

Outside of the four hydroptilids, *Agarodes libalis* was the other overall dominant species at LSC (48, 3.44%). This is consistent with previous findings in sandy-bottom steephead streams, except this genus was not recorded in the LSC aquatic samples. *Agarodes* utilizes sand habitats along the stream edge, so this taxon was probably not collected during aquatic sampling sessions due to the randomized collection protocol established for sand habitat. Similarly, *Agarodes libalis* was also collected in SCB light-trap samples, but larvae were absent from aquatic samples.

Overall, the SCB light-trap collection contained 56 different species, and the LSC light-trap samples contained 63 species. The two sites had thirty-six taxa in common, representing 64% of SCB taxa and 57% of LSC taxa. Eleven of the species were hydroptilids which will not be discussed further for the reasons stated above. Other similarities included several *Oecetis* and hydropsychid species. *Anisocentropus* was also present at both sites, even though it was absent from the aquatic samples from SCB. Some of the variation between the light-trap and aquatic samples could be explained by what is referred to as the Colonization Cycle, in which adult insects exhibit upstream flight behavior (Müller, 1982). Considering the low EPT richness in the aquatic samples below the dam, it is also assumed that a portion of the taxa collected in the light-traps are from aquatic habitats outside of the study stream (e.g., small adjacent seepage streams flowing into the main branch of Spring Canyon). Noteworthy species differences included a narrow-range endemic (*Nyctiophylax morsei*) and a southeastern endemic (*Nyctiophylax serratus*) collected from the reference stream. Additionally, *Rhyacophila carolina*, a common cool-stream species, was only recorded at LSC.

Mean EPT species richness values were not significantly different between SCB and LSC (t_5=0.81, p=0.46; Table 7). Breaking it down by order, 4 Ephemeroptera species, 3 Plecoptera taxa, and 49 Trichoptera species were recorded at SCB, and Trichoptera comprised more than 99% of the total captures. At LSC, 5 Ephemeroptera taxa, 4 Plecoptera taxa, and 54 Trichoptera species were collected with Trichoptera representing 98% of the total light-trap captures. Ephemeroptera and Plecoptera did not constitute a large percentage of either community which was expected, especially for light-trap samples as not all Ephemeroptera and Plecoptera taxa are attracted to light. By order, the two sites shared 1 Ephemeroptera, 2 Plecoptera, and 33 Trichoptera species in common. Average percent Trichoptera (%T) was significantly higher at SCB than LSC

(t_4= -2.79, p=0.05), but mean Ephemeroptera and Plecoptera percentages were not significantly different between the two sites (p>0.05; Table 7).

Table 7. Total richness of the Ephemeroptera (E), Plecoptera (P), Trichoptera (T), and order percentages from below the dam at the impounded stream, Spring Canyon, and at the reference stream, Little Sweetwater Creek, during four light-trapping sessions (May 2012 – February 2013).

EPT Measure	Spring Canyon (Impounded)					Little Sweetwater Creek (Reference)					t-test
	n	Mean ±SE	Min.	Max.	Total	*n*	Mean ±SE	Min.	Max.	Total	p-value
Richness	4	24.8 ±3.8	19	36	56	4	30.3 ±5.7	21	44	63	0.46
%E	4	0.02 ±0.01	0	0.06	0.1	4	0.14 ±0.04	0.06	0.24	1.3	0.07
%P	4	0.03 ±0.02	0	0.09	0.3	4	0.06 ±0.04	0	0.17	0.6	0.49
%T	4	1.53 ±0.02	1.48	1.57	99.6	4	1.41 ±0.04	1.33	1.51	98.1	0.05*

Individual order percentages were calculated by dividing the number of captures from an order by the total number of captures. Percentage data were transformed using an arcsin transformation (arcsin[√%]). For percentage measures, actual percentages are displayed in Total columns, all other data shown for percentage comparisons are transformed. The statistical values from the 2-sample t-tests of the means are indicated.

Hydroptilidae were the most well-represented family at both sites with 16 species at SCB and 17 species at LSC; however, as discussed above, it is likely that the light-trap observations misrepresent the hydroptilid populations actually occurring in the studied aquatic reaches. Leptoceridae were the second most represented family at both sites but were a larger percentage of the EPT community below the dam due to the diversity and high abundances of *Oecetis* species (Fig. 12). Hydropsychids were also well represented at both sites, but, similarly to the aquatic sample findings, this family represented over half of the EPT abundance at SCB and was only 1% of the EPT abundance at LSC. Also, combined, the ten other Trichoptera families represented just 1% of the EPT abundance at SCB community, whereas the eleven other families from LSC accounted for 7% of the overall EPT abundance (Fig. 12).

From the light-trap samples, 9,902 specimens of EPT identified from SCB and 1,412 EPT specimens identified from LSC were categorized into one of the six FFG. Of the six categories, average filtering-collector (FC), scraper (SC), and vegetative-piercer (PC) percentages were significantly different between the two sites ($p \leq 0.05$; Table 8). The SC taxonomic assemblages were similar between SCB and LSC ($t_3=3.56$, $p=0.04$) but were less abundant in the SCB light-trap samples. Hydroptilids were the only PC recorded so the observed significant difference ($t_5=4.26$, $p=0.01$) is not a reliable characterization of either stream. Similarly to the aquatic samples, extremely high abundances of hydropsychid caddisflies (>5,000) explains the higher percentage of FC in the light-trap samples below the dam ($t_3= -3.45$, $p=0.04$). Outside of FC, SCB had a higher percentage of PR EPT than the other FFG which was primarily due to high abundances of *Oecetis*. At LSC, overall FC, PR, SC, and SH percentages were similar. The total percentages of SH were considerably higher at LSC than SCB, even though the difference between the means was not significant. Shredders were numerically similar between the two sites, but within the SCB samples they were easily overshadowed by FC. *Agarodes libalis* was the dominant SH species at both sites.

The aquatic and light-trap samples both demonstrate the differences in the aquatic insect communities at SCB and LSC, and how the impoundment has impacted the Spring Canyon community on a structural and functional level. Structurally, the aquatic community illustrated an uneven abundance distribution at SCB with low total taxonomic richness compared to the reference stream. Both sampling methods also showed a community dominated by only a few taxa, which are not typically characteristic of steephead ravine habitats. Functionally, Spring Canyon is characterized by high percentages of FC, as observed in both sampling methods, and the EPT were particularly dominated by this FFG. These findings support the rejection of the null hypotheses.

Table 8. Summary of percentages of adult Ephemeroptera, Plecoptera, and Trichoptera representing larval functional feeding groups (FFG). Samples collected below the dam at the impounded stream, Spring Canyon (N=9,902), and at the reference stream, Little Sweetwater Creek (N=1,412), during four light-trapping sessions (May 2012 – April 2013).

FFG	Spring Canyon (Impounded)					Little Sweetwater Creek (Reference)					t-test
	n	Mean ±SE	Min.	Max.	Total (%)	*n*	Mean ±SE	Min.	Max.	Total (%)	p-value
GC	4	0.21 ±0.04	0.09	0.29	6.3	4	0.07 ±0.05	0	0.21	1.6	0.08
FC	4	0.90 ±0.17	0.41	1.20	73.2	4	0.26 ±0.07	0.05	0.36	9.1	0.04*
PR	4	0.34 ±0.05	0.25	0.47	14.2	4	0.23 ±0.06	0.04	0.32	7.6	0.22
SC	4	0.04 ±0.02	0	0.08	0.3	4	0.22 ±0.05	0.08	0.31	7.8	0.04*
SH	4	0.08 ±0.02	0.03	0.12	0.9	4	0.17 ±0.04	0.10	0.29	6.4	0.14
PC	4	0.17 ±0.06	0.08	0.30	5.2	4	0.57 ±0.08	0.48	0.80	67.5	0.01*

Percentage data were transformed using an arcsin transformation (arcsin[√%]). Actual percentages are displayed in Total columns, all other data shown are transformed. The statistical values from the 2-sample t-tests of the means are indicated. GC=gathering-collectors; FC=filtering-collectors; PR=predators; SC=scrapers; SH=shredders; PC=vegetative-piercers. *Indicates significant difference for α=0.05.

CHAPTER THREE

AQUATIC INSECT COMMUNITY RECOVERY IN A FLORIDA STEEPHEAD RAVINE, FIVE YEARS POST DAM REMOVAL

3.1 Introduction

Dam removal has become a prominent social and environmental concern in recent years, but the consequences of removal for aquatic ecosystems are still unclear. The majority of dam removal projects have involved small dam structures; however, information on the impacts and recovery of biotic communities in cases of small dam removal are particularly scarce, with fishes having received the most attention to date. Two studies that examined aquatic insect communities after small dam removal on larger streams were presented by Stanley et al. (2002) and Thomson et al. (2005). Information is lacking on aquatic insect community recovery below dams removed in low-order streams. It is important to understand the long-term results of dam removal on biotic communities, including aquatic insects, so better removal strategies may be developed to improve overall biotic recovery and shorten recovery time in all stream sizes.

The goals of this investigation were to 1) characterize the recovery of aquatic insect community structure and function five years after the removal of a small dam on a low-order steephead ravine stream, in comparison with the aquatic insect community of an undisturbed steephead ravine stream; and 2) characterize the recovery of Ephemeroptera, Plecoptera, and Trichoptera (EPT) structure and function, five years after the removal of a small dam on a low-order steephead ravine stream, by using pre-removal and post-removal data.

3.1.1 Study Area

Two steephead ravine streams were studied on the eastern side of the Apalachicola River in the central Florida panhandle (Fig. 14). Both stream systems flow into the Apalachicola River in Liberty County and are located on The Nature Conservancy's Apalachicola Bluffs and Ravines Preserve (Chapter 2, Fig. 4). The first study site is Kelley Branch, described below. The second study site, Little Sweetwater Creek, was selected as the reference stream because it is an intact, undisturbed steephead ravine stream in close proximity and similar in size to Kelley Branch. An above-impoundment reference site was not used because the impoundment impacted such a long stretch of Kelley Branch that only the upper headwaters remained intact. Aquatic insect taxa differ with stream size so comparing the community below the removed dam with that in the headwaters would have provided an unreliable comparison.

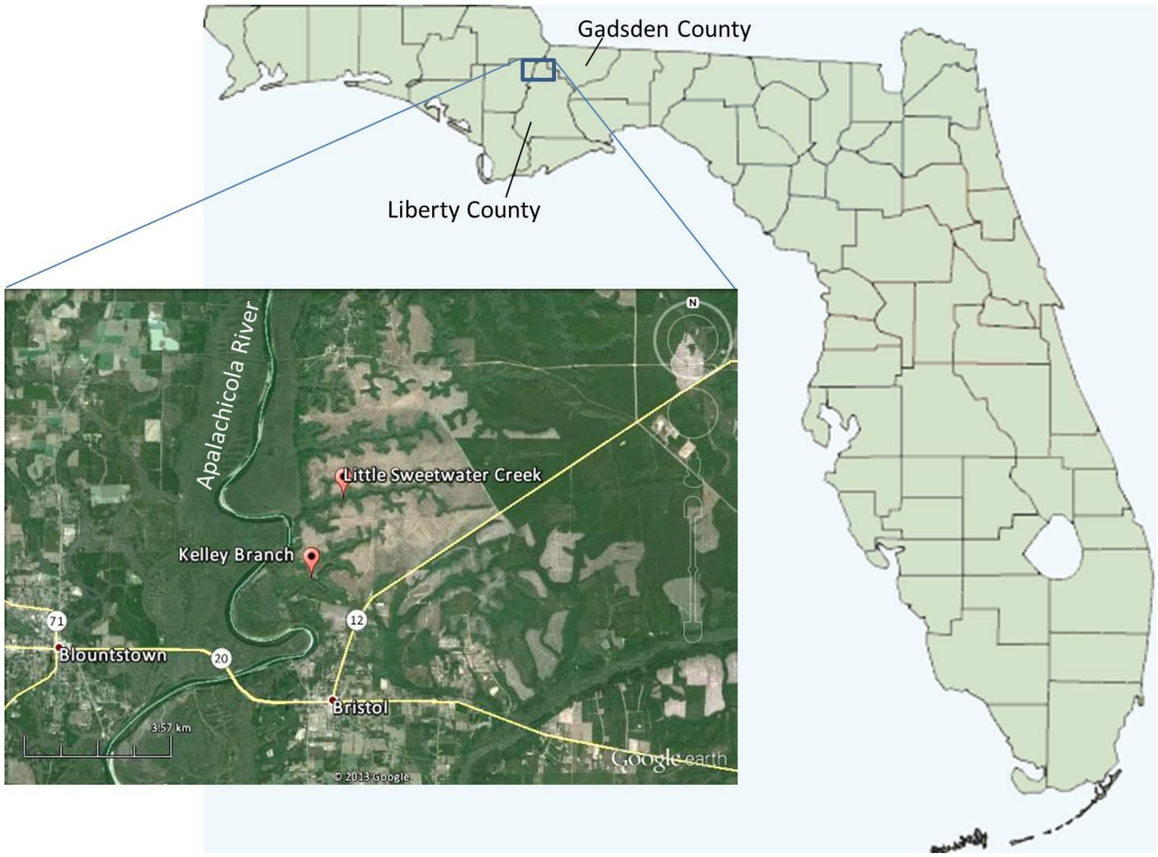

Figure 14. The steephead ravine systems studied in Liberty County, Florida (Little Sweetwater Creek and Kelley Branch). Aerial image courtesy of Google Earth.

3.1.2 Kelley Branch Dam Removal

Kelley Branch is a small, third-order stream that flows into the Apalachicola River on The Nature Conservancy's Apalachicola Bluffs and Ravines Preserve (ABRP) in Liberty County Florida. The Nature Conservancy (TNC) purchased the Garden of Eden Tract, where Kelley Branch is located, in 1982, and the remainder of the property (total of 6,000 acres) was acquired from St. Joe Paper Company in 1984 (D. J. Printiss, personal communication, March 4, 2014) with the goal of restoring and conserving the unique species and habitats found on the property, including several steephead ravine streams like Kelley Branch. However, in the 1950's, before TNC acquired the property, Kelley Branch, was impounded by a 4 meter high earthen dam approximately 2,000 m from its confluence with the Apalachicola River (Fig. 15). The resulting 20 acre impoundment was an intended recreational area for a planned housing development that was never constructed. From February to March 2006, TNC, in conjunction with the Florida Fish and Wildlife Conservation Commission (FWC) and the U.S. Fish and Wildlife Service (USFWS), drained the impoundment to begin the restoration of the steephead ecosystem (Fig. 16). Once the impoundment was drained, almost 800 meters of the stream channel were reconstructed and the dam structure was removed in July 2007. In addition, the old lakebed was planted with more than 1,000 seedlings of slope forest tree species (Ritchie, n.d.). However, many trees were lost to beaver activity in the years since. Efforts were made by TNC to remove the beavers and dams in late 2011, and the area continues to be periodically monitored for activity.

Figure 15. A) The old earthen dam and culvert at Kelley Branch in 2006, and B) the old impoundment in 2006, located on The Nature Conservancy's Apalachicola Bluffs and Ravines Preserve in Liberty County, FL. Images courtesy of Florida Department of Environmental Protection.

Figure 16. The old impoundment bed and new stream channel after dam removal on Kelley Branch at The Nature Conservancy's Apalachicola Bluffs and Ravines Preserve in Liberty County, FL. Retrieved from http://www.nature.org/ourinitiatives/regions/northamerica/unitedstates/florida/interactive-media-kelley-branch-dam-removal.xml

The fish and aquatic insect communities were monitored throughout the planning and restoration processes. The fish community was monitored by TNC personnel in fall and spring from 2005 to 2010. As would be expected, fish passage improved dramatically after dam removal and the community responded quickly to the restored flow regime (S. J. Herrington, personal communication, February 28, 2014). The Florida Department of Environmental Protection (FDEP) monitored the aquatic insect community below the impoundment in January 2006, one year prior to removal; in January 2007, after the impoundment draw-down phase; and for two years after dam removal, in February 2008 and 2009. In 2006, before drawn-down and dam removal, the site failed the FDEP's BioRecon biological community standards for healthy stream ecosystems with a score of four (FDEP, 2006). The BioRecon score range is 0-10, and a stream fails with a score from 0 to 5 and passes with a score from 6 to 10. Six biometric measures were considered when determining this score (total taxa, Ephemeroptera [mayfly] taxa, Trichoptera [caddisfly] taxa, long-lived taxa, clinger taxa, and sensitive taxa) and Kelley Branch failed all of them. However, in 2007, during the draw-down phase, the BioRecon rating improved to passing, with a score of 7, and passed four out of six of the biometric measures (FDEP, 2007). The aquatic insect community improved dramatically, with 18 EPT taxa present, compared to the seven EPT taxa from before the draw-down process. In 2008, after complete dam removal, the FDEP BioRecon classified the stream's biological community as moderately impaired with a failing score of five and failing five of six biometrics (FDEP, 2008). The loss of diversity was likely a result of the upstream dam removal and channel construction which increased sedimentation and siltation downstream (FDEP, 2008). One year later, the community had improved with a BioRecon score of eight and EPT taxa improving from 11, in 2008, to 24, in 2009.

3.2 Materials and Methods

To carry out this investigation, aquatic insects and their terrestrial adults were collected below the old dam location and at an undisturbed reference ravine stream within the Apalachicola River Basin. Relative abundances, taxonomic richness, and EPT taxa measures were determined and compared between the two sites to look for any structural differences between the communities. Aquatic insects were also categorized into functional feeding groups (FFG) and percentages were calculated and compared to look for differences in community function between the two streams. The aquatic EPT families collected below the removed dam were also compared to those collected by the FDEP before dam removal, during impoundment drawdown, and after dam removal to assess EPT community structure recovery since dam removal. Aquatic EPT FFG from below the removed dam were also compared to the previously collected EPT FFG to assess EPT community functional recovery since dam removal. The hypothesis for this investigation was that the structural and functional attributes of the aquatic insect community below the removed dam are in the process of returning to the pre-impoundment state.

3.2.1 Study Site Locations

In the spring of 2012, one 50 meter aquatic reach and one light-trapping station were established at both stream systems. The aquatic reaches were measured using a 50 m tape, starting downstream (00 m) and running upstream (50 m) (Fig. 5). The Kelley Branch aquatic sampling reach was located below the old dam site (KDB; Fig. 17) with the light-trapping station located along the bank at 25 m (N 30°27’30.9”, W 084°58’53.0”). The Little Sweetwater Creek aquatic sampling reach (LSC; Chapter 2, Fig. 7) was measured in the lower reaches of the system to obtain a reach similar in size and order to the Kelley Branch study site. The corresponding light-trapping station was located along the bank at 25 m (N 30°28'34.1", W 084°58'22.8").

Figure 17. Kelley Branch aquatic sampling site (KDB) on Apalachicola Bluffs and Ravines Preserve in Liberty Co., FL. The 00 m (downstream) coordinates for the aquatic sampling reach are as follows: N 30°27’31.0”, W 084°58’52.2”. Image courtesy of Google Earth.

3.2.2 Abiotic and Habitat Parameters

Several abiotic parameters were measured over the course of the study. Air temperature, wind speed, relative humidity, general weather conditions, water depth, and canopy cover were collected during each site visit using the same protocols as those described in Chapter Two. Water temperature was collected continuously on a one hour interval as previously described. Conductivity, pH, dissolved oxygen (DO), current velocity, and stream width were measured quarterly using the protocols discussed in the previous chapter. Habitat assessments were conducted by FDEP personnel in June 2012 and January 2013 as previously discussed.

3.2.3 Aquatic Insect Sampling

Aquatic insects were sampled seasonally from May 2012 – February 2013. Kelley Branch samples were collected 17 May, 16 August, 15 November, and 14 February, and Little Sweetwater Creek samples were collected 19 May, 18 August, 17 November, and 16 February. Samples were collected from major habitats using the protocol described in Chapter Two.

3.2.4 Light-Trap Sampling

Light-trapping was conducted to collect the terrestrial adult stages of Ephemeroptera, Plecoptera, and Trichoptera (EPT) in order to obtain species-level data, as discussed in the previous chapter. Trapping was conducted from May 2012 – May 2013. Kelley Branch samples were collected 8 August, 2 November, 19 March, and 7 May, and Little Sweetwater samples were collected 2 May, 8 August, 2 November, and 19 March. The KDB May light-trap sample was taken in 2013 because the first sample obtained in 2012 had been collected from within the old impoundment bed. It was later determined that the May 2012 sample would be inadequate in representing the taxa at the study site below the removed dam, so a replacement sample was collected the following year in May. Light-trap samples were collected following the protocol described in Chapter Two.

3.2.5 Specimen Identification

The May and August aquatic samples were sorted completely using a Leica S6E stereomicroscope (6.3-40X) and fiber optic light source (Techniquip FOI-150) with no additional sample preparation or subsampling. The November and February samples were prepared and sorted following the protocol described in the previous chapter. Aquatic insects were identified and light-trap samples were sorted and identified using the materials and methods detailed in Chapter Two. For all KDB aquatic samples, Dipterans were identified to family, EPT were identified to species when possible, and all others were identified to genus. Specimens were also categorized into FFG as previously described in Chapter Two.

3.2.6 Statistical Analysis

Physicochemical properties of KDB and LSC were analyzed as described in the previous chapter. Taxon databases were developed so structural and functional aspects could be analyzed and compared between KDB and LSC using the same procedure from Chapter Two.

Kelley Branch aquatic insect collection data from 2006-2009 was acquired from FDEP to examine any changes in the aquatic insect community since the removal process began. The FDEP bioassessment collection protocol uses an in-field hand picking method for aquatic insect sampling. While useful for bioassessment purposes, this method underrepresents many of the smaller organisms such as chironomids. Therefore, only the EPT taxa were analyzed from the FDEP data since representatives of these orders are generally more visible and adequately represented in handpicked samples. In addition, because the FDEP samples were all collected in January or February, only the February aquatic sample from this study was used for community comparisons. Statistical comparisons were not used because only one sample per year was available for evaluation. Graphical displays of EPT family percentages and FFG percentages were used to analyze any changes in community and/or recovery since dam removal.

3.3 Results and Discussion

3.3.1 Physicochemical Parameters

The physicochemical characteristics of both streams are summarized in Table 9. Average stream width was significantly narrower in KDB as compared to LSC (t_{19}= -3.35, $p<0.01$). However, before impoundment, KDB may have been a wider stream but channelization has occurred due to dam presence and subsequent dam removal. Average stream depth was slightly shallower at KDB as compared to LSC (t_{61}= -2.40, p=0.02); mean current velocity was not significantly different ($p>0.05$) between KDB and LSC. Average canopy cover was statistically significantly different between KDB and LSC (t_{35}= -2.96, $p<0.01$), but, as discussed in the previous chapter, the difference was biologically slight since both streams had on average more than 95% canopy cover.

The highest temperatures recorded throughout the study were 30.6°C and 25.7°C at KDB and LSC respectively. The lowest recorded temperatures were 7.1°C below the removed dam and 9.0°C at the reference stream. Average daily water temperatures below the removed dam were not significantly different between the two sites when taking the entire year into account (n=365; t_{694}= -1.29, p=0.20; Table 9). However, by separately analyzing the mean daily summer and winter water temperatures, it was found that water temperature in KDB were on average slightly warmer than those of LSC during the summer months (n=182; t_{346}=5.67, $p<0.001$) but not significantly different in the winter period (n=183; t_{357}= -1.56, p=0.12; Fig. 18). The warmer temperatures in the summer can

Table 9. Summary of the physical and chemical properties of the removed-dam study stream, Kelley Branch, and the reference stream, Little Sweetwater Creek, from May 2012-May 2013 (*n*= number of samples).

Properties	Kelley Branch (Removed Dam)				Little Sweetwater Creek (Reference)				*t*-test
	n	Mean ±SE	Min.	Max.	*n*	Mean ±SE	Min.	Max.	p-value
Physical									
Stream Width (m)	12	2.9 ±0.2	2.1	3.4	12	3.9 ±0.2	3.0	5.1	<0.01*
Water Depth (m)	33	0.15 ±0.01	0.08	0.24	33	0.18 ±0.01	0.12	0.31	0.02*
Canopy Cover (%)	33	97.8 ±0.6	87.5	100	33	99.6 ±0.1	96.9	100	<0.01*
Avg. Daily Water Temp.: May ’12 – May ‘13 (°C)	365	19.5 ±0.2	9.3	26.4	365	19.1 ±0.2	10.7	24.2	0.20
Avg. Daily Water Temp.: May ’12 – Oct ‘12 (°C)	182	23.2 ±0.2	13.5	26.4	182	22.0 ±0.1	13.8	24.2	<0.01*
Avg. Daily Water Temp.: Nov ‘12 – May ‘13 (°C)	183	15.8 ±0.2	9.3	21.5	183	16.2 ±0.2	10.7	21.1	0.12
Current Velocity (m/s)	4	0.3 ±0	0.3	0.3	4	0.3 ±0	0.3	0.3	**
Chemical									
pH	4	5.1 ±0.3	4.7	5.7	4	3.7 ±0.4	2.8	4.9	0.04*
Dissolved Oxygen (mg/L)	4	9.8 ±0.6	8.4	10.8	4	10.3 ±0.8	8.8	12.0	0.62
Conductivity (umhos/cm)	4	17.5 ±0.9	16	19	4	14.5 ±0.3	14	15	0.05*

The statistical values from the 2-sample t-tests are indicated. *Indicates significant difference for α=0.05. **Values are identical.

be explained by the upstream stretch that runs through the old impoundment bed. Although there is no longer an impoundment present to collect large amounts of solar radiation, the 800 m of stream running through the old impoundment bed is still wide open with little to no canopy cover. As the stream runs through the old impoundment bed, it is warmed by solar radiation and ambient warming, thereby resulting in warmer waters below the old dam site. Similarly, KDB experiences lower minimum temperatures in the winter months because the old impoundment bed is more susceptible to ambient cooling due to the lack of forestation (Fig. 18).

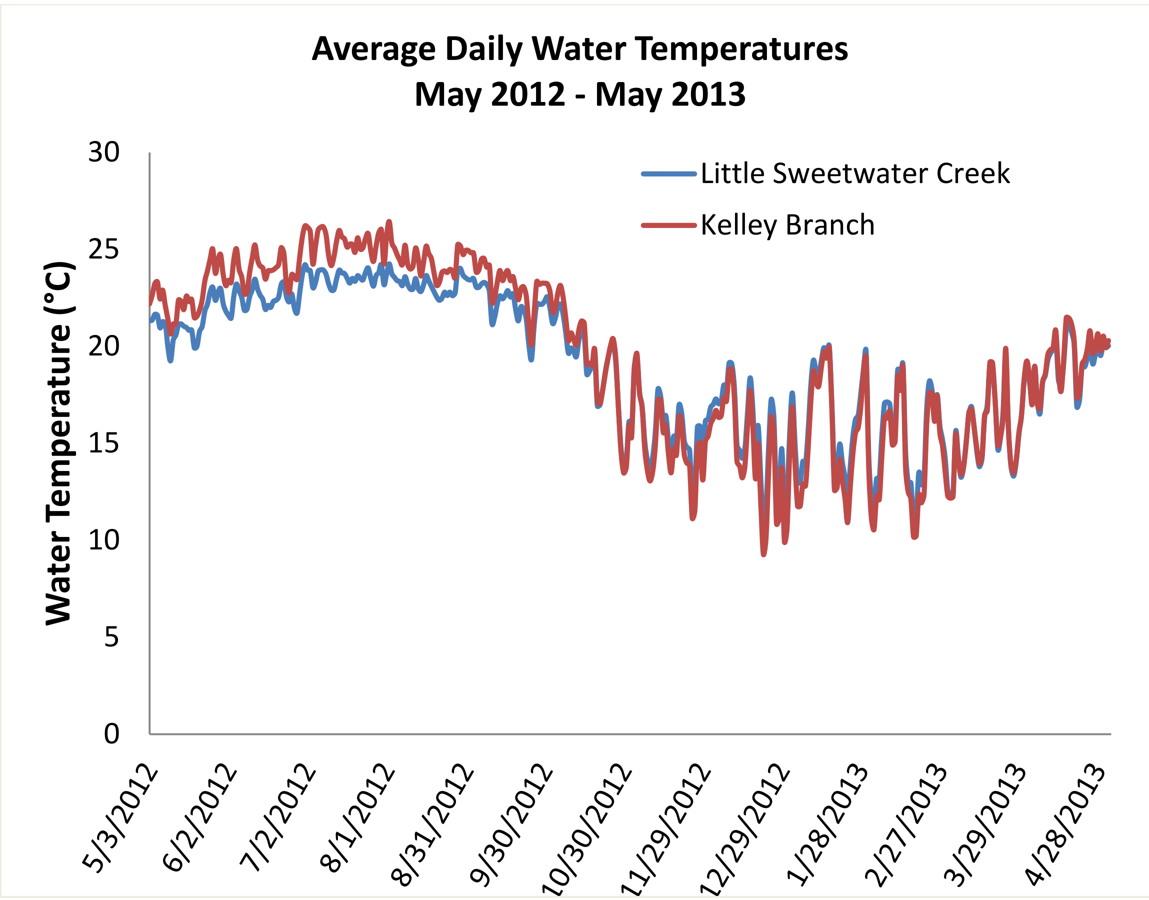

Figure 18. Average daily water temperatures below the removed dam of the study stream, Kelley Branch, and at the reference stream, Little Sweetwater Creek, from May 2012 – May 2013, in Liberty Co., FL.

As was discussed in Chapter Two, aquatic insects often cannot tolerate increased water temperatures. The maximum temperatures below the removed dam are close to the upper temperature limit for many aquatic insects, although it is less extreme than what probably occurred prior to dam removal. In addition, dissolved oxygen (DO) levels remained similar to those recorded at the reference stream (t_5= -0.53, p=0.62) and recorded levels never dropped below 8 mg/L.

Average pH was significantly different between the two sites (t_4=2.89, p=0.04). LSC was more acidic than KDB but, as previously mentioned, the sandy streams of the Panhandle are characteristically acidic. KDB was substantially less acidic which could be a residual effect of impoundment (Neel, 1963; Ziser, 1985).

Conductivity levels were low at both sites. The mean conductivity levels were slightly higher in KDB than in LSC; however, the slight difference (3 umhos/cm) is probably not biologically meaningful. As mentioned in the previous chapter, further data collection of the chemical properties of both streams is necessary before reliable comparisons can be made regarding these parameters.

3.3.2 In-stream Habitat

Sand, roots, and snags were major components of the in-stream habitat below the removed dam throughout the study. KDB had noticeably more exposed root habitat than LSC which could be a result of the scouring that likely occurred during the impoundment draw-down and dam removal. Leaf packs were consistently available at LSC and in higher amounts than KDB. Leaf packs were only a major habitat component at KDB during the November sampling session, when leaf fall was near its peak in the wooded riparian area. KDB has less leaf litter and other organic material input due to the openness of the old impoundment bed and the minimal stream length before and within the study stretch where input can occur from the surrounding riparian habitat. The canopy cover at the reference stream is intact along the entire length of the stream so leaf litter input can be more consistent throughout the year.

For the habitat assessments (HA) conducted by FDEP, KDB received an overall suboptimal ranking of 118/160 in June 2012, but received an overall optimal rating of 121/160 for the January HA. LSC HA results were discussed in the previous chapter. Based on the overall HA scores, KDB is in border-line optimal condition in terms of stream health, but breaking down the scores into primary and secondary habitat components, the in-stream habitat at KDB has not fully recovered after dam removal. For primary habitat components, KDB only scored 42/80 in June 2012, with marginal scores for substrate diversity, substrate availability, and habitat smothering. In January 2013, KDB scored 44/80 for the primary habitat components, with marginal scores continuing for substrate diversity and substrate availability, but a suboptimal score for habitat smothering. As discussed above, the lack of trees within the old impoundment bed limits the amount of organic matter input which limits the types and amount of habitat available. Habitat smothering is scored based on the number of pools available within the stretch which can be impacted by silt or sand smothering. In this case, sand accumulation has limited the number of pool habitats at KDB. This is likely a continued result of dam removal and stream channel construction within the old impoundment bed. Increased or improved sediment transport is a known result of dam removal, although it can cause habitat smothering in reaches immediately below the old dam site (Born et al., 1998; Doeg & Koehn, 1994; Sethi et al., 2004). This may be a prolonged issue at Kelley Branch due to the fact that so much channel reconstruction was required following dam removal which may have further increased the amount of sediment transport downstream.

For the secondary habitat components, KDB received a score of 76/80 in June 2012, with optimal scores for all components. However, in January 2013, KDB still scored a 77/80, but received a suboptimal score for bank stability on one side of the stream. The stream bank still shows signs of extensive scouring in places which likely occurred when flows were increased during impoundment draw-down. Significant rainfall events may have a greater impact on these areas of scouring, leading to periodic suboptimal scores.

3.3.3 Aquatic Samples

Thirteen dip-net samples were collected from Kelley Branch and 15 from Little Sweetwater Creek over the entire sampling period (Table 10). These samples contained a total of 5,790 specimens from 56 distinct taxa. The total number of specimens and taxa collected from KDB and LSC were 4,010 (44) and 1,780 (45) respectively. Of these specimens, 3,806 from KDB and 1,712 from LSC were identified to the target taxon level (family or genus) and were used for community relative abundance (%) calculations (Appendix A) and taxon abundance plots (Fig. 19).

The abundance plot illustrates that the aquatic insect communities below the removed dam and at the reference stream were similar in overall structure. The communities were comparable in taxonomic richness and showed similar abundance distributions. The overall five dominant taxa at KDB were Chironomidae (2,037; 53.52%), *Microcylloepus* (431; 11.32%), *Hydropsyche* (343; 9.04%), Empididae (116; 3.05%), and *Triaenodes* (103, 2.71%). Chironomidae, *Hydropsyche,* and *Microcylloepus* were consistently in the top five taxa, with Chironomidae and *Hydropsyche* in the top three for all four sampling sessions. As discussed in the previous chapter, chironomids typically occur in high abundances, and *Hydropsyche* is a common genus of net-spinning

Table 10. Aquatic insect samples collected by month and habitat type from below the removed dam at the study stream, Kelley Branch, and at the reference stream, Little Sweetwater Creek (Liberty Co., FL).

	Kelley Branch (Removed Dam)					Little Sweetwater Creek (Reference)				
	Habitat Type				Total	Habitat Type				Total
Sampling Month	Root	Snag	Sand	Leaf Pack		Root	Snag	Sand	Leaf Pack	
May 2012	X	X	X		3	X	X	X	X	4
August 2012	X	X	X		3		X	X	X	3
November 2012	X	X	X	X	4	X	X	X	X	4
February 2013	X	X	X		3	X	X	X	X	4
Total	4	4	4	1	13	3	4	4	4	15

caddisflies found in a variety of Florida's streams. Also, *Microcylloepus* is one of the more tolerant elmid genera although low abundances at LSC are not well understood.

Triaenodes is a case-building caddisfly genus typically only found in roots or aquatic macrophytes (Glover, 1996). KDB had ample amounts of root habitat available to support a population since no aquatic vegetation was growing below the removed dam. *Lype*, *Chimarra*, *Nectopsyche*, Simuliidae, and *Oxyethira* were also represented in the top five at KDB at some point during the study. Of these, *Oxyethira*, a microcaddisfly genus, was not collected in the reference stream samples.

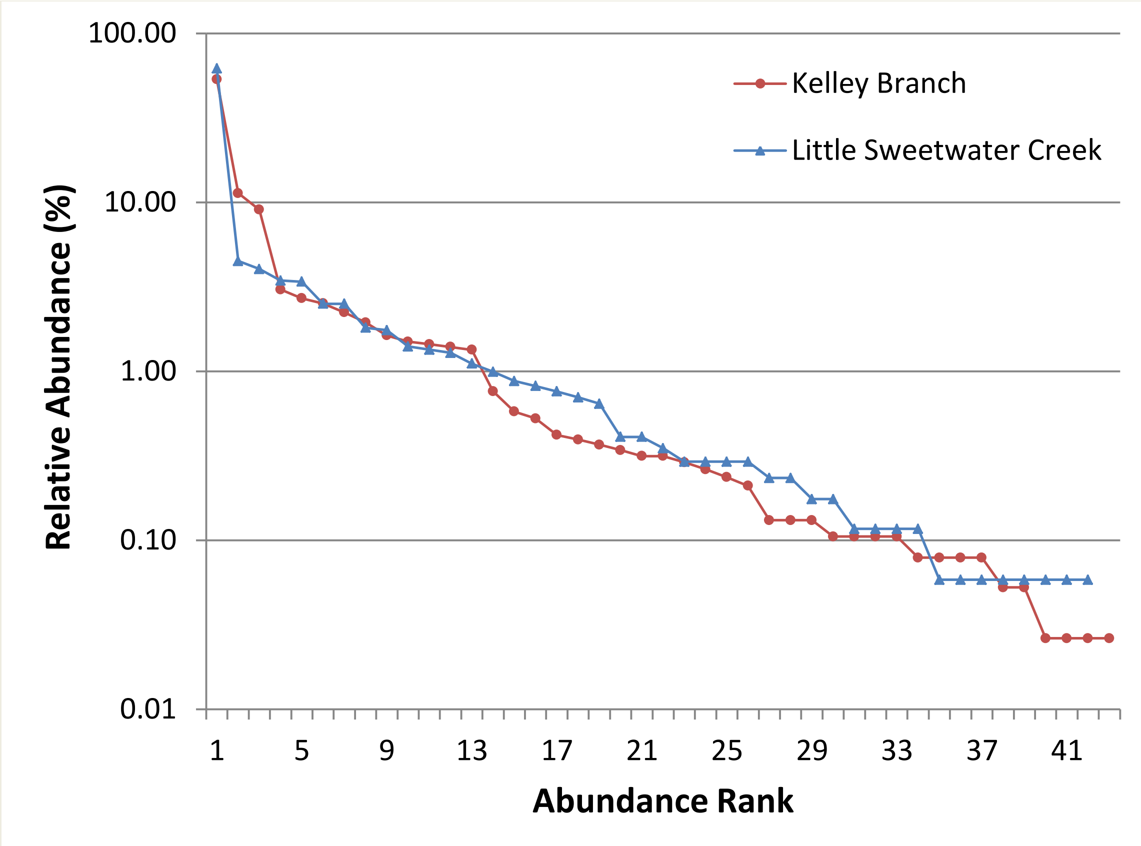

Figure 19. Aquatic insect taxon abundance plots for the removed-dam study stream, Kelley Branch, and the reference stream, Little Sweetwater Creek. Taxon abundance ranks according to those listed in Appendix A.

As discussed in the previous chapter, the five dominant taxa overall at LSC were Chironomidae, Empididae, *Anisocentropus*, *Hydropsyche*, and *Acroneuria.* Each of these taxa was also found at KDB, although *Anisocentropus* and *Acroneuria* were represented by only a few specimens collected in the November 2012 leaf pack sample. Both taxa are strongly associated with leaf packs which were limited below the removed dam during the other three sampling periods. Further reforestation of the old impoundment bed will be necessary for the relative abundances of these taxa to increase.

Taxonomic richness means were not significantly different between the two sites for root, snag, or sand habitat, and leaf pack richness means could not be compared due to small sample size (Table 11). Roots were the most taxon rich habitat at KDB and LSC with 35 and 33 taxa respectively. Within the root habitat, the two sites shared 20 taxa in common during the study. The difference in the two root communities was largely attributed to the presence of hydroptilids and hydropsychids and a greater diversity of Odonates in the roots below the removed dam. KDB snag habitat was also productive with 32 distinct taxa recorded, while only 25 taxa were reported on snags at LSC. The average number of taxa reported on snag habitat at the two sites were not significantly different (t_3=2.49, p=0.09), but the higher total taxa at KDB could be attributed to greater amounts of snag habitat available. In general, KDB had more large coarse woody debris than LSC throughout the study, which could support an overall greater abundance of aquatic insects. Of the 25 taxa reported at LSC, 18 of them were also collected at KDB indicating that the KDB snag community was similar but more taxon rich than the reference stream. In addition, the two sites had 13 taxa in common in leaf pack habitat even though only one leaf pack sample was collected from KDB. The KDB leaf pack community was similar in composition to that of LSC, but low taxonomic richness can be attributed to the lack of quality leaf pack habitat throughout the year.

Overall, 35 genera from 27 families were identified from the KDB aquatic samples, and 30 families and 34 genera were identified from the LSC aquatic samples. Nine orders were represented in the LSC samples, of these, only Megaloptera were not collected in the KDB aquatic samples. Over the course of the study, the two streams had 33 taxa in common, representing 75% of the aquatic taxa collected from KDB and 73% of the taxa from LSC. Taxonomic richness means were not significantly different between KDB and LSC (t_4=0.26, p=0.81), and total taxonomic richness was similar between the two sites. These results demonstrate that KDB is taxonomically similar to LSC. The main differences observed were greater Odonate richness at KDB and more shredder taxa at LSC.

Table 11. Summary of the aquatic insect taxonomic richness values at the removed-dam study stream, Kelley Branch, and the reference stream, Little Sweetwater Creek, during four dip-net sampling sessions (May 2012-February 2013) and in four different habitat types (root, snag, sand, and leaf pack).

Taxonomic Richness	Kelley Branch (Removed Dam)					Little Sweetwater Creek (Reference)					t-test
	n	Mean± SE	Min.	Max.	Total	*n*	Mean± SE	Min.	Max.	Total	p-value
Sample	4	29.8± 2.2	25	34	44	4	26.0± 3.8	21	37	45	0.81
Root	4	21.3± 1.9	16	24	35	3	18.0± 4.7	11	27	33	0.59
Snag	4	19.3± 0.5	18	20	32	4	13.5± 2.3	7	17	25	0.09
Sand	4	5.8± 0.6	4	7	14	4	3.8± 1.0	2	6	8	0.17
Leaf Pack	1	19	19	19	19	4	15.3± 2.8	8	21	28	**

The statistical values from the 2-sample t-tests of the means are indicated. **Sample size too small for comparison.

Average EPT richness was also not significantly different between the two sites (t_4= -0.29, p=0.79; Table 12). Twenty-four distinct EPT taxa (including those identified to species) were recorded in dip-net samples from below the removed dam. Five Ephemeroptera taxa, four Plecoptera taxa, and 15 Trichoptera taxa were present in the KDB samples. Trichoptera represented 90% of the aquatic EPT collection at KDB, with hydropsychids being the dominant family and comprising 43% of the aquatic EPT specimens (Fig. 20). Leptocerids were also a major component of the aquatic EPT assemblage, containing 24% of the community. The remaining six Trichoptera families at KDB represented 23% of the aquatic EPT captures from KDB. The Plecoptera family, Perlidae was well represented at LSC (13%), but only made up 2% of the aquatic EPT assemblage below the removed dam. In Florida, this group lives primarily in leaf pack habitats (Pescador et al., 2000), so low amounts of available leaf litter at KDB likely made this an inadequate site for higher perlid populations. A full description of the LSC EPT assemblage was provided in Chapter Two.

Average EPT richness values were not significantly different in root, snag, or sand habitats, and leaf pack samples could not be compared due to small sample size (Table 12). More EPT taxa were reported in LSC leaf pack habitat than were not found in KDB leaf packs; however, several of these taxa were collected from other KDB habitats.

Overall, the two sites shared 4 Ephemeroptera, 2 Plecoptera, and 12 Trichoptera taxa in common, representing 75% of the KDB EPT taxa and 41% of the total taxa. At LSC, this represented 69% of the EPT taxa and 40% of the total taxa. The individual order percentage means did not differ between the two sites ($p>0.05$; Table 13).

Table 12. Summary of the larval Ephemeroptera, Plecoptera, and Trichoptera (EPT) collected at the removed-dam study stream, Kelley Branch, and the reference stream, Little Sweetwater Creek, during four dip-net sampling sessions (May 2012-February 2013) and in four different habitat types (root, snag, sand, and leaf pack).

EPT Richness Measure	Kelley Branch (Removed Dam)					Little Sweetwater Creek (Reference)					t-test
	n	Mean ±SE	Min.	Max.	Total	*n*	Mean ±SE	Min.	Max.	Total	p-value
Sample	4	15.8 ±1.3	13	19	24	4	16.0 ±2.3	11	22	26	0.79
Root	4	12.3 ±0.5	11	13	19	3	9.7 ±1.7	8	13	18	0.28
Snag	4	10.8 ±1.1	8	13	18	4	7.5 ±1.5	3	9	14	0.14
Sand	4	1.3 ±0.3	1	2	4	4	0.8 ±0.5	0	2	2	0.41
Leaf Pack	1	9	9	9	9	4	8.8 ±2.1	4	14	16	**

Individual order percentages are proportions of the sample's EPT captures, not total captures. The statistical values from the 2-sample t-tests of the means are indicated. *Indicates significant difference for α=0.05. **Sample size too small for comparison.

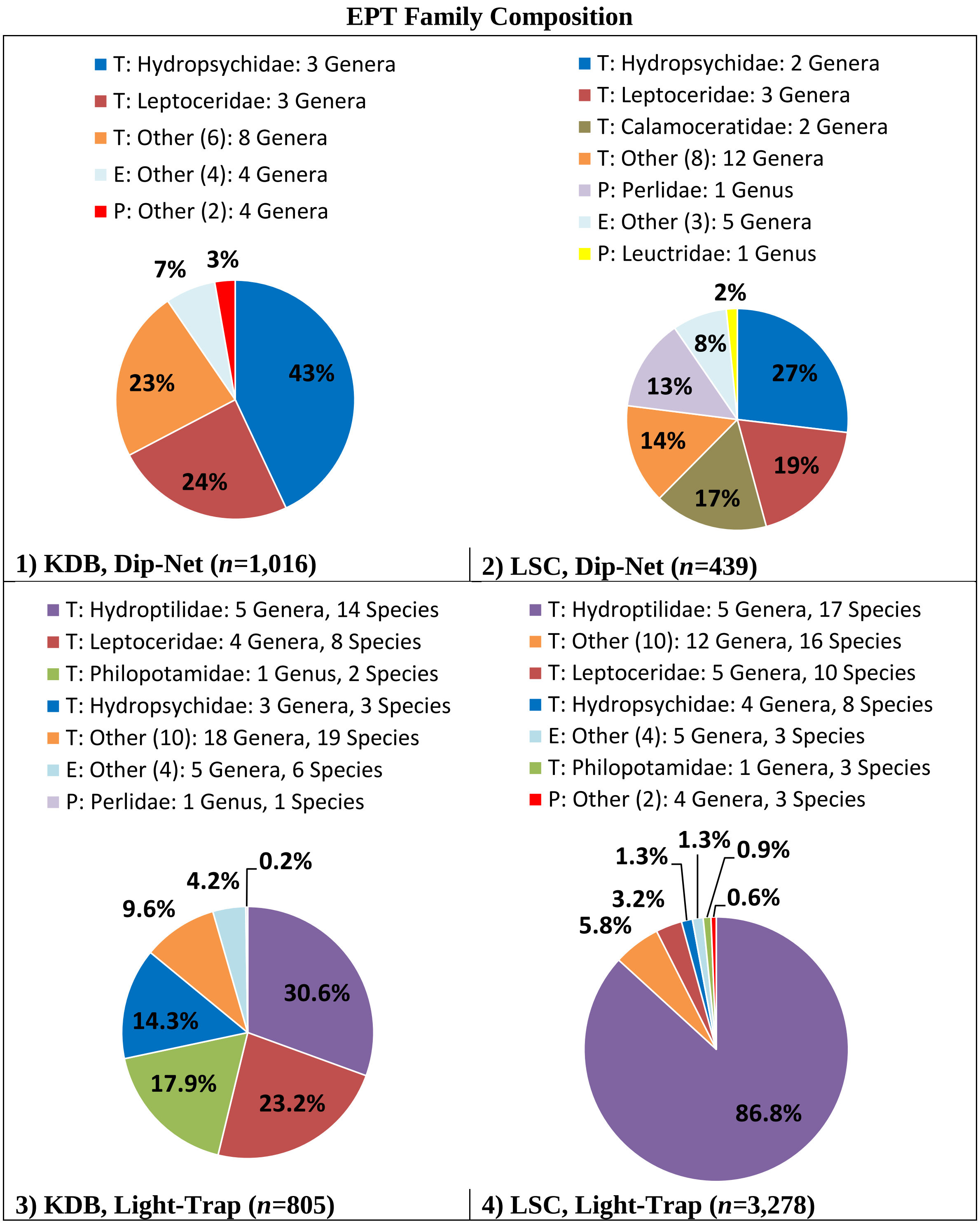

Figure 20. Composition of Ephemeroptera, Plecoptera, and Trichoptera (EPT) families from 1), 2) dip-net samples (May 2012 – February 2013) and 3), 4) light-trap samples (May 2012 – May 2013) from below the removed dam of the study stream, Kelley Branch, and from the reference stream, Little Sweetwater Creek (Liberty Co., FL).

Table 13. Summary of the Ephemeroptera, Plecoptera, and Trichoptera (EPT) percentages below the removed-dam study stream, Kelley Branch, and at the reference stream, Little Sweetwater Creek, during four dip-net sampling sessions (May 2012 – February 2013).

EPT % Measure	Kelley Branch (Removed Dam)					Little Sweetwater Creek (Reference)					t-test
	n	Mean ±SE	Min.	Max.	Total (%)	*n*	Mean ±SE	Min.	Max.	Total (%)	p-value
% EPT	4	0.58 ±0.06	0.41	0.68	26.5	4	0.57 ±0.05	0.45	0.68	26.1	0.88
% E	4	0.15 ±0.01	0.13	0.17	2.3	4	0.14 ±0.01	0.11	0.17	2.1	0.74
% P	4	0.08 ±0.02	0.04	0.11	0.7	4	0.22 ±0.04	0.16	0.35	3.9	0.06
% T	4	0.55 ±0.06	0.37	0.65	23.5	4	0.48 ±0.03	0.40	0.53	20.1	0.42

Individual order percentages were calculated by dividing the number of captures from an order by the total number of captures. Percentage data were transformed using an arcsin transformation (arcsin[√%]). Actual percentages are displayed in Total columns, all other data shown are transformed. The statistical values from the 2-sample t-tests of the means are indicated.

From the aquatic samples, 3,867 KDB specimens were categorized into one of the six FFG. None of the FFG category means were significantly different between SCB and LSC ($p>0.05$; Table 14). Overall, gathering-collectors (GC) made up more than 65% of the aquatic samples at both sites as was expected per the discussion in the previous chapter. Combined, predators (PR) and filtering-collectors (FC) made up 21% of both communities, but KDB had an overall higher percentage of FC than LSC. The dominant FC taxa group below the removed dam was hydropsychid caddisflies, particularly *Hydropsyche*. A sizeable *Hydropsyche* population was observed living on aquatic macrophytes growing in abundance in the upstream stretch that runs through the old impoundment bed. It is possible that the *Hydropsyche* population below the removed dam is being periodically supplemented by the drift of organisms from the aquatic macrophytes upstream, thereby increasing their relative abundance downstream.

Table 14. Summary of percentages of aquatic insects assigned to functional feeding groups (FFG). Samples collected below the removed dam at the study stream, Kelley Branch (N=3,867), and at the reference stream, Little Sweetwater Creek (N=1,720), during four dip-net sampling sessions (May 2012 – February 2013).

FFG	Kelley Branch (Removed Dam)					Little Sweetwater Creek (Reference)					t-test
	n	Mean ±SE	Min.	Max.	Total (%)	*n*	Mean ±SE	Min.	Max.	Total (%)	p-value
GC	4	0.91 ±0.07	0.78	1.09	68.8	4	0.89 ±0.05	0.77	1.02	65.8	0.83
FC	4	0.40 ±0.03	0.30	0.46	14.5	4	0.30 ±0.05	0.21	0.43	9.5	0.15
PR	4	0.27 ±0.04	0.21	0.40	6.6	4	0.38 ±0.05	0.24	0.51	11.9	0.19
SC	4	0.23 ±0.05	0.13	0.35	5.2	4	0.23 ±0.01	0.20	0.27	5.1	0.94
SH	4	0.17 ±0.01	0.15	0.20	3.2	4	0.27 ±0.04	0.18	0.35	7.6	0.08
PC	4	0.13 ±0.06	0.05	0.29	1.7	4	0.02 ±0.01	0	0.05	0.1	0.16

Percentage data were transformed using an arcsin transformation (arcsin[√%]). Actual percentages are displayed in Total columns, all other data shown are transformed. The statistical values from the 2-sample t-tests of the means are indicated. GC=gathering-collectors; FC=filtering-collectors; PR=predators; SC=scrapers; SH=shredders; PC=vegetative-piercers.

3.3.4 Light-Trap Samples

A total of 4,083 EPT specimens representing 74 taxa were identified from the eight light-trap samples. The total number of specimens and taxa collected from KDB and LSC were 805 (50) and 3,278 (63) respectively. Of the specimens collected, 384 from KDB and 1,394 from LSC were identified to species and were used for community relative abundance (%) calculations (Appendix B) and species abundance plots (Fig. 21). The abundance plot illustrates that the KDB terrestrial EPT assemblage had a similar abundance distribution as that of LSC but overall species richness was low compared to LSC. The overall five dominant species collected from KDB were *Oecetis sphyra* (43, 11.20%), *Nectopsyche candida* (30, 7.81%), *Hydropsyche elissoma* (30, 7.81%), *Oxyethira novasota* (20, 5.21%), and *Maccaffertium smithae* (18, 4.69%; Appendix B). However, fifteen different species were represented in the top five at some point during the course of the study, which further illustrates the even abundance distribution at KDB. Of these fifteen, only *Oxyethira elerobi*, a microcaddisfly, was not also collected from LSC, indicating a high degree of similarity between the two sites.

Regarding the overall dominant species, *O. sphyra* is a widespread species throughout the panhandle which was also present in the aquatic samples. *Nectopsyche candida* was collected in the light-trap samples from both sites and *Nectopsyche* was 2% of the KDB aquatic community. *Oxyethira novasota*, a microcaddisfly, occurs throughout the southeast, but in Florida its distribution is primarily restricted to steephead ravine streams of the Apalachicola River Basin and the western panhandle (Rasmussen et al., 2008). *Maccaffertium smithae* is a commonly occurring mayfly species in the panhandle, known to live in a variety of lotic habitats and utilize a variety of habitat substrates. This species was encountered at both sites and the genus was represented in the aquatic samples from KDB and LSC. *Hydropsyche elissoma*, a net-spinning caddisfly, is the dominant hydropsychid species of the lower reaches of Florida’s steephead streams (Rasmussen, 2004). As previously discussed, this group may have been supplemented by the drift of upstream larval populations.

As it was discussed in Chapter 2, the dominance of hydroptilids at LSC is likely a misrepresentation of the aquatic community because so few were collected in the aquatic samples. However, hydroptilids were more abundant in the KDB aquatic samples and correspondingly abundant in the light-trap samples which suggests that the KDB light-trap hydroptilid collection is more reliable.

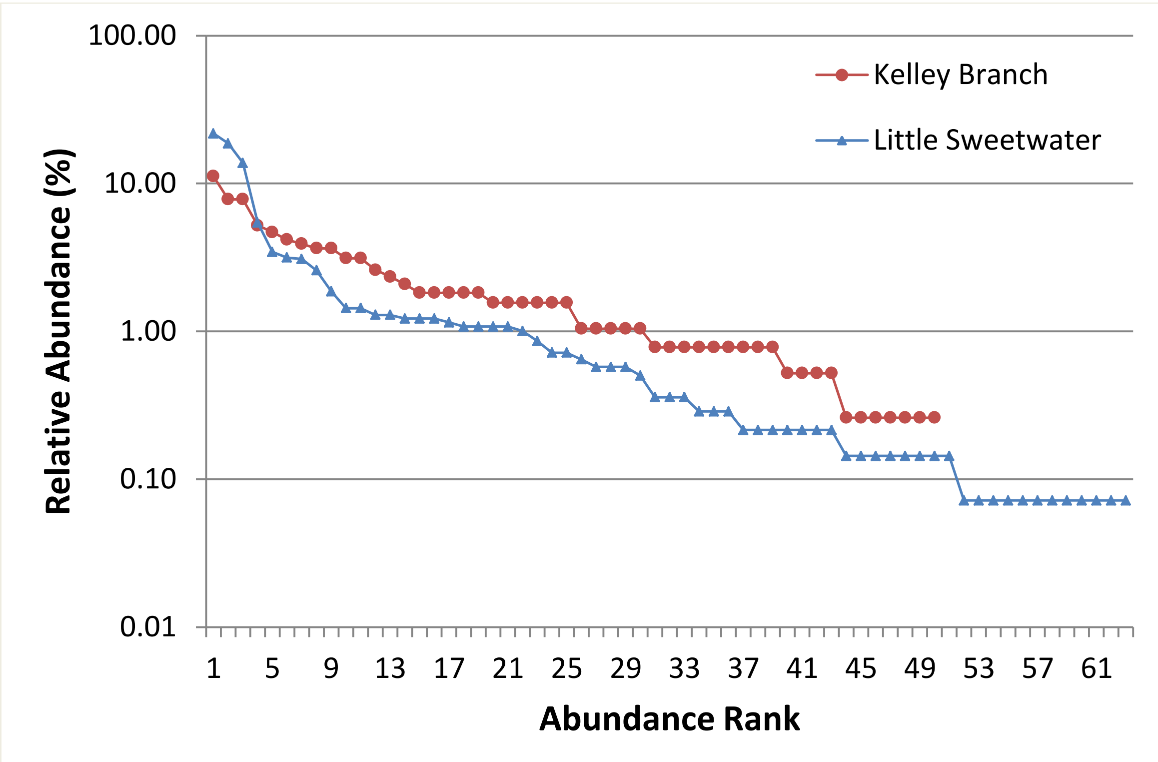

Figure 21. Adult Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxon abundance plots from below the removed dam of the study stream, Kelley Branch, and the reference stream, Little Sweetwater Creek. Taxon abundance ranks according to those listed in Appendix B.

Overall, the KDB light-trap collection contained 50 different EPT species, whereas the LSC light-trap samples contained a total of 63 species. Thirty-six taxa were collected from both sites, representing 78% of KDB species and 57% of LSC species. Eleven of the species were hydroptilids which will not be discussed further. Other similarities included a diversity of caddisfly taxa and FFG. Noteworthy differences included the absence of several shredder taxa from KDB including *Heteroplectron americanum*, two *Lepidostoma* species, and *Leuctra*.

Mean species richness was not significantly different between the KDB and LSC light-traps (t_4= -1.29, p=0.27; Table 15). Breaking it down by order, 6 Ephemeroptera species, 1 Plecoptera species, and 43 Trichoptera taxa were recorded at KDB, and Trichoptera represented more than 96% of the total captures. LSC light-trap samples contained 5 Ephemeroptera taxa, 4 Plecoptera taxa, and 54 Trichoptera species, with Trichoptera representing 98% of the total captures. Thirty-six taxa were collected at both sites, representing 72% of KDB taxa and 57% of LSC taxa. By order, the sites shared 3 Ephemeroptera and 33 Trichoptera species in common. The individual order percentage means were not significantly different between KDB and SCB (p> 0.05; Table 15).

Table 15. Total richness of the Ephemeroptera (E), Plecoptera (P), Trichoptera (T), and order percentages from below the removed dam at the study stream, Kelley Branch, and at the reference stream, Little Sweetwater Creek, during four light-trapping sessions (May 2012 – May 2013).

EPT Measure	Kelley Branch (Removed Dam)					Little Sweetwater Creek (Reference)					t-test
	n	Mean ±SE	Min.	Max.	Total	*n*	Mean ±SE	Min.	Max.	Total	p-value
Richness	4	21.8 ±3.4	13	27	50	4	30.3 ±5.7	21	44	63	0.27
%E	4	0.15 ±0.06	0	0.27	4.2	4	0.14 ±0.04	0.06	0.24	1.3	0.89
%P	4	0.04 ±0.04	0	0.15	0.2	4	0.06 ±0.04	0	0.17	0.6	0.72
%T	4	1.39 ±0.03	1.30	1.46	95.6	4	1.41 ±0.04	1.33	1.51	98.1	0.69

Individual order percentages were calculated by dividing the number of captures from an order by the total number of captures. Percentage data were transformed using an arcsin transformation (arcsin[√%]). For percentage measures, actual percentages are displayed in Total columns, all other data shown for percentage comparisons are transformed. The statistical values from the 2-sample t-tests of the means are indicated.

Hydroptilids were the most well-represented family at KDB with 14 species reported. Leptoceridae were also well represented at both sites but were a larger portion of the EPT assemblage at KDB due to higher abundances of *Oecetis*, *Nectopsyche*, and *Triaenodes* species. Philopotamidae and Hydropsychidae were also well represented at KDB. Combined, these families only comprised 2% of the EPT light-trap samples from LSC. Philopotamidae were represented by one genus, *Chimarra*, another group of net-spinning caddisflies (Fig.20). The two species observed are common components of Florida streams, including steephead ravines although the *Chimarra* collected at LSC were less than 1% of the light-trap captures. Combined, the 10 remaining Trichoptera families comprised over 9% of the KDB light-trap samples, but the 10 other Trichoptera families collected at LSC represented less than 6% of the overall LSC light-trap samples (Fig. 20).

From the light trap samples, 613 KDB and 1,412 LSC specimens were categorized into FFG. Of the six FFG categories, only average percent FC was significantly different between KDB and LSC (t_5=3.25, p=0.02; Table 16). FC taxa were similar between the two sites but the portion of FC below the removed dam was higher due to greater abundances of *Chimarra* and *Hydropsyche* taxa.

Table 16. Summary of percentages of adult Ephemeroptera, Plecoptera, and Trichoptera representing larval functional feeding groups (FFG). Samples collected below the removed dam at the study stream, Kelley Branch (N=613), and at the reference stream, Little Sweetwater Creek (N=1,412), during four light-trapping sessions (May 2012 – May 2013).

FFG	Kelley Branch (Removed Dam)					Little Sweetwater Creek (Reference)					t-test
	n	Mean ±SE	Min.	Max.	Total	*n*	Mean ±SE	Min.	Max.	Total	p-value
GC	4	0.10 ±0.06	0	0.20	4.2	4	0.07 ±0.05	0	0.21	1.6	0.68
FC	4	0.58 ±0.07	0.45	0.74	44.2	4	0.26 ±0.07	0.05	0.36	9.1	0.02*
PR	4	0.29 ±0.11	0.10	0.55	19.2	4	0.23 ±0.06	0.04	0.32	7.6	0.61
SC	4	0.29 ±0.03	0.25	0.34	9.8	4	0.22 ±0.05	0.08	0.31	7.8	0.28
SH	4	0.20 ±0.09	0	0.43	8.8	4	0.17 ±0.04	0.10	0.29	6.4	0.80
PC	4	0.37 ±0.06	0.25	0.48	13.7	4	0.57 ±0.08	0.48	0.80	67.5	0.08

Percentage data were transformed using an arcsin transformation (arcsin[√%]).Actual percentages are displayed in Total columns, all other data shown are transformed. The statistical values from the 2-sample t-tests of the means are indicated. GC=gathering-collectors; FC=filtering-collectors; PR=predators; SC=scrapers; SH=shredders; PC=vegetative-piercers. *Indicates significant difference for $\alpha=0.05$.

3.3.5 Previous Data Comparisons

The dominant families when the dam was still intact were hydropsychids and heptageniids, widespread, tolerant caddisfly and mayfly taxa respectively. Drawdown seemed to greatly improve taxonomic richness and the overall evenness of the EPT community (Fig. 22). However, once the dam was fully removed, the community was again dominated by the same two families. In 2009, the EPT community had once again improved in regards to family evenness. In 2013, the family abundance distribution was more heavily skewed towards hydropsychids as it was before dam removal. As discussed above, the hydropsychid population below the removed dam could be higher than was expected due to organism drift from the old impoundment bed. Hydroptilids were well represented in the sample from this study but cannot be compared to previous samples due to different sampling methods. Leptocerids represented similar portions of the Kelley Branch EPT community during drawdown and in the 2009 and 2013 samples (Fig. 22). The overall KDB aquatic samples and the LSC aquatic samples also contained similar portions of leptocerids (Fig. 20).

Before dam removal began, FC were the dominant FFG at KDB due to hydropsychids, and heptageniids accounted for the SC taxa (Fig. 23). During drawdown, FFG were more evenly represented within the EPT community. In 2008, SC and FC portions increased as heptageniids and hydropsychids increased again, and in 2009, FC, GC, and PR were the dominant FFG and represented similar portions of the community. As was seen in the previous years, FFG composition closely followed EPT family composition again in 2013, with hydropsychids being the dominant FC taxa and hydroptilids representing the PC.

The aquatic and light-trap samples illustrated that the aquatic insect community below the removed dam is still in a stage of recovery. However, the previously collected data would have suggested that the EPT community was at a better point of recovery in 2009 than in 2013. Based on the observed EPT community and the increased hydropsychid abundances, the old impoundment bed likely continues to impact community structure below the removed dam via organism drift and dispersal. This in turn has also shifted the functional grouping of the community back to a FC dominated community as it was before dam removal. In addition, the lack of woody vegetation within the old impoundment bed continues to affect the function of the community because fewer SH taxa are supported due to the lack of leaf litter input upstream.

Kelley Branch Aquatic EPT Family Composition

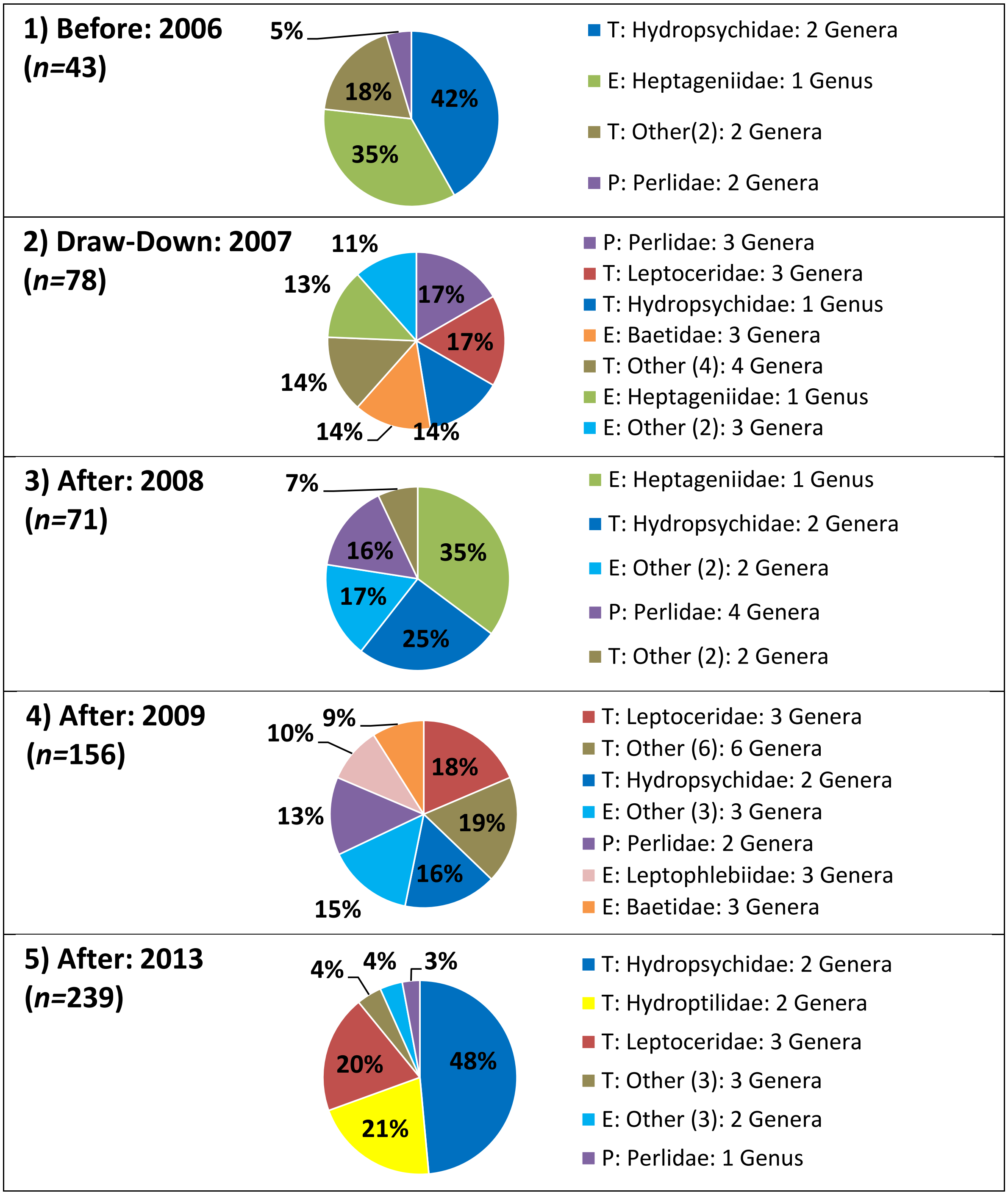

Figure 22. Percent composition (total individuals per family/total number of EPT individuals collected from the site) of Ephemeroptera, Plecoptera, and Trichoptera (EPT) families from aquatic samples collected 1) before dam removal (2006); 2) during impoundment drawdown (2007); and 3), 4), 5) after dam removal (2007, 2008, and 2013) from below the Kelley Branch dam site (Liberty Co., FL).

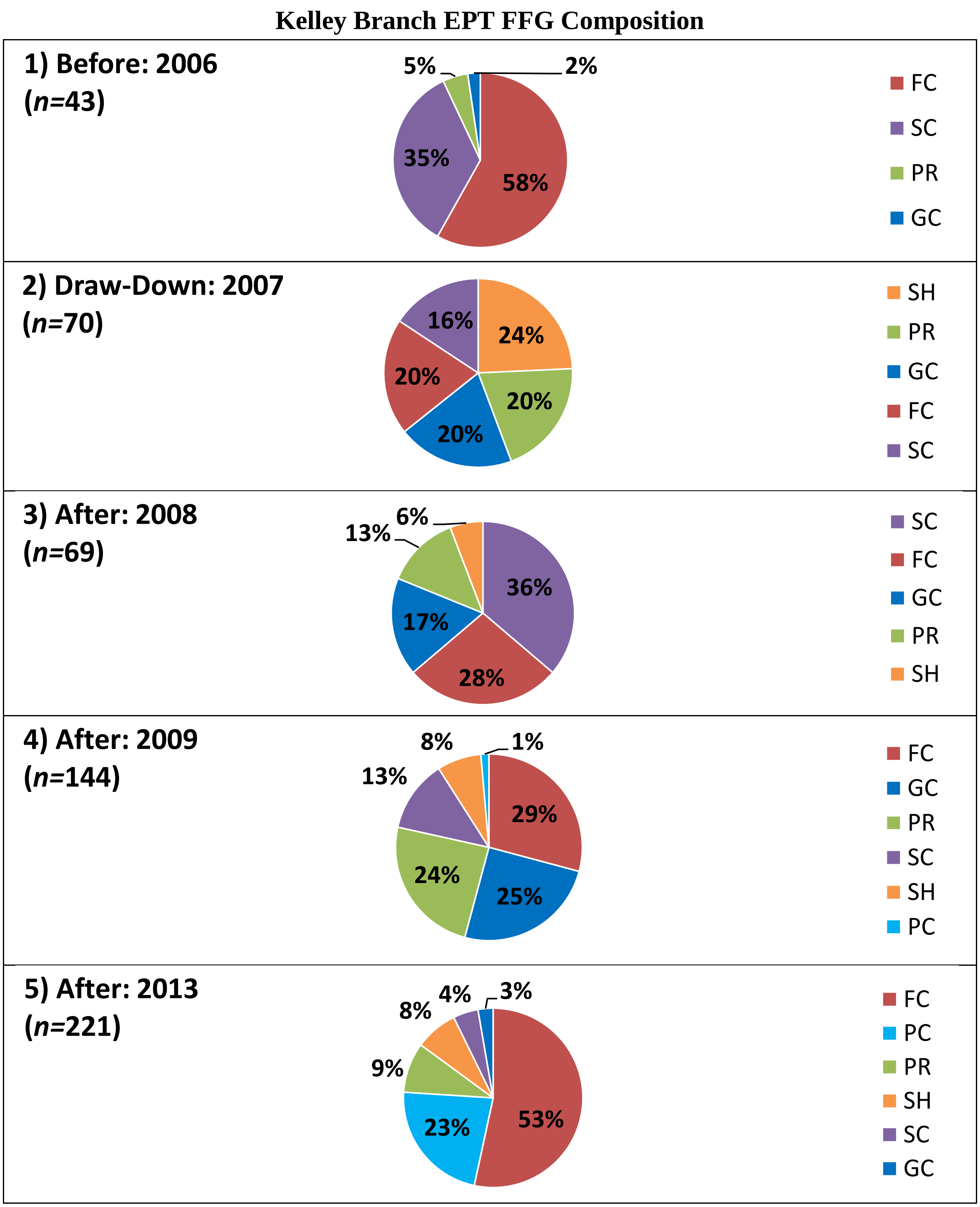

Figure 23. Functional feeding group (FFG) composition (%) of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa from aquatic samples collected 1) before dam removal (2006); 2) during impoundment drawdown (2007); and 3), 4), 5) after dam removal (2007, 2008, and 2013) from below the Kelley Branch dam site (Liberty Co., FL). GC=gathering-collector; FC=filtering-collector; PR=predator; SC=scraper; SH=shredder; PC=vegetative piercer; *n*=number of specimens.

CHAPTER FOUR

CONCLUSIONS

The aquatic insect community of Spring Canyon below the dam was structurally and functionally different from that at the reference stream due in part to the presence of the impoundment. Increased summer water temperatures resulting from impoundment is one of the primary factors limiting aquatic insect diversity and precluding species adapted to cool spring runs of steephead ravines. These changes to ravine stream habitats caused by a small dam result in an aquatic insect community characterized by low EPT and total taxonomic richness and very high abundances of tolerant taxa. Rather than exhibiting evenly distributed functional feeding groups (FFG) (outside of gathering-collectors), the SCB community was dominated by filtering-collectors (FC) while shredders (SH) were restricted. Increased amounts of drifting food material from the impoundment and minimal upstream inputs of coarse woody debris, including leaf-litter, due to dam presence have likely contributed to this shift in community functionality.

The structural aspects of the aquatic insect community below the removed dam on Kelley Branch are nearly recovered. Some taxonomic differences were still evident but overall the KDB community exhibited similar EPT, and overall taxonomic richness with a similar abundance distribution as that of the reference stream. Summer water temperatures were elevated but less so than those observed below the intact dam. Community functionality, however, was still in the process of recovery. Overall SH and FC percentages were improved over what was observed at SCB. FC percentages were not statistically significant when compared to LSC; however, when examining the previously collected data, EPT FC percentages were increased at KDB. The stream running through the old impoundment bed harbors a sizable population of aquatic macrophytes which support large numbers of EPT FC which are suspected of supplementing the community below the removed dam via drift and dispersal. In addition, the lack of woody vegetation within the old impoundment bed still restricts leaf litter availability and consequently SH production. The reference condition of FFG cannot be restored below the removed dam until the old impoundment bed is sufficiently reforested to shade out aquatic vegetation and provide allochthonous input.

Future Research

Further research of both systems would be beneficial in understanding the extent of impact and recovery. Additional collection of the chemical properties of these streams is necessary to understand what other factors may be influencing the aquatic insect communities at SCB and KDB. Dissolved oxygen is especially important, and regular data collection, as was done for temperature during this study, would be recommended to determine any possible limitations of DO in these systems. Furthermore, surveys at different distances downstream of SCB would aid in understanding how far the observed impacts of the dam extend downstream. Additional surveys downstream of KDB would be similarly useful. Also, analysis of material already collected from the stretch within the old impoundment bed would provide greater understanding of the extent of recovery occurring throughout the impacted Kelley Branch. In addition to survey work, biological and ecological studies of aquatic taxa that are still poorly understood would be very beneficial for understanding taxonomic presence/absence or abundances.

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APPENDICES

Appendix A.

Aquatic insect taxa composition for below the dam of the impounded stream, Spring Canyon (SCB), the reference stream, Little Sweetwater Creek (LSC), and below the removed dam at the study site, Kelley Branch (KDB). Abundances are given as the percent of total specimens identified (SCB=12,526; LSC=1,712; KDB=3,806).

Spring Canyon (Impounded)		**Little Sweetwater Creek (Reference Stream)**		**Kelley Branch (Removed Dam)**	
Taxa (*n*=32)	**%**	**Taxa (*n*=42)**	**%**	**Taxa (*n*=43)**	**%**
Chironomidae	63.08	Chironomidae	62.03	Chironomidae	53.52
Hydropsychidae *Hydropsyche*	10.54	Empididae	4.50	Elmidae *Microcylloepus*	11.32
Hydropsychidae *Cheumatopsyche*	5.56	Calamoceratidae *Anisocentropus*	4.03	Hydropsychidae *Hydropsyche*	9.04
Leptoceridae *Oecetis*	5.35	Hydropsychidae *Hydropsyche*	3.45	Empididae	3.05
Elmidae *Microcylloepus*	5.33	Perlidae *Acroneuria*	3.39	Leptoceridae *Triaenodes*	2.71
Elmidae	3.06	Elmidae *Stenelmis*	2.51	Psychomyiidae *Lype*	2.52
Elmidae *Stenelmis*	2.98	Hydropsychidae *Diplectrona*	2.51	Leptoceridae *Nectopsyche*	2.23
Simuliidae	1.15	Leptoceridae *Triaenodes*	1.81	Elmidae *Stenelmis*	1.94
Leptoceridae *Ceraclea*	0.85	Leptoceridae *Oecetis*	1.75	Simuliidae	1.63
Ceratopogonidae	0.57	Elmidae *Gonielmis*	1.40	Philopotamidae *Chimarra*	1.50
Coenagrionidae *Argia*	0.42	Ceratopogonidae	1.34	Elmidae *Gonielmis*	1.45
Leuctridae *Leuctra*	0.18	Psychomyiidae *Lype*	1.29	Leptoceridae *Oecetis*	1.39
Corydalidae *Nigronia*	0.18	Simuliidae	1.11	Hydroptilidae *Oxyethira*	1.34
Empididae	0.17	Ephemerellidae *Teloganopsis*	0.99	Hydropsychidae *Diplectrona*	0.76
Hydroptilidae *Orthotrichia*	0.11	Psephenidae *Ectopria*	0.88	Heptageniidae *Maccaffertium*	0.58
Leptoceridae *Triaenodes*	0.10	Tipulidae	0.82	Hydropsychidae *Cheumatopsyche*	0.53
Coenagrionidae	0.07	Philopotamidae	0.76	Elmidae *Ancyronyx*	0.42

Enallagma		*Chimarra*			
Tabanidae	0.06	Brachycentridae *Brachycentrus*	0.70	Hydroptilidae *Hydroptila*	0.39
Collembola	0.05	Leptoceridae *Nectopsyche*	0.64	Perlidae *Perlesta*	0.37
Tipulidae	0.05	Heptageniidae *Maccaffertium*	0.41	Coenagrionidae *Argia*	0.34
Coenagrionidae *Nehalennia*	0.05	Leuctridae *Leuctra*	0.41	Ceratopogonidae	0.32
Libellulidae *Neurocordulia*	0.02	Lepidostomatidae *Lepidostoma*	0.35	Leptophlebiidae *Habrophlebiodes*	0.32
Curculionidae	0.01	Collembola	0.29	Collembola	0.29
Elmidae *Gonielmis*	0.01	Leptophlebiidae *Habrophlebiodes*	0.29	Libellulidae *Neurocordulia*	0.26
Gyrinidae *Dineutus*	0.01	Veliidae *Rhagovelia*	0.29	Leuctridae *Leuctra*	0.24
Phoridae	0.01	Polycentropodidae *Nyctiophylax*	0.29	Ephemerellidae *Eurylophella*	0.21
Mesoveliidae *Mesovelia*	0.01	Libellulidae *Neurocordulia*	0.23	Tipulidae	0.13
Nepidae *Ranatra*	0.01	Calamoceratidae *Heteroplectron*	0.23	Leptophlebiidae *Paraleptophlebia*	0.13
Calopterygidae *Calopteryx*	0.01	Elmidae *Microcylloepus*	0.18	Veliidae *Rhagovelia*	0.13
Calopterygidae *Hetaerina*	0.01	Ephemerellidae *Eurylophella*	0.18	Tabanidae	0.11
Odontoceridae *Psilotreta*	0.01	Dixidae	0.12	Calopterygidae *Calopteryx*	0.11
Psychomyiidae *Lype*	0.01	Corydalidae *Nigronia*	0.12	Perlidae *Acroneuria*	0.11
		Gomphidae *Progomphus*	0.12	Limnephilidae *Pycnopsyche*	0.11
		Hydroptilidae *Hydroptila*	0.12	Baetidae	0.08
		Elmidae *Ancyronyx*	0.06	Gomphidae *Progomphus*	0.08
		Leptohyphidae	0.06	Calamoceratidae *Anisocentropus*	0.08
		Leptophlebiidae *Paraleptophlebia*	0.06	Hydroptilidae *Mayatrichia*	0.08
		Corydalidae *Chauliodes*	0.06	Psephenidae *Ectopria*	0.05
		Calopterygidae *Calopteryx*	0.06	Coenagrionidae *Enallagma*	0.05

	Cordulegasteridae *Cordulegaster* 0.06	Calopterygidae *Hetaerina* 0.03
	Dipseudopsidae *Phylocentropus* 0.06	Coenagrionidae *Nehalennia* 0.03
	Hydroptilidae *Mayatrichia* 0.06	Perlidae *Paragnetina* 0.03
		Polycentropodidae *Nyctiophylax* 0.03

Appendix B.

Adult Ephemeroptera, Plecoptera, and Trichoptera (EPT) composition below the dam at the impounded stream, Spring Canyon (SCB), the reference stream, Little Sweetwater Creek (LSC), and from below the removed dam at the study stream, Kelley Branch (KDB), Abundances are given as the percent of total specimens identified to species (SCB=3,371; KDB=384; LSC=1,394).

Spring Canyon (Below Dam)		Little Sweetwater Creek (Reference Stream)		Kelley Branch (Removed Dam)	
Species (*n*=56)	**%**	**Species (*n*=63)**	**%**	**Species (*n*=50)**	**%**
Cheumatopsyche analis	22.34	*Hydroptila quinola*	21.81	*Oecetis sphyra*	11.20
Oecetis inconspicua	15.63	*Hydroptila armata*	18.65	*Nectopsyche candida*	7.81
Ceraclea maculata	15.49	*Oxyethira janella*	13.77	*Hydropsyche elissoma*	7.81
Hydropsyche decalda	9.23	*Oxyethira novasota*	5.45	*Oxyethira novasota*	5.21
Hydroptila armata	7.68	*Agarodes libalis*	3.44	*Maccaffertium smithae*	4.69
Oecetis georgia	7.33	*Oxyethira maya*	3.16	*Chimarra aterrima*	4.17
Cheumatopsyche virginica	3.86	*Hydroptila waubesiana*	3.08	*Hydroptila armata*	3.91
Oecetis osteni	3.68	*Phylocentropus carolinus*	2.58	*Chimarra falculata*	3.65
Orthotrichia aegerfasciella	2.64	*Neotrichia vibrans*	1.87	*Agarodes libalis*	3.65
Oxyethira lumosa	2.34	*Neotrichia armitagei*	1.43	*Nectopsyche pavida*	3.13
Oecetis sphyra	1.57	*Oecetis inconspicua*	1.43	*Oxyethira abacatia*	3.13
Agarodes libalis	1.04	*Acroneuria lycorias*	1.29	*Neotrichia armitagei*	2.60
Acroneuria lycorias	1.01	*Oecetis sphyra*	1.29	*Phylocentropus carolinus*	2.34

Oxyethira janella	1.01	*Maccaffertium smithae*	1.22	*Orthotrichia aegerfasciella*	2.08
Oxyethira maya	0.62	*Phylocentropus lucidus*	1.22	*Molanna blenda*	1.82
Triaenodes milnei	0.53	*Diplectrona modesta*	1.22	*Triaenodes ignitus*	1.82
Cernotina spicata	0.44	*Orthotrichia aegerfasciella*	1.15	*Ceraclea maculata*	1.82
Anisocentropus pyraloides	0.24	*Hexagenia limbata*	1.08	*Lype diversa*	1.82
Oxyethira novasota	0.21	*Anisocentropus pyraloides*	1.08	*Polycentropus sp.*	1.82
Perlesta sp.	0.21	*Nectopsyche candida*	1.08	*Hydroptila quinola*	1.56
Hydroptila quinola	0.18	*Lype diversa*	1.08	*Hexagenia limbata*	1.56
Nectopsyche candida	0.18	*Chimarra falculata*	1.00	*Oxyethira elerobi*	1.56
Maccaffertium smithae	0.15	*Psilotreta frontalis*	0.86	*Oxyethira lumosa*	1.56
Diplectrona modesta	0.15	*Lepidostoma griseum*	0.72	*Psilotreta frontalis*	1.56
Neotrichia minutisimella	0.15	*Rhyacophila carolina*	0.72	*Agarodes crassicornis*	1.56
Orthotrichia instabilis	0.15	*Neotrichia minutisimella*	0.65	*Anisocentropus pyraloides*	1.04
Oecetis cinerascens	0.15	*Oxyethira glasa*	0.57	*Cheumatopsyche pinaca*	1.04
Psilotreta frontalis	0.15	*Nectopsyche pavida*	0.57	*Molanna tryphena*	1.04
Phylocentropus lucidus	0.12	*Pycnopsyche antica*	0.57	*Pycnopsyche antica*	1.04
Cheumatopsyche pinaca	0.12	*Oecetis ditissa*	0.50	*Diplectrona modesta*	1.04
Hydroptila waubesiana	0.12	*Hydropsyche incommoda*	0.36	*Maccaffertium exiguum*	0.78
Neotrichia armitagei	0.12	*Nyctiophylax morsei*	0.36	*Habrophlebiodes brunneipennis*	0.78
Orthotrichia curta	0.12	*Labiobaetis sp.*	0.36	*Mayatrichia ayama*	0.78
Triaenodes ignitus	0.12	*Oxyethira abacatia*	0.29	*Oxyethira maya*	0.78
Hydroptila remita	0.09	*Chimarra aterrima*	0.29	*Oxyethira pallida*	0.78
Hydroptila sykorai	0.09	*Stenacron sp.*	0.29	*Oecetis georgia*	0.78
Lype diversa	0.09	*Heteroplectron americanum*	0.22	*Oecetis inconspicua*	0.78
Caenis maccafferti	0.06	*Cheumatopsyche*	0.22	*Labiobaetis*	0.78

	analis	*frondalis*
Hydropsyche elissoma 0.06	*Hydropsyche rossi* 0.22	*Lepidostoma sp.* 0.78
Orthotrichia baldufi 0.06	*Macrostemum carolina* 0.22	*Perlinella drymo* 0.52
Oecetis persimilis 0.06	*Oecetis nocturna* 0.22	*Hydroptila waubesiana* 0.52
Eurylophella doris 0.03	*Molanna blenda* 0.22	*Oxyethira janella* 0.52
Maccaffertium exiguum 0.03	*Chimarra florida* 0.22	*Rhyacophila carolina* 0.52
Phylocentropus placidus 0.03	*Paraleptophlebia volitans* 0.14	*Paraleptophlebia volitans* 0.26
Hydroptila novicola 0.03	*Cheumatopsyche edista* 0.14	*Phylocentropus placidus* 0.26
Lepidostoma latipenne 0.03	*Oxyethira lumosa* 0.14	*Oecetis cinerascens* 0.26
Nectopsyche pavida 0.03	*Oxyethira verna* 0.14	*Hydroptila remita* 0.26
Oecetis ditissa 0.03	*Leptocerus americanus* 0.14	*Phylocentropus lucidus* 0.26
Pycnopsyche antica 0.03	*Oecetis georgia* 0.14	*Neureclipsis sp.* 0.26
Molanna blenda 0.03	*Triaenodes ignitus* 0.14	*Nyctiophylax sp.* 0.26
Chimarra aterrima 0.03	*Molanna tryphena* 0.14	
Chimarra florida 0.03	*Eccoptura xanthenes* 0.07	
Ptilostomis postica 0.03	*Neoperla carlsoni* 0.07	
Cernotina calcea 0.03	*Cheumatopsyche pinaca* 0.07	
Neureclipsis crepuscularis 0.03	*Hydropsyche elissoma* 0.07	
Leuctra sp. 0.03	*Mayatrichia ayama* 0.07	
	Orthotrichia instabilis 0.07	
	Oxyethira zeronia 0.07	
	Lepidostoma latipenne 0.07	
	Ceraclea maculata 0.07	
	Nyctiophylax serratus 0.07	
	Polycentropus cinereus 0.07	
	Leuctra sp. 0.07	