QUANTIFICATION AND ECOLOGICAL ROLE OF SNAG HABITAT IN THE APALACHICOLA RIVER, FLORIDA

By

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To my parents, George and Linda Burgess

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LIST OF ABBREVIATIONS

ACF	Apalachicola-Chattahoochee-Flint
ACFRBC	Apalachicola-Chattahoochee-Flint River Basin Compact
ACT	Alabama-Coosa-Tallapoosa
cfs	Cubic feet per second
cms	Cubic meters per second
DEP	Department of Environmental Protection
FFWCC	Florida Fish and Wildlife Conservation Commission
ft	Feet
GPS	Global positioning system
IFIM	Instream Flow Incremental Methodology
km	Kilometer
LWD	Large woody debris
m	Meter
MB	Megabyte
MOA	Memorandum of Agreement
NGVD 29	National Geodetic Vertical Datum of 1929
NOAA	National Oceanic and Atmospheric Administration
NWFWMD	Northwest Florida Water Management District
PHABSIM	Physical Habitat Simulation
PVC	Polyvinyl chloride
RM	River mile
TNC	The Nature Conservancy
UNESCO	United Nations Educational, Scientific, and Cultural Organization
US	United States

USACE	United States Army Corps of Engineers
USCB	United States Census Bureau
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
WGS 84	World Geodetic System of 1984

Abstract of Thesis Presented to the Graduate School
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The Apalachicola River has the greatest discharge of any river in the State of Florida, and has one of the largest forested floodplains of any river in the southeastern US. Rapid urbanization of the upper ACF basin, as well as several extensive droughts in the region have resulted in increased demands for water withdrawals and flow regulation for municipal, commercial, and agricultural uses throughout the basin. This has positioned the states of Florida, Georgia, and Alabama, along with several federal agencies, in a surface water allocation and management conflict. Of particular concern are minimum flows and levels required to maintain healthy biotic communities during periods of regional drought, when water demands are at their highest, and the resource is most limited.

Very low flows confine the primarily sandy-bottomed Apalachicola River to its main channel, where snags serve as the only stable structural habitat for the riverine fauna. This study aimed to assess the ecological importance, and quantify the amount of snags submerged in the Apalachicola River as a result of variations in flow regime. Additionally, the distributions of snags based on river-bank habitat and geomorphological characteristics were also examined to assess their role in snag habitat availability. Traditional aquatic snag quantification methods

were not feasible in the exceedingly turbid, high-order Coastal Plain river; therefore, a novel method using digital imaging analysis techniques was developed for analysis.

Area and percentage of available snag habitat submerged were analyzed at 0.5-m elevations above water level for 60 sampling stations along the Apalachicola River at a relatively-low river discharge. Snag area values generated using the novel methods presented in this study were credible for steep sloping bank, gentle sloping bank and dike field habitat types, but were erroneously-elevated for sandbar stations. Snag quantities of non-sandbar stations were found to be greatest within the first 1.0 m of exposed bank for nearly all sites surveyed, while steep sloping banks typically exhibited the most uniform vertical snag distribution. In most cases, geomorphic classifications demonstrated no considerable effect on non-sandbar habitat snag distribution; however, the outer banks of river bends exhibited a more balanced vertical distribution of snags than inner banks of river bends or along the banks of river straightaways. Limitations and proposed modifications to the digital imaging methods are discussed, and recommendations are provided in support of increased water flows for ecological sustainability in the Apalachicola River.

CHAPTER 1
INTRODUCTION AND OBJECTIVES

The Apalachicola River is the State of Florida's largest river by discharge, and has one of the most extensively forested floodplains of any river in the southeastern Coastal Plain. Home to a diverse fauna, including many threatened and endangered species, the basin is a significant natural resource worthy of conservation efforts. The Apalachicola River and Bay system is remarkable in that it has remained relatively intact as urbanization has transformed comparable riverine systems within North America. Therefore, conservation and rehabilitation efforts that are enacted in the Apalachicola River are currently proactive, rather than reactive; consequently, such efforts tend to be less expensive and have an increased likelihood of success over time. In light of several major droughts, concerns have arisen over declining water availability in the river, as well as decreasing freshwater inputs into the Apalachicola Bay estuary, due to rapid population growth and accelerated water consumption throughout the greater ACF basin. Regulated flows and competing water withdrawals have generated disputes over states' water rights, as well as management of federal reservoirs within the region. This has positioned the states of Florida, Georgia, and Alabama in a water allocation conflict, the eventual consequences of which may jeopardize biological communities that are dependent on the annual flow cycles of the rivers. Of particular concern are low-flow conditions and habitat availability during critical phases in the ecology of river-dependent biota.

Numerous local, state, and federal agencies are responsible for management and optimization of water resources within the 50,764-km^2 ACF basin; consequently, water-allocation disputes within the region have been contentious. As the metropolitan Atlanta area continues to grow and water demands multiply, Georgia-based interest groups have endeavored to store as much surface water as possible in upstream Chattahoochee River reservoirs.

Maintaining equity of water use among consumers in the upper reaches of the ACF basin with the ecological needs and consumptive use requirements of downstream users has been challenging. Determining the quantities of reasonable and beneficial river water use varies greatly among stakeholders depending upon their location within the ACF basin. In watersheds throughout the world, water managers are beginning to implement adaptive management techniques to resolve conflicts similar to those exhibited in the ACF basin. Successful adaptive management is achieved by collecting baseline data of critical components, monitoring deviations over time, and making sensible modifications to management plans as environmental conditions change (Richter et al. 2003). The monitoring and documentation of rehabilitation efforts in fluvial systems is critical to the success of adaptive management strategies (Bash and Ryan 2002).

With habitat loss considered a primary threat to imperiled species, critical habitats within fluvial ecosystems have been targeted as goals for conservation. In the context of rivers, snags are defined as any woody structures, regardless of size, located within the banks of a watercourse. During low water flows, the primarily sandy-bottomed Apalachicola River becomes almost completely confined to its main channel, and as a result, snags provide the only stable structural habitat and cover available for its river-based fauna. For years, aquatic ecologists have recognized the importance of snag habitat in maintaining productive ecosystems. However, it was not until fairly recently that focused efforts were made to quantify riverine snag habitat. Due to the dynamic nature of a river system, it is difficult to predict rates of wood input and depletion as they vary over space and time. Wallace and Benke (1984) stated that the quantification of riverine wood habitat is essential in assessing past or potential impacts of snag removal on ecosystem processes. Furthermore, determining the amount of available snag material and its ecological role is vital in understanding how anthropogenic alterations to fluvial systems influence ecosystem success and dynamics. In Florida, very little research has been performed that quantifies the loss of habitat for aquatic fauna when snag material is removed from the system, or when woody material is exposed during decreasing river discharges. As minimum flows and levels are discussed within the management of the ACF basin, it is imperative to know how changes in flow affect snag habitat availability. Scientists in various disciplines have been requested to collect data and provide input to assist officials in making informed decisions concerning the regulation and management of water resources within the ACF basin. This study endeavored to provide insight into the quantity of aquatic snags submerged in the Apalachicola River at various flow regimes using a novel digital imaging analysis technique. In addition, the distribution of snags based on river bank habitat and geomorphological characteristics was also examined to indicate their prospective influence on snag habitat availability within the Apalachicola River.

CHAPTER 2
APALACHICOLA RIVER, ACF BASIN, AND THE TRI-STATE WATER CONFLICT

Apalachicola River and ACF Basin

Two large rivers in the southeastern US, the Chattahoochee and the Flint, converge to form Lake Seminole, a shallow 152-km^2 man-made reservoir in southeastern Georgia (Figure 2-1). The Apalachicola River headwaters are created by the tailwater discharge of Lake Seminole at the Jim Woodruff Lock and Dam; a USACE operated navigation and hydroelectric generation facility that was completed in 1954. The 805-km long ACF river system drains 50,764 km^2 in the states of Florida, Georgia and Alabama, with approximately 13% of the total ACF basin contained within the State of Florida (Light et al. 2006).

The Apalachicola River is the largest river in terms of mean annual discharge in Florida, and has the fourth largest discharge of rivers in the southeastern US (Leitman et al. 1991). Located in the Florida panhandle, the Apalachicola River basin drains 6,216 km^2 of land within the state, of which about half is drained by the spring-fed Chipola River, its largest tributary (Leitman et al. 1983). The Apalachicola River is a moderately sinuous, highly turbid, alluvial bed Coastal Plain river that flows 171 km from the Florida-Georgia state line to Apalachicola Bay (Light et al. 1998). The river has a relief of 12.5 m from its headwaters at the Jim Woodruff Lock and Dam to its convergence with the Gulf of Mexico; relatively low for a river of its length (Meeter et al. 1979). Light et al. (2006) reported that the average Apalachicola River discharge from 1929-2004 at the USGS gage near Chattahoochee, Florida was 21,900 cfs; however, flows were greatly influenced by upstream water management practices in the states of Alabama and Georgia, as well as seasonal variations in precipitation. The lowest daily discharge at the Apalachicola River, Chattahoochee gage was 3,900 cfs during a severe drought in November 1987; the highest recorded flow was 291,000 cfs, which occurred during a historic flood in

March 1929 (Light et al. 2006). Average annual precipitation in the Apalachicola River basin is 1.42 m, with the driest period typically occurring in the months from September through November (Leitman et al. 1991).

Unlike most large-river basins of the southeastern US, the floodplain of the Apalachicola River lacks a major urban center, and is consequently sparsely populated. Land uses along the river are primarily forestry or conservation lands, which contributes to the scarcity of development in the region (Leitman et al. 1991). Additional factors contributing to the rural nature of the Apalachicola River basin include the relatively depressed economic status of the area, and limited ability for cross-river transportation and exchange of goods, as only four highway bridges traverse the river along its 171-km length.

The extensive 454-km^2 Apalachicola River floodplain is the largest of all rivers in Florida, and is abundantly forested by many species of cypress, tupelo, and other mixed-bottomland hardwood tree species, of which most have been logged several times since the year 1870 (Leitman et al. 1983; Elder et al. 1988). The ample forests of the Apalachicola River floodplain deposit approximately 3.6×10^5 metric tons of organic material onto the floodplain annually (Elder and Cairns 1982). With one of the highest computed litter fall rates of any temperate forest, the Apalachicola River ecosystem is reliant on seasonal floods to mobilize, distribute, and recycle the large quantities of organic material and nutrients that accumulate on its floodplain.

The habitats in and along the Apalachicola River basin contain one of the most species-rich assemblages of flora and fauna in the southeastern US (Livingston 1984; Leitman et al. 1991; Mitsch and Gosselink 2000). An estimated 1,500 plant species, 450 vertebrate species, and a myriad of invertebrate taxa, including the largest number of freshwater mollusk species in the US, inhabit the Apalachicola River watershed (Heard 1977; Means 1977; Leitman 1984).

Additionally, many endemic and federally listed endangered or threatened species are found in the basin, making it one of the most significant ecosystems remaining in the US (Myers and Ewel 1990).

Water Use Within the ACF Basin

Both surface and groundwater resources are used by consumers within the ACF basin. Surface water is used for domestic and commercial supply, cooling of power plants, hydropower generation, transportation, irrigation, and recreation. The largest urban center in the basin, the greater Atlanta metropolitan area, draws nearly all of its water supply from the upper reaches of the Chattahoochee River at Lake Lanier, with a portion of the water returning to the basin as treated wastewater. As the city of Atlanta continues to grow, most models indicate that another source of water will need to be tapped in the near future, or extreme conservation efforts will need to be enacted in order to maintain the existing growth rate of the urban area. The Georgia DEP asserts that its projections for future water supply, relying on the Chattahoochee River as a primary water source, are still sound, while the USACE indicates that the city has already exploited surface water quantities that equal, or at times have exceeded projections established for the year 2030 (Seabrook 2002). Discussions of alternative water sources for the Atlanta metropolitan area have included the proposed construction of several water retention reservoirs on the upper Flint River, as well as the possibility of interbasin water transfers from the ACT, or Tennessee River watersheds (Hoehn 2001). These alternatives have failed to materialize due to intense objections from a variety of political, environmental, and economic interests.

Groundwater resources in the ACF basin are primarily used for self-supply domestic drinking wells, industrial and commercial applications, and for center-pivot irrigation of agriculture. Large quantities of land in rural southwest Georgia are cultivated for ornamental, vegetable, fruit and nut, forage, and row-crop agriculture. These large farms are dependent on groundwater from the Chattahoochee and Flint river watersheds for irrigation, and ultimately for economic subsistence. In addition, portions of the extensive Floridan Aquifer are recharged within the ACF basin, therefore intimately linking surface and groundwater systems (Elder et al. 1988).

Modifications Within the ACF Basin

Over the past 200 years, various navigational enhancements have been made in the ACF basin to facilitate the movement of commodities on the river. In the 1820's, the first steamboats began to transport row crops such as corn, cotton, peanuts, and soybeans from cities in Alabama and Georgia to the seaport of Apalachicola, Florida (Kirkland 2001). Mueller (1990) indicates that these early steamers would often encounter sections of the rivers where trees and snags traversed the entire waterway and would have to be displaced to allow boat passage. From 1875 to 1998, selective de-snagging operations in the Apalachicola and lower Chattahoochee rivers had removed approximately 334,000 woody obstructions including snags, stumps, bushes, overhanging trees, and drift logs (T. Hoehn, FFWCC, and S. Leitman, NWFWMD, unpublished data). Ager et al. (1985) indicated that snag removal associated with navigational improvements has left much of the main channel Apalachicola River with a rather unproductive sandy bottom devoid of critical habitat. Since 1956, when selective desnagging was most prevalent, the main channel of the Apalachicola River has actually become shallower and wider, due in part to removal of riparian snags (H. Light, USGS Tallahassee, personal communication). A total of 16 dams and associated reservoirs (13 on the Chattahoochee River; 3 on the Flint River) have been constructed and maintained upstream of the Apalachicola River for water storage, recreation, flood mitigation, navigation, and hydroelectricity generation (Ager et al. 1985). In 1946, as part of the River and Harbors Act of the previous year, the US Congress authorized the USACE Mobile District to dredge and de-snag a 2.75-m x 30.5-m navigation channel through a large portion of the ACF basin to encourage economic development and promote interstate commerce (Leitman 2005). The constructed navigation channel traversed the entire length of the Apalachicola River, and extended up to Columbus, Georgia and Phoenix City, Alabama on the Chattahoochee River, and Bainbridge, Georgia on the Flint River. A plethora of problems emerged following the completion of the watercourse, but the disposal of large quantities of dredge spoil at unprotected floodplain easement sites within the banks of the rivers, and the inability to maintain a mandated 2.75-m depth for barge traffic 95% of the year were critical shortfalls revealed over time (Leitman et al. 1991). During the years of operation and maintenance of navigational conditions within the ACF basin, the quantity of barge traffic failed to meet the utility in which it had been designed and constructed, due primarily to inconsistencies in channel reliability. Consequently, the majority of freight (sand, gravel, petroleum, and fertilizer) that was routinely transported by barges in other areas of the US ultimately was transported by truck or rail within the ACF basin. Leitman et al. (1991) indicated that in the late 1980's, the unit cost of maintaining the ACF waterway per ton-mile of freight shipped was the highest of any waterway in the US, and more than 40 times the national average. In 2003, nine barges used the ACF waterway to transport freight, while it was estimated that US$6.5 million dollars in tax revenue was spent to maintain the channel (Leitman 2005). Navigational channel maintenance in the Apalachicola River was carried out by the USACE until 2005, when the Florida DEP utilized its authority under the Clean Water Act to rescind the annual river dredging permit.

In addition to the extensive dredging and de-snagging operations implemented to facilitate river transportation, another modification called 'navigation windows' were used on the Apalachicola River beginning in the year 1990. During a navigation window, the USACE would intermittently discharge large quantities of water from upstream reservoirs for a temporary ten-day, to two-week period to allow scheduled barge traffic movement during low-water conditions (Leitman et al. 1991). Navigation windows unfortunately altered the natural chemical, physical, and biological components of the floodplain habitat and its ecology during what were normally low-flow river conditions. Another issue with navigation windows was the dramatic vertical drop in river stage over a short period of time, or 'fall rate', which would occur at the end of a scheduled navigation window. This was particularly evident in April 2000, when a navigation window decimated fish populations in the Apalachicola River by stranding millions of reproducing fish and invertebrates in desiccating pools on the floodplain (Hoehn 2001). In addition, residents with property on upstream reservoirs, especially those on Lake Lanier, objected to navigation windows that visibly lowered water levels, consequently leaving their recreational docks high and dry. The practice of navigation windows was effectively discontinued with the revocation of the Florida DEP dredging permit in 2005, and the cessation of commercial navigation.

Other Apalachicola River modifications implemented for enhanced water conveyance and improved river transportation included: 1) the creation of several bend easings and meander cutoffs to straighten navigationally difficult natural bends in the river; 2) the relocation and removal of rocky shoals in portions of the upper river; 3) the periodic release of water stores for flood control and reservoir management; and 4) the installation of river-training dikes to augment channel bed scour and increase downstream sediment transport (Leitman et al. 1991; Light et al. 1998).

Tri-State 'Water Wars' Conflict

As the human population continues to grow, there are increased demands for natural resources with limited availability. One of the greatest public concerns is the accessibility to freshwater resources, particularly in areas that have already exceeded their capacity to support growth. Officials at multiple levels of government have been assigned the task of maintaining adequate water stores for natural and consumptive uses, as well as planning for future water demands. Due to precipitation generated by the warm moist air from the Gulf of Mexico and Atlantic Ocean, southeastern US states have historically had sufficient water resources to meet the requirements of their growing populations (Leitman et al. 1991). However, concerns have arisen recently in regards to adequate freshwater availability during seasonal droughts, and as rates of urbanization continue to increase throughout the region. Two substantial droughts occurred in the southeastern US during the 1980's that generated significant consideration to the limits of water availability in the ACF basin, and triggered the states to become increasingly acrimonious in regards to their water resources. This has placed the states of Alabama, Georgia, and Florida in a surface water allocation conflict, the eventual consequences of which may jeopardize the ecology of the biota in the ACF basin, as well as in the extensive Apalachicola Bay estuary.

In the US, the states have the authority to regulate the use of freshwater within their borders, with the exception of waterways used for navigation, and federally constructed reservoirs. Each state has established its own water laws and mechanisms for modification of water regulations; however, congressional approval is required for any change in interstate water resource allocation. When the US Congress authorized the development of the ACF waterway for navigation, hydropower, and flood control in 1946, they fundamentally removed state control of surface water in the basin in favor of federal management. As with any water allocation and management decision in the US, an array of federal laws and regulations must be adhered to by the states. While not entirely comprehensive, some of the principal legislative acts pertinent to water management in the ACF basin include: the Rivers and Harbors Acts of 1945 and 1946, the Rivers and Harbors Flood Control Act of 1944, 1958, and 1962, the Water Resources Act, the Federal Power Act, the Clean Water Act, the Water Quality Control Act, the National Environmental Policy Act, the Endangered Species Act, the Migratory Bird Treaty Act, the Marine Mammal Protection Act, the Fish and Wildlife Coordination Act, and the Coastal Zone Management Act (Sherk 2005).

The ACF basin is one of the largest freshwater contributors into the Gulf of Mexico (Livingston 1984). The inputs from the Apalachicola River into Apalachicola Bay deliver significant quantities of valuable detritus and nutrients, as well as beneficially reduce the salinity of the estuary, therefore furnishing an ideal environment for shellfish, shrimp, and crabs. Inputs from the river also control the distribution of essential estuarine habitats including seagrass beds, mangrove stands, tidal marshes, and mud flats necessary for productive estuaries (Myers and Ewel 1990). Coastal cities in Franklin and Gulf counties near the mouth of the Apalachicola River rely heavily on the commercial fisheries of Apalachicola Bay for their existence and livelihood. Apalachicola Bay, a 549-km^2 lagoon-and-barrier island complex, is regarded as one of the most productive and least polluted estuaries in the US, and generates approximately 90% of Florida’s oysters, making up about 10% of the United States’ annual harvest (Livingston 1984; Ruhl 2005; Wang et al. 2008). The salinity pulse generated by the seasonal variation in river flows is critical to the health of oyster populations in the bay (Livingston et al. 2000). In addition, Apalachicola Bay serves as an important nursery area for many marine species, and produces Florida’s third largest shrimp fishery, as well as a robust blue crab fishery (Oesterling and Evink 1977). The Apalachicola Bay estuary has been recognized as a water body worthy of special protection, and is therefore designated as a UNESCO ‘International Biosphere Reserve’, a NOAA ‘National Estuarine Research Reserve’, a Florida DEP ‘Aquatic Preserve’ and ‘Outstanding Florida Water’.

Analysis of time spent participating in outdoor activities within the US revealed that Florida trailed only Texas in time devoted to wildlife-related recreation (Struhs and Mainella, 2000). A report by Southwick and Allen (2003) indicated that in 2001, over 40% of Floridians participated in wildlife-related activities, and in addition, over 800,000 visitors came to the state specifically to partake in wildlife related recreation. The Apalachicola River basin is of tremendous economic importance to Florida, and its floodplain and environs provide abundant opportunity for outdoor recreational activities such as fishing, hunting, camping, and hiking. The FFWCC (2008) boasts the state as being the ‘Fishing Capital of the World’, emphasizing the values of both recreational and commercial fishing to the state economy. A combined report by the USFWS and USCB (2002) estimated that in 2001, freshwater fishing in Florida generated US$1.057 billion in retail sales alone. Decreased water flow in the Apalachicola River severely affects fisheries in the river, as well as in Apalachicola Bay, two highly significant economic contributions to the State of Florida.

With multiple authorities having control of water resources in the ACF basin, the states of Alabama and Florida along with the USACE quarreled with the State of Georgia in the 1960’s and 1970’s regarding the opinion that water consumption for urban and agriculture uses within Georgia had superseded the needs of other water-users and ecosystems to the south and the west. In January 1982, in lieu of imminent litigation, a ‘Memorandum of Agreement’ (MOA) was signed by the governors of the three states, and the USACE, in which all four parties agreed to develop a comprehensive basin-wide water assessment, a navigation maintenance plan, a drought management plan, and a water management strategy for the ACF watershed (Leitman et al.

1991). The ACF basin MOA attempted to develop a rational and scientific water management policy based on a '*live and let-live*' principle. However, after several extensions, the MOA had failed to result in a compromise due to insufficient funding, incomplete habitat assessments, withholding of information, threats of new litigation, changes in elected officials, and overall differences in opinion regarding definitions of reasonable water use (Dellapenna 2004). In early 1996, the states of Florida, Georgia, and Alabama revisited the water allocation problems, and began drafting an outline to develop an interstate water compact (Carriker 2000). During their 1997 legislative sessions, the three states passed the framework program, the governors signed the measure, and sent the proposal to Washington D.C. for congressional approval. The US Congress ratified the Apalachicola-Chattahoochee-Flint River Basin Compact (ACFRBC) on November 20, 1997, that established an agenda, timeline, and an interstate basin commission, for developing an agreement to allocate water from the shared river system (Carriker 2000; Sherk 2005). There were many points of contention during the years of negotiations under the ACFRBC, but the fundamental disagreement among the parties concerned whether the State of Georgia would be responsible for delivering only the minimum flow of water, regardless of hydrologic conditions, to the Florida border (Snowden 2005). It seemed as if a stalemate was inevitable; however, on July 22, 2003, after numerous deadline extensions, the states of Florida, Georgia, and Alabama signed a 'Memorandum of Understanding' that indicated that they had reached an 'agreement in principle' regarding various terms of a water allocation formula, and committed the states to additional negotiations (Sherk 2005). Unfortunately, little progress was made during talks in the months that followed as the states became increasingly unwilling to make concessions; therefore, on August 31, 2003, without another extension, the ACFRBC terminated (Dellapenna 2004). The expiration of the ACFRBC officially ended objective dialogue between the states of Florida, Georgia, and Alabama regarding water allocation in the ACF basin, and has forced all subsequent decisions to be made within the federal court system.

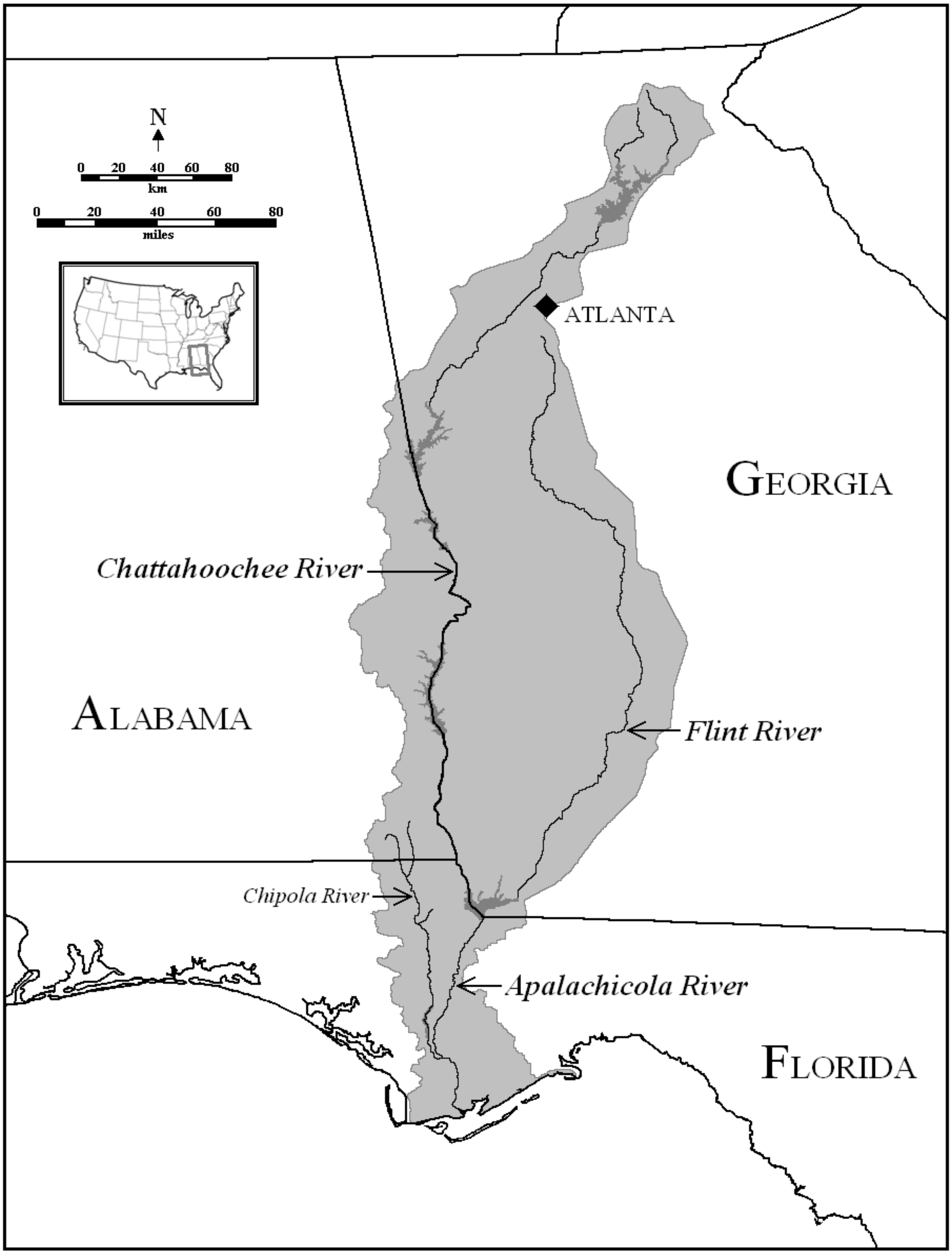

Figure 2-1. Map of the ACF drainage basin in Florida, Georgia, and Alabama.

CHAPTER 3
CHARACTERIZATION, FLUVIAL EFFECTS, AND SIGNIFICANCE OF SNAGS

Definition of Snags

The term ‘snag’ is used in lotic systems to refer to any woody structure, regardless of size, that lies within the banks of a watercourse. Snags are ubiquitous and integral components of rivers and streams flowing through forested floodplains. The importance of snags as a habitat component in the ecology of aquatic systems has become increasingly recognized during the last 30 years (e.g., Bilby and Likens 1980; Wallace and Benke 1984; Bilby and Ward 1991; Beechie and Sibley 1997). Tree trunks, limbs, branches, leaves, seeds, bark, and root wads make up the majority of allochthonous input deposited in and along rivers and streams (Hynes 1975; Dolloff and Webster 2000). However, during high water inundation, roots, trunks, and low-hanging limbs of living trees can also serve as snag habitat for aquatic dwellers. In the southeastern US, downed woody material can be a result of bank undercutting and erosion, natural tree fall, forestry operations, fire, flood, lightning, windthrow, disease, insect infestation, or episodic weather events such as tornadoes and hurricanes.

The majority of existing research concerning the ecology and fluvial effects of aquatic woody material has been carried out in the high-gradient rivers and streams flowing through timber forests of Australia, as well as those in the US Pacific Northwest (e.g., Anderson and Sedell 1979; Naiman and Sedell 1979; Triska and Cromack 1980; Webb and Erskine 2003; Woldendorp et al. 2004). Some of the earliest studies of the subject compared natural abundances of wood in undisturbed watersheds with those where wood was deposited as a result of timber harvesting (Warren and Olsen 1964; Bilby 1981). Other studies focused primarily on woody material of at least 0.1-m in diameter, known as ‘large woody debris’ (e.g., Keller and Swanson 1979; Bisson et al. 1987; Lienkaemper and Swanson 1987; MacDonald and Keller

1987). The term ‘large woody debris’ (LWD) originated in the logging industry, and referred to waste products, specifically the larger branches, limbs, and unusable trash logs left at a logging site after the profitable stems were removed (Murphy and Koski 1989). With increased recognition of the significance of woody material in fluvial systems, aquatic ecologists have recently suggested that the term LWD be abandoned in favor of the term ‘large wood,’ thereby eliminating any possible negative connotation associated with the word ‘debris’ (Gregory et al. 2003).

Effects of Snags on Energy Flow and Nutrient Cycling

Ecotones are defined as regions of transition associated with sharp gradients in environmental factors or in community composition (Stiling 1999). Riparian zones such as floodplain forests are vital ecotones characterized by high species diversity, and high productivity, between terrestrial upland and lowland aquatic ecosystems. Often depicted in two dimensions, it is imperative to consider that riparian zones are three-dimensional, biogeochemically open systems in nature; they are laterally connected to upslope terrestrial localities, as well as functionally connected longitudinally to upstream aquatic locations (Mitsch and Gosselink 2000). By serving as superior filters, floodplains are functional and valuable ecotones within fluvial systems.

Aquatic snags have a prominent influence on numerous ecosystem processes, including energy budgeting, primary and secondary production, as well as nutrient resource cycling and exchange. The pattern of nutrient loading, transport, utilization, and storage on the floodplain is critical to the survival of healthy riparian communities (Vannote et al. 1980; Baldwin and Mitchell 2000). This pattern is regulated by the ‘flood pulse’ concept; a periodic flooding of fluvial systems that provides a vital ecological interaction between the river channel and its floodplain (Junk et al. 1989; Benke 2001). Organic input is an essential component of the energy budget of aquatic habitats (Benke and Wallace 1990). The bulk of riverine animal biomass is derived from production of organic matter that occurs on the floodplain (Junk et al. 1989). During floods, a substantial flux of organic matter and nutrients are flushed off the floodplain where they either accumulate behind snags, or are deposited into the main river channel (Elder et al. 1988; Benke and Wallace 1990). Snags further influence the routing of sediments, particulate organic matter, and nutrients by creating areas of low shear stress and low water velocity, facilitating deposition of suspended materials from the water (Wallerstein 2003). Seasonal flooding also influences soil oxygen levels on the floodplain, whereby transitions from aerobic to anaerobic conditions affect rates of organic matter decomposition and nutrient mineralization. Floodplains act as nutrient sinks for lateral runoff from uplands, but also serve as nutrient transformers for downstream water flows (Mitsch and Gosselink 2000). By trapping particulate organic matter, snags help to retain potential nutrient and energy resources from washing rapidly downriver, as well as serve as a long-term source of nutrient loading for the floodplain (Wallace and Benke 1984).

Critical linkages exist between natural flow variations and ecological fitness of fluvial systems. Alterations of the timing, magnitude, and duration of extreme flows in rivers are significant, particularly in anthropogenically controlled watersheds (Gibson et al. 2005). Artificial adjustments of flow regime have cascading effects on the chemical, physical, and biological components of a river. Lower than normal riverine flows: 1) reduce the wetted perimeter of a waterway that decreases riparian habitat availability; 2) alter water-quality parameters by reducing dilution effects; and 3) limit connectivity between the river and its floodplain (Light et al. 1998). Increasing flows and rising water levels can be critical biological cues for riverine organisms to migrate up- or downstream, or for others to move from habitats in the main river channel to those on the floodplain where food resources and structure are readily abundant for reproduction (Ross and Baker 1983; Light et al. 1998; Postel and Richter 2003). Variations in peak flow timing and magnitude can disrupt the mobilization of organic matter and nutrients from the floodplain, alter the rates of sediment erosion and deposition, and significantly influence the migratory and recruitment success of aquatic species (Knighton 1998).

Geomorphic and Hydraulic Effects of Snags

Two major factors that govern channel morphology are sediment supply and temporal flow variability, both of which can be influenced by the presence or absence of snags. Due to their irregular shapes and sizes, snags can accumulate upon each other and develop into hazards for river transportation, as well as accumulate floating litter. Therefore, snags have been removed from rivers for many years in an attempt to create waterways suitable for navigation, as well as for aesthetic purposes. Other justifications employed for the removal of snags from rivers include: 1) to eliminate barriers to fish migration; 2) to reduce the risk of downstream bridge piling and navigational marker damage; 3) to reduce rates of river bank erosion; and 4) to improve flood mitigation efforts (Harmon et al. 1986; Bisson et al. 1987; Gippel et al. 1992, Diehl and Bryan 1993). It was often believed that removal of snags from rivers was a good course of action in flood prevention; however, snag removal tends to increases stream velocity, decreases riparian vegetation quantity, and increases channel bed scouring and incision that promotes bank failure as sediment is carried downstream (Zimmerman et al. 1967; Wilson 1973; MacDonald and Keller 1987; Erskine 1994). Increased bank stability generated by snag material is especially important along the outside bank of meander bends where the channel tends to migrate laterally (Bisson et al. 1987; Diehl and Bryan 1993). Conserving the snag material that naturally accumulates in the riparian areas of fluvial systems helps to fortify the channel banks, trap sediment in natural levees, increase structural complexity and habitat, as well as reduce the mean velocity of water in the channel and on the floodplain during high-water events (Díez et al. 2001).

The most conspicuous effects of snags in rivers and streams are physical: wood can influence channel shape, sediment storage and erosion rates, and the path of water flow. Downed wood on the floodplain that is large enough to be immobile during floods can have a significant influence on morphology, geometry, and profile of the floodplain (Braudrick et al. 1997). When a log lies parallel to the flow of a floodplain slough, the streambed may be eroded in front of the log forming a lateral scour pool. Snags can divert the mainstream flow of a slough from one side of the channel to the other, forming a backwater pool. Beechie and Sibley (1997) found that pool area and pool spacing varied greatly with channel width and slope in streams containing snags. Pools formed by wood on the floodplain provide essential habitat and complexity to the entire bionetwork (Bilby 1981; Dolloff 1994; Baillie and Davies 2002). Pool formation is just one of the physical effects that snags can have on fluvial systems.

Another physical effect of snags in fluvial systems is bank armoring, or flow divergence. Snag material can divert stream flow away from a bank, therefore limiting downstream shoreline erosion (Gippel 1995; Berg et al. 1998). Snags can also divert water into a bank, causing bank undercutting and stream widening. Conversely, wood lying alongside and parallel to the bank can prevent bank erosion (Cherry and Beschta 1989). Snags can also shift water flow from one side of a channel to the other, divert flow from the main channel to a side channel, or even force the creation of alternate channels. Channel migration can also create point bars of alluvial sediments, oxbow lakes, and backwater swamps. The influence of snags on formation of these local hydrologic features is extensive; however, the impact of humans on fluvial systems tends to constrain channel processes and linearizes corridors, while natural hydrology and geomorphology tends to create heterogeneity of these habitats.

Zones of erosion, storage and transport, and deposition are defined along the course of a river system based on its geomorphology, hydrology, and physical processes. Snags can influence the deposition patterns of sediments in rivers and on their floodplains. Water may be forced to go around, over, under, or along a snag, resulting in a range of water velocities. Fine sediments such as silts and clays will accumulate where the water velocity is lowest, such as in backwater pools, whereas only coarse sediments remain stationary in faster flows. The presence of snags in rivers can reduce the scouring effect of high water flow, and aid in sediment deposition (Baillie and Davies 2002). Rootwads may act as streambank anchors against high flows during floods, reducing soil loss to erosion and downstream sedimentation. In relatively low-gradient floodplains, snags can form logjams which can increase the immediate water level and inundate large areas that do not normally experience regular flooding. Often these temporary floods deposit considerable amounts of suspended materials (Ward and Aumen 1986). Snags are therefore considered an important factor in fluvial sediment deposition.

Ironically, in many locations around the world substantial amounts of time and money are invested in redepositing snags into rivers and streams that were previously cleared (Gore 1985; Andrus et al. 1988; Osborne and Koviac 1993). Determining where snags should be reintroduced, how they should be oriented, and the quantities of woody material that should be returned vary by watershed (Hrodey and Sutton 2008). Piégay (2003), Swanson (2003), and Reich et al. (2003) propose various frameworks for wood input and redistribution based on specific parameters of individual fluvial systems. The challenge for river managers is to find and establish a reasonable balance between the effects of bank erosion and navigational concerns with the ecological benefits of leaving snags in rivers (Hilderbrand et al. 1998).

Snag Decomposition and Deadhead Logging

Most stream inhabiting organisms do not feed directly on snags, but the presence or absence of these resources can have a significant influence on food availability and the diversity of fauna in the ecosystem. Helfman (2007) indicated that in low-gradient coastal rivers, snags are the most biologically rich habitat available, and can produce four times as much prey than mud or sand habitats alone. The ecological link between energy availability and production of higher trophic levels has been well documented (e.g., Myers 1977; Benke et al. 1984; Mattraw and Elder 1984; Kemp and Boynton 2004; Kiffney and Roni 2007). Leaves and small twigs that fall into the stream are an important food source for microbes, bacteria, and many species of aquatic invertebrates (Bilby and Likens 1980). Large, stable snags trap smaller pieces of wood, that in turn, capture leaves and other organic detritus creating nutrient reservoirs on which organisms feed (Bilby 1981). A 1984 study by Wallace and Benke determined that 75% of the organic matter stored in a South Carolina stream and floodplain system was directly associated with snag material. Wood on the floodplain located above existing water levels contributes to available refugia at higher water levels (Benke et al. 1985). Logs suspended above the channel can be colonized by a variety of plants and animals, creating a lush microcosm above the water (Brown 2002). Eventually, the suspended log will enter the waterway and add to its wood supply.

The methods and rates of dead wood decomposition in terrestrial and aquatic habitats are remarkably different. Decay of dead woody material in terrestrial systems is primarily driven by fungi, and rate of decomposition is rapid due to an abundance of available atmospheric oxygen (Rayner and Todd 1979; Elder and Cairns 1982; Baker et al. 1983). In aquatic systems however, decomposition of snags is achieved primarily by single-celled bacteria, namely actinomycetes (Aumen et al. 1983), and rate of decomposition is determined by microbial quantity and detritivore palatability (Boling et al. 1975; Bilby 2003). Bacteria and algae living on the surfaces of submerged snags form a slimy coating called biofilm, which becomes food for other microscopic biota (Robertson et al. 2001). Biofilms on snags can also beneficially withdraw nutrients from water flowing by, helping to regulate potentially harmful algal blooms and downstream macrophyte growth, as well as conserve potential energy resources (Tank and Webster 1998). With the exception of the outermost layers of submerged dead wood, the dissolved oxygen concentrations throughout the core of a snag are very low; therefore, breakdown of its structural components is much slower than in a terrestrial system (e.g., Anderson et al. 1978; Whiles and Wallace 1997; Bilby et al. 1999; Spänhoff and Meyer 2004).

Old-growth longleaf pine and cypress forests were extensively logged throughout the southeastern US in the 19th and early 20th centuries to support worldwide demands for lumber. Typically, felled timbers would be floated down rivers to coastal sawmills for conversion into useable boards. It is estimated that five to ten percent of the dense heartwood timbers became waterlogged and sank on their attempted journey to the mill sites (Dye and Frydenborg 1999). In coastal rivers and streams of the southeastern US where clays, mud, and sand constitute the primary bottom sediments, large wood can become partially or entirely buried in anoxic conditions, and is therefore increasingly resistant to decay (Mitsch and Gosselink 2000). These submerged ‘deadhead logs’ can remain on the river bottom for hundreds of years, and are extremely valuable in the commercial wood-products industry due to their tight grain pattern and unique wood colors. Individual high-grade deadhead logs can fetch three to five thousand US dollars apiece, and generate lumber that is upwards of ten times more valuable than conventional wood (Dye and Frydenborg 1999). The high-end wood produced from old-growth deadhead logs are utilized primarily in the furniture, paneling, and flooring industries.

Removing deadhead logs from rivers has created legal disputes between log salvaging operations and environmental groups. Log salvaging representatives argue that the logged-timbers were not present in rivers in the first place, and should therefore be removed. Environmental and cultural interests assert that deadhead logs have become integral parts of rivers over time, and should consequently remain undisturbed. Management of these non-renewable submerged resources became increasingly necessary, so several US states prohibited underwater deadhead logging operations while environmental studies were conducted to evaluate potential physical, chemical, and biological impacts of deadhead log removal. These studies generated contradicting results, and consequently many states have developed an annual fee-based permitting process to manage log removal quantities, and to monitor impacts of deadhead removal while additional research is conducted.

Influence of Snags on Aquatic Fauna

The presence of snags increases the quantity and diversity of available habitats for aquatic-dwelling organisms. Snags generate effective habitat heterogeneity due to their complex structure that in turn provides a variety of nooks and crannies suitable for refuge. Snags are especially important in altered or degraded rivers, as well as in rivers where other more stable bottom substrates, such as rock or gravel, are absent.

Macroinvertebrates such as insects, mollusks, crustaceans, and worms are highly sensitive to changes in their environs; therefore, their population levels are often investigated and monitored to assess the overall health and biodiversity of aquatic ecosystems. Due to this, many scientific studies have been conducted through the years to examine invertebrate use of snag habitat (e.g., Nilsen and Larimore 1973; Anderson et al. 1978; Molles 1982; Benke et al. 1984;

Smock et al. 1985; Benke et al. 1985; Thorp et al. 1985; Lemly and Hilderbrand 2000; Braccia and Batzer 2001). Most of these studies have shown that a greater number of individuals and a greater diversity of invertebrate species are found in fluvial reaches containing snags in comparison to similar reaches without snags. Aquatic insects utilize snag habitats for feeding (whether scraping, grazing, or filtering), resting, predator avoidance, egg deposition, and for pupation and emergence (Anderson et al. 1978, Benke et al. 1984; Drury and Kelso 2000). In a lower Coastal Plain river in the southeastern US, Benke et al. (1984) found that the diversity of invertebrate taxa on snags was much greater than that of sand or mud; however, snag habitat made up only 6% of the available substrate. Ager et al. (1985) determined that in a southeastern US fluvial system, snag habitat had the highest density of macroinvertebrates when compared to all other potentially available benthic substrates. In large sandy bottom rivers, like those found in the Coastal Plain of the southeastern US, snags are extremely vital as they provided the only stable periphyton habitat for many invertebrates.

The unionid mussel fauna of the southeastern US was at one time the most species-rich in the world, and the ACF basin is considered to be the center of unionid mussel speciation (Brim Box and Williams 2000). However, modern ACF mussel assemblages have experienced an abrupt decline, perhaps by as much as 50%, as a result of habitat degradation, pollution, impoundment, and introduction of exotic species (Williams et al. 1993; Golladay et al. 2004). Chastain et al. (2006) indicated that the Flint River had 76% of its historical mussel species remaining, while Brim Box and Williams (2000) reported that only remnant and isolated unionid populations remain in the Chattahoochee River. Of the remaining ACF mussel assemblages, 64% of the species are categorized as either endangered, threatened, or of special concern (Brim Box and Williams 2000). Unionid reproduction is unique in that it is reliant on specific fish species to serve as hosts for their specialized larvae; successful mussel reproduction is therefore dependent on the spawning and rearing success of these critical fish species. Freshwater mussels are an important component of Coastal Plain fluvial ecosystems as they filter nutrients from the water and accumulate potentially harmful chemicals and metals into their bodies and shells (Hart and Fuller 1974). Decreased water flows in the ACF basin leaves unionid mussel populations vulnerable to extirpation which could ultimately have impacts on water-quality within the watershed.

An increasing number of studies have been published recently regarding snags as essential fish habitat. Analogous to the structural cover provided by coral reefs in marine ecosystems, snags serve as a feeding, spawning, and nursery site for fish in fluvial systems. Submerged snags are especially significant in shifting sandy-bottomed lotic systems, where they provide niche variation, velocity refuge, and often the only available stable substrate and cover (Wallace and Benke 1984; Braccia and Batzer 2001). Benke et al. (1984) found that several species of game fish foraged exclusively on invertebrates associated with snag material, and Benke et al. (1985) reported that major warm-water fishes in a southeastern US Coastal Plain stream obtained at least 60% of their prey biomass entirely from snag habitat. Anglers have historically been the greatest advocates for snags in fluvial systems, as they have long recognized that snags serve as effective fish aggregation structures (Lobb and Orth 1991). Ager et al. (1985) indicated that river habitat containing snags was found to have twice as many game fish species as similar habitats without snags. Alteration of snag abundance can also influence the distribution of apex predators and prey. Studies by Angermeier and Karr (1984) and Peterson and Bayley (1993) determined that the removal of snags from streams altered the predator-prey balance by effectively reducing the number of apex predators, and increasing the number of smaller fish species. In high gradient streams, fish often congregate in dammed plunge pools created by snags and large wood as an escape from high water velocities, and as refuge from predators (Angermier and Karr 1984). Reduction in riverine snag quantity and distribution can severely limit the quantity of food resources and protective cover available to fish, two essential components of healthy fish populations.

Invertebrates and fish are not the only fauna that use snags as habitat in fluvial systems. Snags emerging from a river provide a preferred site for resting, foraging, and roosting for many aquatic turtle, snake, and bird species. Snags on the floodplain can be valuable habitat for herpetofauna such as salamanders, skinks, lizards, frogs and toads, as well as for mammalian predators such as rat, raccoon, opossum, and armadillo that often forage at floodplain snags sites (Means 1977). Next to humans, the most influential contributor to snag habitat abundance is the beaver, whose impoundments and lodges constructed from wood obtained on floodplains can significantly alter stream hydrology, and introduce large quantities of snag material into fluvial systems (McTaggart and Nelson 2003; Pollock et al. 2003).

CHAPTER 4
NOVEL METHOD FOR QUANTIFICATION OF EXPOSED SNAG MATERIAL IN FLUVIAL SYSTEMS USING DIGITAL IMAGING ANALYSIS

History of Aquatic Snag Quantification

To assess the potential importance of snags as habitat for aquatic fauna, it is necessary to quantify the woody material in lotic systems. Historically, estimates of snag abundance and density in freshwater aquatic systems were generated as a byproduct of other biological or hydrologic investigations, and it was not until the late-1970's that studies began to focus specifically on the quantification of aquatic wood (Gippel et al. 1996). Using methods originated in the forestry industry for calculation of residual logging waste, ecologists looked to enhance existing techniques for the quantification of aquatic wood (e.g. Anderson et al. 1978; Triska and Cromack 1980; Wallace and Benke 1984; Benke and Wallace 1990). The line intersect technique developed by Warren and Olsen (1964) and further modified by Van Wagner (1968), Bailey (1970), and de Vries (1974) was a practical method used to estimate the quantity of wood lying on or near the ground by simply obtaining the diameters of all woody stems that crossed a hypothetical sample line. Applied to fluvial systems, the line-intersect technique was reasonably effective for rapid assessment of large diameter wood exposed along the banks, or submerged in clear water, low-order reaches. In turbid, tannic, or high-order fluvial systems, however, the line-intersect technique was not a feasible option for quantification of submerged wood. In addition, the method was found to be significantly inaccurate at estimating quantities of small diameter woody material, and did not reveal much information about the distribution of wood along the sample transect (Gippel et al. 1996).

Due to the dynamic nature of fluvial systems, it is difficult to predict rates of wood recruitment and depletion over space and time. In addition to natural variations in river stage and downstream snag transport, anthropogenic influences such as forestry operations, and deadhead logging perturb total wood volumes in fluvial systems. Due to the limitations of the line-intersect method for wood quantification in turbid or lengthy river reaches, alternative methods such as low-level aerial surveys and side-scanning sonar have been used (Hamilton and Bergersen 1984). Unfortunately, these techniques have substantial drawbacks as well. Aerial surveys of river banks: 1) are generally expensive; 2) tend to be obscured by overhanging vegetation; and 3) do not provide an accurate vertical scale for the distribution of snags above the waterline (Fitzgerald et al. 2006). Side-scanning multi-beam sonar can be used to profile submerged snags even in turbid water. However, the technique: 1) is exclusively limited to the mapping of underwater material; 2) tends to perform better in deeper water with a uniform bottom profile; and 3) is susceptible to failure when the boat-towed sonar transducer becomes ensnared in the woody material that it is mapping (Flug et al. 1998).

In situations where stream flow was a limiting factor in the availability of aquatic habitat resources, an incremental methodology called IFIM, and a model dubbed PHABSIM were developed in the 1970's by a collaboration of federal and state agencies to predict changes in physical microhabitat associated with modifications in hydraulic flow regime (Bovee 1982, 1986). One of the most commonly utilized outputs from the IFIM and the PHABSIM model is an 'index of weighted usable area' that aids water managers in negotiating low-flow targets and assessments of critical habitat availability (Milhaus et al. 1989). The IFIM and the PHABSIM model are challenging to use because they entail multiple pathways and many steps that are based on user-level understanding of the problem at hand, and they require integrating existing long-term streamflow datasets of flow regimes, surface elevations, and water velocities, which are not always available (Milhaus et al. 1984). In addition, the IFIM and the PHABSIM model are limiting because: 1) their final outputs vary due to disparities in user-defined indices of channel characteristics; 2) the methodology and model have several significant assumptions that are difficult to overcome; and 3) the methodology and model do not account for critical water-quality or energy-input components desired in the interest of aquatic ecology (Milhaus 1999).

Very little research, especially in Florida, has been done to quantify the loss of habitat for aquatic fauna when snag material is exposed above water level during low-water conditions. Several studies by Ray (1999a-d) discussed the ecological and morphological significance of old growth deadhead logs as habitat for macroinvertebrates in several Florida panhandle rivers and streams, while a report by Estevez and Sprinkel (2000) examined the distribution and abundance of wood in the Santa Fe and lower Suwannee Rivers based on water level fluctuations. Wood quantification techniques used in these surveys were not feasible options for the conditions present in the Apalachicola River. Therefore it became apparent that another method of snag habitat quantification would need to be developed, perhaps using digital image analysis.

Digital image analysis is a computer-based technique used to generate numerical data by examining detailed properties of an image. Many scientific disciplines have turned to digital image analysis due to its widespread applicability, ease of use, and low cost. By utilizing a variety of methods and procedures, computers are able to accurately, dependably, and repeatedly identify and measure properties of an image that the human eye is unable to quantify. Historical limitations of using digital image analysis were primarily rooted in low image resolutions and slow computer processors; however, these restrictions have been surpassed by increasingly affordable advances in technology. In many scientific fields, digital image analysis has developed into an ideal method for rapid and inexpensive data analysis.

Study Design and Methods

Ager et al. (1987) defined six bank habitat types for the Apalachicola River based on steepness of the riverbank, substrate type, and presence of submerged aquatic vegetation. Steep sloping banks (bank angle greater than or equal to 45° from horizontal), gentle sloping banks (bank angle less than 45° from horizontal), and sandbar habitats comprised 92.6% of the Apalachicola River banks (41.7%, 27.0%, and 23.9% respectively), while the remaining 7.4% of bank habitats were classified as dike fields (2.3%), rock (2.1%), and submerged aquatic vegetation (3.0%) (Ager et al. 1987). Rock habitats were found exclusively in the uppermost portion of the Apalachicola River, while submerged aquatic vegetation was found exclusively in the tidal reaches of the river.

Four major reaches in the Apalachicola River were defined by Leitman (1984) based on differences in their geomorphic and physiographic characteristics: A) upper reach, B) middle reach, C) non-tidal lower reach, and D) tidal lower reach (Figure 4-1). For this study, an intensive study reach of 55.3 km was selected within the middle and non-tidal lower reach portions of the Apalachicola River; from just downstream of Estiffanulga Landing (RM 62.4) to just upstream of the confluence of the Apalachicola River with the Chipola River (RM 28.1) (Figure 4-2). This particular study reach was chosen based on its incorporation of four (steep banks, gentle banks, sandbars, and dike fields) of the six bank habitat types delineated by Ager et al. (1987) (Figure 4-3). Within the study reach, 30 sites were designated equidistantly from each other (≈2.0 km by river) using USGS 7.5-minute topographic quadrangle maps (Figure 4-4). Specific details regarding the location of the individual study sites are found in Table 4-1. At each of the 30 sites, both the left- and right-descending banks were evaluated for a total of 60 sampling stations. Geomorphic and bank habitat classifications of the sampling stations are summarized in Table 4-2.

In the field, a Garmin GPS III Plus™ (ver. 2.06) navigation unit was used to position the survey crew at each of the 30 pre-selected river study sites. Each sampling station consisted of four semi-contiguous 10-m long segments of riverbank. Every effort was made to have concurrent 10-m long segments for photography; however, certain segments were shifted upstream slightly (typically no more than 5 m) due to site topography. A Bushnell Yardage Pro® laser rangefinder was used by the boat operator to position a boat 35- to 40-m from the shoreline, bow directed upstream, into the current. From this position on the water, each of the 10-m segments of shoreline was digitally photographed three to five times in repetition by the photographer on the boat. In each photograph, a vertical reference scale was attained by having a field technician stand on the riverbank holding a stadia rod perpendicular to the water surface at the point where the bank and water converge. The stadia rod used for vertical reference was a 3.5-m x 0.05-m PVC pipe having 0.5-m black and white gradations along its length. Photographs were taken of the river shoreline from the existing water level to at least bankfull elevation or higher, with the stadia rod held vertically at the downstream end of each 10-m long segment (Figure 4-5). Field notes in regards to specific parameters for each sampling station were recorded on custom generated snag abundance datasheets (Appendix A).

Color photographic images were captured using a Nikon Coolpix® E5600 (ver. 1.0) handheld digital camera. The ‘normal’ imaging mode setting was used, with automatic focus and white balance, neutral exposure, single image capture, and standard color, which generated photographs with a pixel resolution of 2592 x 1944. Digital photographs were saved on Secure Digital (SD) memory cards in .jpg (24 bit) format. The file size for each digital image was approximately 1.0 MB.

Digital Photograph Processing and Analysis

All digital photographs were uploaded from the SD memory cards onto a *Microsoft*® *Windows*-based desktop computer where they were appropriately labeled and catalogued into a photographic database. For each of the 10-m shoreline segments photographed, a single representative photograph was selected from the three to five images captured in the field. The representative photograph was chosen based on its optimization of the following seven criteria: 1) image clarity; 2) focus; 3) resolution; 4) brightness; 5) image-shoreline parallelism; 6) shoreline-stadia rod perpendicularity; and 7) absence of avian or flying-insect interference.

Segment photographs were optimized for digital image analysis using *Adobe Photoshop* 7.0 (© 2002 Adobe Systems, Inc.) photo editing software. Original color images (Figure 4-6A), were 'inverted' to form a photographic negative and then converted to a grayscale format consisting of 256 shades of gray (Figure 4-6B). Grayscale images were reduced in size, and saved as uncompressed tagged image files (.tif).

Calculation of snag area was accomplished using *Image J* (ver. 1.35s) image processing and analysis software. *Image J* is a Java-based computer program developed at the US National Institutes of Health that runs in most major operating systems and is available free of charge as public domain software (Rasband 2002). By converting individual picture elements (pixels) to a unit of measurement, *Image J* is capable of calculating areas of irregular-shaped objects using user-defined scales.

Each grayscale .tif image file was opened in *Image J* for digital image analysis. Measurement scale was reset for each image by instructing the program to count the number of pixels necessary to traverse 1.0 m on the stadia rod ($n \approx 110$). After the scale for a photograph was defined, a 10.0-m horizontal line was stretched out digitally along the water surface using the 'line' and 'measurement' tools, and ultimately sketched on the photograph using the 'draw' command. Utilizing the 'box' tool, a 0.5-m tall x 10.0-m long rectangle block was expanded adjacent to the digitally sketched horizontal line, and the vertical stadia rod as a 5.0-m^2 measurement boundary for calculation. The entire image was then filtered in *Image J* with a simple thresholding algorithm at a threshold pixel brightness value of 128 to produce a binary image suitable for analysis (Ridler and Calvard, 1978) (Figure 4-6C). The thresholding procedure enhanced the snag material from its surroundings by converting all the grayscale pixels in the image to either a black or white pixel depending on its grayscale brightness value (0-128=black pixel; 129-256=white pixel).

Within the 5.0-m^2 measurement block, *Image J* counted all of the black pixels using the 'analyze particles' command, and generated an area (m^2) value based on the previously defined measurement scale. The measurement block was then moved 0.5 m up the stadia rod using the 'hand' tool, and the measurement process was repeated until all seven of the 5.0-m^2 blocks had been measured (Figure 4-6D). Area values generated by *Image J* were recorded into *Microsoft® Office Excel* (ver. 2003 SP2) spreadsheets for data management and further analyses. Detailed guidelines for digital photograph optimization, processing, and analysis techniques utilized in this study are contained in Appendix B.

Prior to actual data collection on the Apalachicola River, two preliminary studies were conducted to assess the repeatability, sensitivity, and logistics of using the proposed digital image analysis techniques for snag area quantification. In the first study, a 0.5-m x 0.5-m Plexiglas square was constructed, painted white, and subsequently framed with a 0.025-m wide black border as a sample object of known area. The test square was attached 1.75 m above ground level on a vertical pole, with one edge of the square parallel to the ground surface. A 100.0-m measuring tape was extended outward from the pole, and survey flags were inserted into the ground at 10.0-m increments. Three un-zoomed photographs of the test square were taken at each 10.0-m distance away from the pole. The test square was then rotated 45° and the photographic procedure was repeated. All photographs were then analyzed using the digital image analysis techniques to determine errors associated with variation in horizontal distance and target orientation. The results of the first pilot study indicated that horizontal distances within 80.0-m of the target object produced area values with less than 5.0% error, while distances within 50.0-m of the target produced less than 2.0% error. Orientation of the target was found to have no measurable effect on calculated areas using the proposed methods.

A second pilot study was conducted on the Suwannee River to address logistical details of the field-based methods. During this trial, a field assistant was positioned along the river shore with the 3.5-m stadia rod, and photographs were taken at various distances (~15.0-60.0 m) from the bank to assess variations in snag detail. Additionally, 10.0- to 25.0-m lengths of shoreline were photographed at a distance of 40.0-m from the bank to evaluate the length of shoreline necessary to generate abundant area data within each of the 0.5-m elevations above water level. Results of the second study indicated that photography at a distance of 40.0-m from the shore generated ample resolution of even the smallest snags along the bank, and 10.0-m lengths of shoreline produced sufficient area data for analyses.

Table 4-1. River mile and geocoordinates of field sites located within the 55.3-km intensive study reach of the Apalachicola River. River mile is the approximate distance upstream from the mouth of the river determined by USGS topographic maps. GPS positions are based on the WGS 84 datum.

Field Site Number	River Mile	Latitude	Longitude
1	62.4	30° 18’ 22.53” N	85° 02’ 59.08” W
2	61.1	30° 18’ 00.90” N	85° 03’ 25.74” W
3	59.9	30° 17’ 08.06” N	85° 03’ 25.74” W
4	58.8	30° 16’ 20.94” N	85° 03’ 41.66” W
5	57.5	30° 15’ 32.89” N	85° 03’ 42.74” W
6	56.3	30° 14’ 54.27” N	85° 04’ 19.94” W
7	55.2	30° 14’ 23.99” N	85° 04’ 46.05” W
8	54.0	30° 13’ 41.19” N	85° 04’ 57.32” W
9	52.8	30° 13’ 14.78” N	85° 05’ 25.93” W
10	51.7	30° 12’ 41.40” N	85° 06’ 03.12” W
11	50.5	30° 12’ 49.13” N	85° 06’ 40.85” W
12	49.3	30° 12’ 28.43” N	85° 07’ 11.06” W
13	48.1	30° 11’ 43.16” N	85° 07’ 19.82” W
14	46.9	30° 10’ 52.64” N	85° 07’ 57.00” W
15	45.9	30° 10’ 07.22” N	85° 08’ 23.99” W
16	44.6	30° 09’ 31.99” N	85° 07’ 54.86” W
17	43.4	30° 08’ 36.99” N	85° 08’ 14.29” W
18	42.3	30° 08’ 08.41” N	85° 08’ 16.97” W
19	41.4	30° 07’ 19.90” N	85° 08’ 39.48” W
20	40.0	30° 06’ 55.95” N	85° 07’ 51.97” W
21	38.7	30° 06’ 32.16” N	85° 08’ 20.72” W
22	37.7	30° 06’ 22.89” N	85° 08’ 51.80” W
23	36.5	30° 05’ 42.88” N	85° 09’ 08.76” W
24	35.4	30° 05’ 16.61” N	85° 08’ 20.01” W
25	34.3	30° 04’ 18.37” N	85° 08’ 18.76” W
26	33.1	30° 03’ 38.04” N	85° 08’ 11.80” W
27	31.9	30° 02’ 36.71” N	85° 08’ 09.12” W
28	30.7	30° 01’ 50.05” N	85° 07’ 20.40” W
29	29.4	30° 01’ 26.11” N	85° 06’ 26.86” W
30	28.1	30° 00’ 56.13” N	85° 05’ 37.79” W

Table 4-2. Geomorphic and bank habitat classifications of sampling stations within the 55.3-km intensive study reach of the Apalachicola River. Geomorphic classifications derived from USGS topographic maps, and bank habitat classifications based on Ager et al. (1987).

Station Number	Left or Right Descending Bank	Geomorphic Classification	If a Bend, Inner or Outer Bank	Bank Habitat Classification
1A	Left Bank	Straightaway		Gentle Bank
1B	Right Bank	Straightaway		Steep Bank
2A	Left Bank	Straightaway		Gentle Bank
2B	Right Bank	Straightaway		Gentle Bank
3A	Left Bank	Exiting Bend	Outer Bank	Steep Bank
3B	Right Bank	Exiting Bend	Inner Bank	Gentle Bank
4A	Left Bank	Straightaway		Steep Bank
4B	Right Bank	Straightaway		Gentle Bank
5A	Left Bank	Exiting Bend	Outer Bank	Steep Bank
5B	Right Bank	Exiting Bend	Inner Bank	Sandbar
6A	Left Bank	Bend Apex	Outer Bank	Gentle Bank
6B	Right Bank	Bend Apex	Inner Bank	Sandbar
7A	Left Bank	Exiting Bend	Outer Bank	Steep Bank
7B	Right Bank	Exiting Bend	Inner Bank	Gentle Bank
8A	Left Bank	Exiting Bend	Outer Bank	Steep Bank
8B	Right Bank	Exiting Bend	Inner Bank	Gentle Bank
9A	Left Bank	Bend Apex	Outer Bank	Steep Bank
9B	Right Bank	Bend Apex	Inner Bank	Sandbar
10A	Left Bank	Bend Apex	Outer Bank	Steep Bank
10B	Right Bank	Bend Apex	Inner Bank	Sandbar
11A	Left Bank	Straightaway		Gentle Bank
11B	Right Bank	Straightaway		Steep Bank
12A	Left Bank	Exiting Bend	Inner Bank	Sandbar
12B	Right Bank	Exiting Bend	Outer Bank	Steep Bank
13A	Left Bank	Straightaway		Gentle Bank
13B	Right Bank	Straightaway		Steep Bank
14A	Left Bank	Exiting Bend	Outer Bank	Steep Bank
14B	Right Bank	Exiting Bend	Inner Bank	Sandbar
15A	Right Bank	Bend Apex	Outer Bank	Steep Bank
15B	Left Bank	Bend Apex	Inner Bank	Sandbar
16A	Left Bank	Exiting Bend	Outer Bank	Steep Bank
16B	Right Bank	Exiting Bend	Inner Bank	Sandbar
17A	Left Bank	Exiting Bend	Inner Bank	Sandbar
17B	Right Bank	Exiting Bend	Outer Bank	Steep Bank
18A	Left Bank	Exiting Bend	Inner Bank	Gentle Bank
18B	Right Bank	Exiting Bend	Outer Bank	Steep Bank
19A	Left Bank	Exiting Bend	Inner Bank	Gentle Bank
19B	Right Bank	Exiting Bend	Outer Bank	Steep Bank
20A	Left Bank	Bend Apex	Outer Bank	Steep Bank

Table 4-2. Continued.

Station Number	Left or Right Descending Bank	Geomorphic Classification	If a Bend, Inner or Outer Bank	Bank Habitat Classification
20B	Right Bank	Bend Apex	Inner Bank	Gentle Bank
21A	Left Bank	Entering Bend	Outer Bank	Steep Bank
21B	Right Bank	Entering Bend	Inner Bank	Sandbar
22A	Left Bank	Straightaway		Steep Bank
22B	Right Bank	Straightaway		Gentle Bank
23A	Left Bank	Bend Apex	Inner Bank	Sandbar
23B	Right Bank	Bend Apex	Outer Bank	Steep Bank
24A	Left Bank	Bend Apex	Outer Bank	Steep Bank
24B	Right Bank	Bend Apex	Inner Bank	Sandbar
25A	Left Bank	Bend Apex	Inner Bank	Gentle Bank
25B	Right Bank	Bend Apex	Outer Bank	Steep Bank
26A	Left Bank	Exiting Bend	Inner Bank	Steep Bank
26B	Right Bank	Exiting Bend	Outer Bank	Dike Field
27A	Left Bank	Bend Apex	Inner Bank	Steep Bank
27B	Right Bank	Bend Apex	Outer Bank	Gentle Bank
28A	Left Bank	Straightaway		Gentle Bank
28B	Right Bank	Straightaway		Steep Bank
29A	Left Bank	Exiting Bend	Outer Bank	Steep Bank
29B	Right Bank	Exiting Bend	Inner Bank	Gentle Bank
30A	Left Bank	Bend Apex	Outer Bank	Steep Bank
30B	Right Bank	Bend Apex	Inner Bank	Gentle Bank

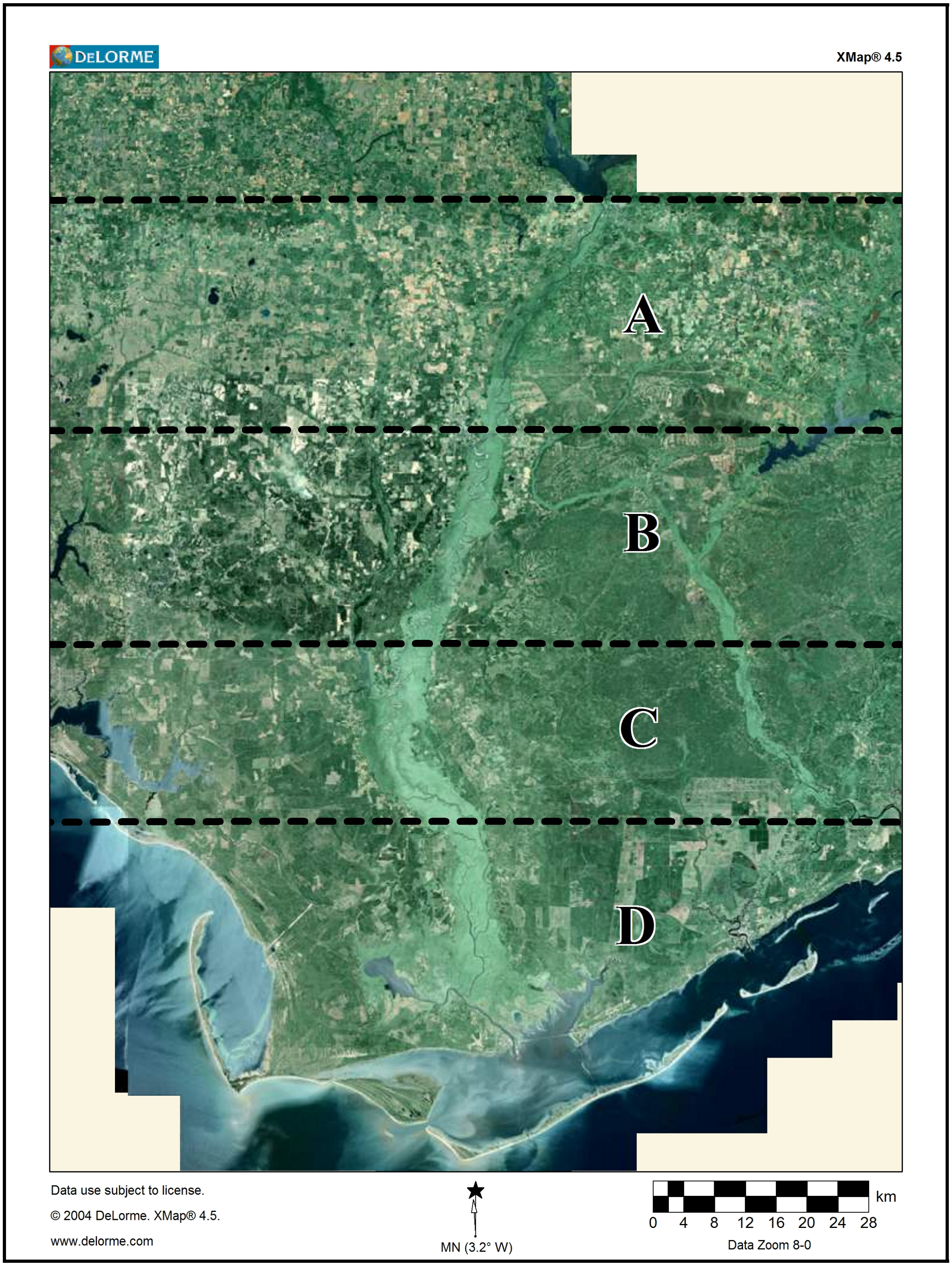

Figure 4-1. The four major reaches of the Apalachicola River as defined by Leitman (1984). A) Upper reach (RM 106.3-77.4). B) Middle reach (RM 77.4-42.0). C) Non-tidal lower reach (RM 42.0-20.6). D) Tidal lower reach (RM 20.6-0.0).

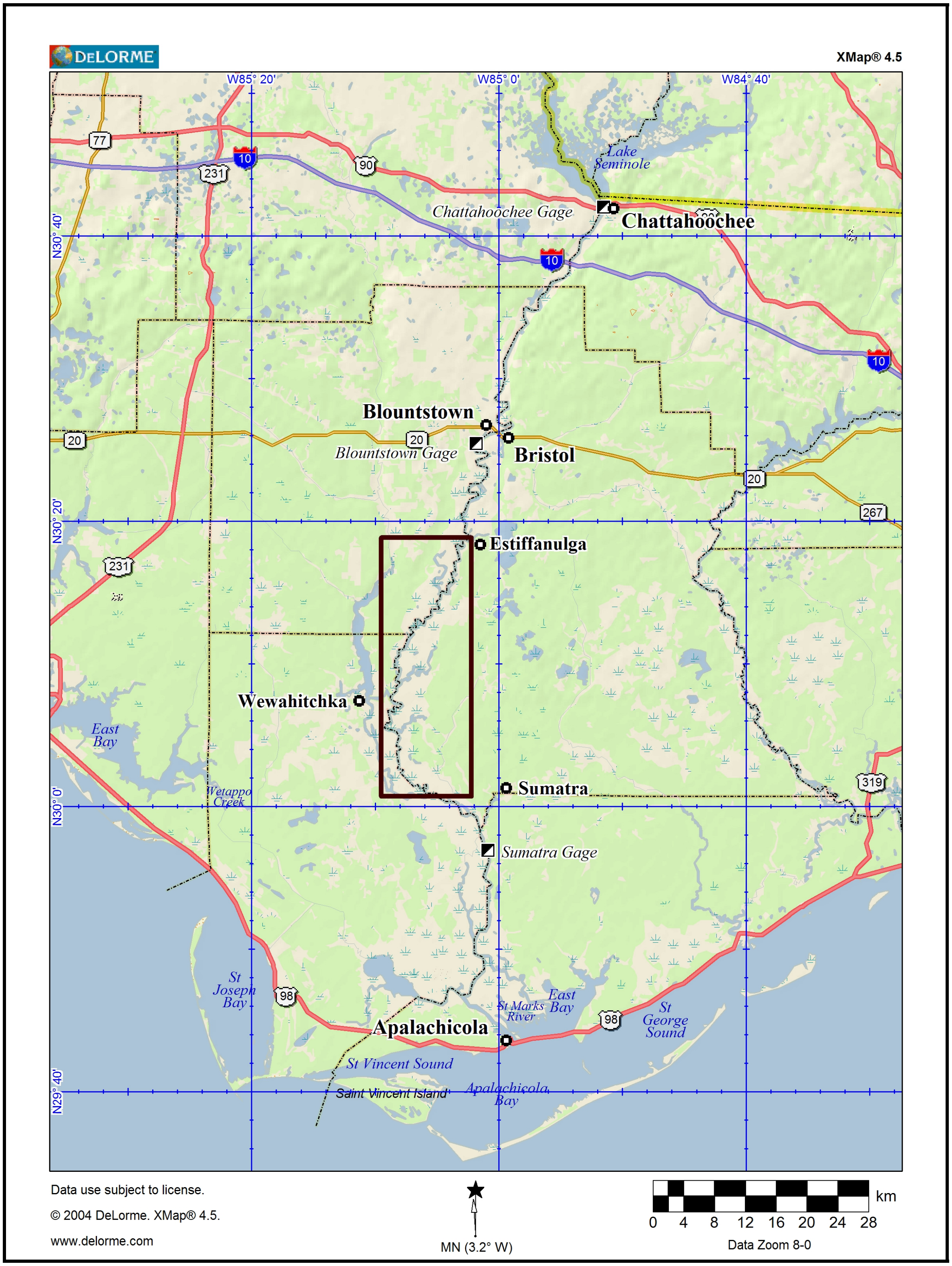

Figure 4-2. Map of the Apalachicola River basin and surrounding region. Large box indicates the location of the 55.3-km intensive study reach. Selected cities are denoted by circles, and river gaging stations indicated by squares.

Figure 4-3. Representative photographs of the four bank habitat types defined by Ager et al. (1987) encountered during this study. A) Steep sloping bank. B) Gentle sloping bank. C) Sandbar. D) Dike field. These images not used for analysis purposes.

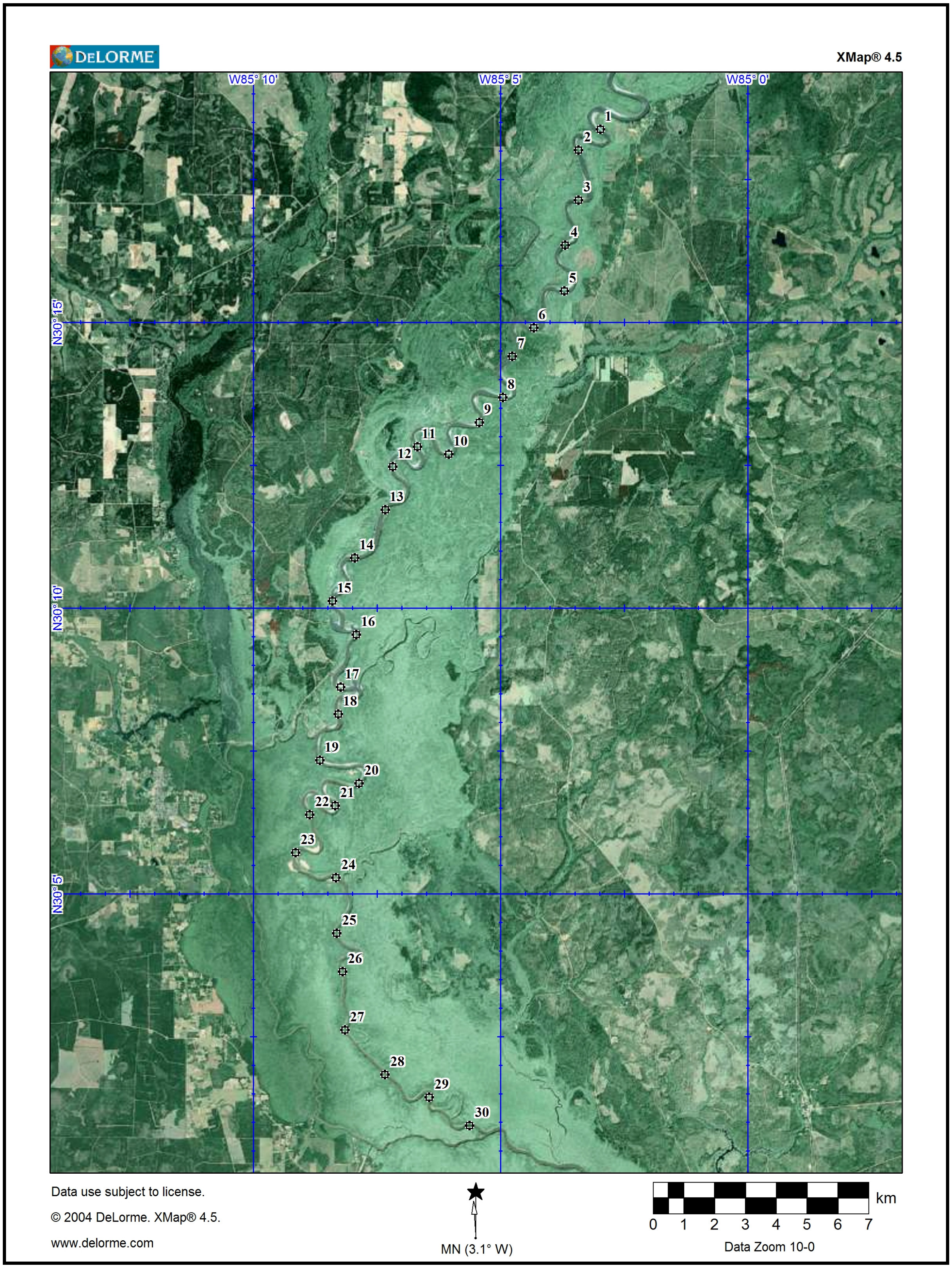

Figure 4-4. Map of the 55.3-km intensive study reach of the Apalachicola River indicating the specific locations of the 30 study sites. At each study site, both the left- and right-descending banks were evaluated for a total of 60 sampling stations.

Figure 4-5. Example of a digital image captured using the methods detailed in this study. Note the field technician on the riverbank holding the 3.5-m long stadia rod perpendicular to the water surface at the location where the bank and water converge.

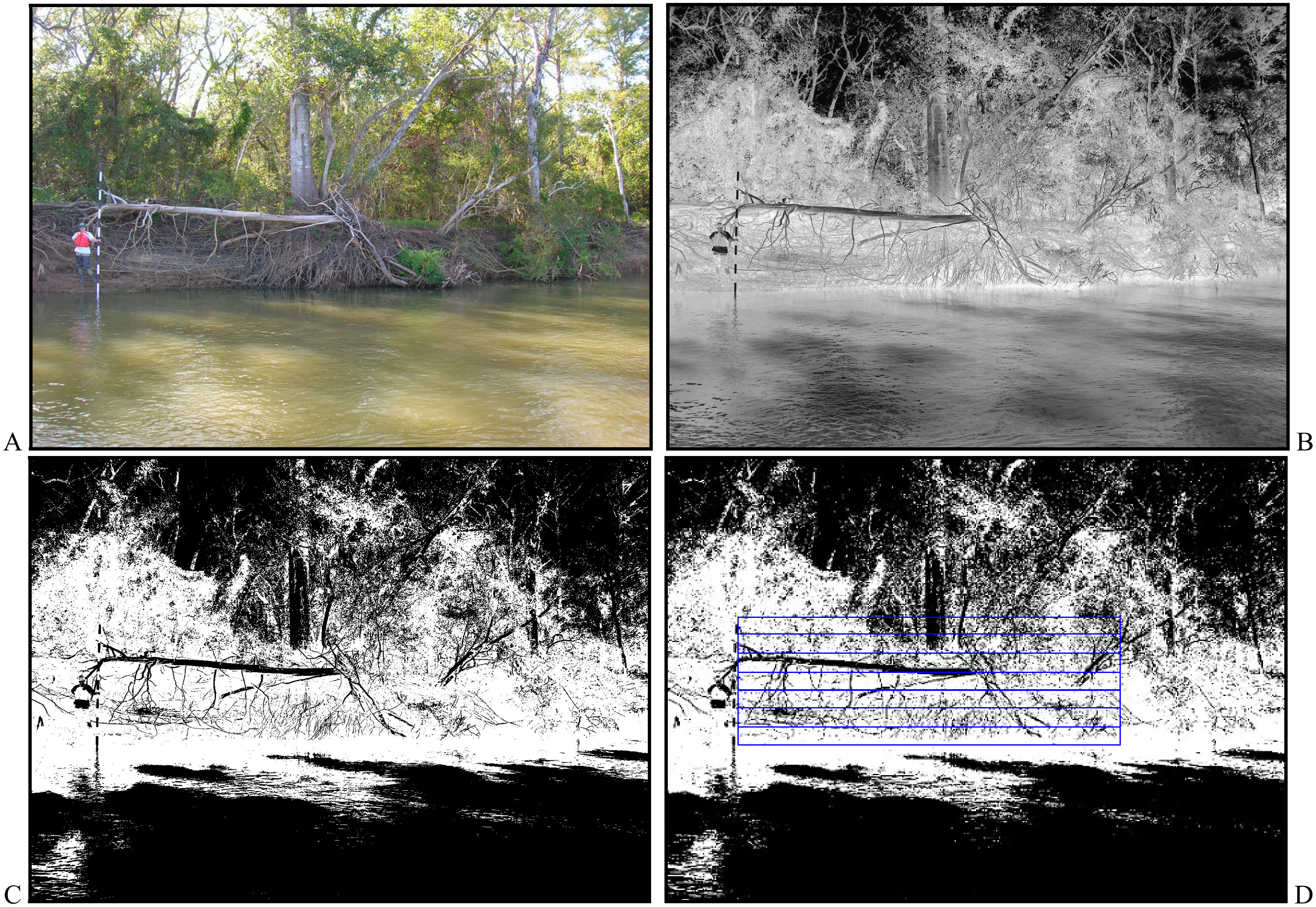

Figure 4-6. Filtering and processing steps used to prepare photographs for digital image analysis. A) Original color image. B) Grayscale negative of the original color image. C) Thresholded black-and-white image. D) Thresholded image illustrating the seven, 0.5-m tall x 10.0-m long superimposed measurement blocks used for digital image analysis.

CHAPTER 5
HYDROLOGIC ANALYSES, RESULTS, AND DISCUSSION

Hydrologic Analyses

In order to appraise riverine habitat availability in relation to river stage or discharge, it is necessary to understand the hydrology of the basin. By examining time series data from gaging stations, inferences can be made regarding changes in river geomorphology with time, as well as the likelihood of experiencing similar hydrologic conditions in the future (McCuen 2005). For this study, real-time hydrologic stage and flow data were available from three USGS surface-water gaging stations on the Apalachicola River. Gages were located at Chattahoochee (USGS # 02358000) (RM 105.8), Blountstown (USGS # 02359170) (RM 77.6), and near Sumatra (USGS # 02358700) (RM 20.6) (Figure 4-2). Data from the Chattahoochee gage were used in this study for all long-term flow analyses based on its extensive period of record (1929-present). All short-term flow analyses were made from data generated at the Blountstown gage due to its proximity to the 55.3-km intensive study area, therefore minimizing lag time effects. According to Light et al. (1998), Apalachicola River flows below 15,000 cfs at the Sumatra gage are generally tidally influenced, so data from this gaging station were not utilized in this low-flow study.

Based on 1929-2006 daily discharge data obtained from the Chattahoochee gaging station, a linear mean daily discharge duration curve was constructed to determine the percentage of time a given discharge was equaled or exceeded (Figure 5-1). In order to interpret the likelihood and magnitude of extreme flow events, the daily discharge data were transformed to produce a logarithmic mean daily discharge plot (Figure 5-2). Because the State of Florida and the USACE use data from the Chattahoochee gage to monitor and regulate water flows in the Apalachicola River, and because this study utilized stage and discharge data obtained from the Blountstown gage, an equivalency curve approximating the relation of Chattahoochee gage discharge readings with Blountstown gage stage readings for the Apalachicola River was generated (Figure 5-3). Additionally, a stage-discharge rating curve (Figure 5-4) and a stage-discharge hydrograph (Figure 5-5) were constructed from data obtained at the Blountstown gage spanning the duration of this study (March 2004-March 2006).

Due to the dynamic nature of fluvial systems, and because parameters and units used to specify water levels, flows, and elevations can become confounded, a datum designated as 'baseline' was defined in this study to signify the river conditions during data collection. Baseline readings at the Blountstown gage were a stage of 4.08 ft, and a discharge of 10,100 cfs. Snag quantification for this study was performed at a total of seven, 0.5-m distances above baseline. Table 5-1 contains equivalent stage, discharge, and NGVD 29 sea-level elevations at the Blountstown gage for baseline conditions, as well as at each of the 0.5-m distances above baseline.

Results

From 10-13 October 2005, a total of 1,060 digital photographs were captured of shoreline habitat within the 55.3-km intensive study reach of the Apalachicola River. Within this reach, 240 digital images totaling 2.4 km of river shoreline were quantified for snag abundance and distribution in each of the seven 0.5-m tall x 10.0-m long measurement blocks above baseline, using the digital imaging analysis techniques detailed in Chapter 4. Based on bank habitat classifications defined by Ager et al. (1987), nearly all of the 60 sampling stations in this study were classified as steep sloping banks, gentle sloping banks, or sandbar habitats; only one dike field station was encountered during the survey (Table 5-2). Most of the sampling stations (96.6%) were geomorphically classified using USGS topographic maps as being located at the apex of a bend, exiting a river bend, or along a river straightaway, while only 2 stations were located at the entrance to a bend in the river (Table 5-3).

Visual inspection of the original color images of river banks captured in the field indicated that sandbar habitats typically possessed the lowest quantity of available snag material (e.g., Figure 4-3C); however, comparison of mean snag area values generated for sandbar habitats using the novel methods presented in this study, with those of the three other habitats encountered, revealed that sandbar area values were erroneously elevated (Figure 5-6). Data from sandbar habitats were consequently omitted from subsequent analyses due to their inaccuracy. Analysis of mean snag area submerged for non-sandbar habitats indicated that the greatest quantities would occur within the first meter above baseline; gentle sloping bank habitat having the highest mean area submerged (3.19 m^2), followed by steep sloping banks, and dike field habitats (2.56 and 1.71 m^2, respectively) (Table 5-4; Figure 5-7). Mean snag area submerged for non-sandbar habitats based on geomorphic classification revealed the greatest quantities of snags would be submerged at a river stage between 0.5-m to 1.0-m above baseline; river banks entering a bend having the highest mean snag area submerged (3.97 m^2), followed by straightaway, exiting bend, and bend apex locations (2.97, 2.83, and 2.37 m^2, respectively) (Table 5-5; Figure 5-8). Further geomorphic analyses of non-sandbar habitats showed that within the first 0.5-m above baseline the inner banks of bends in the river have the most snag area submerged (3.09 m^2), followed the outer banks of river bends that would have a maximum mean area submerged of 2.50 m^2 at a distance between 0.5-m to 1.0-m above baseline (Table 5-6; Figure 5-9).

Examination of the sampling stations subdivided into Ager et al. (1987) habitat classifications indicated that 50% of the total available snag material would be submerged at 0.92 m above baseline for gentle sloping banks, 0.97 m above baseline for dike field habitat, and 1.32 m above baseline for steep sloping banks (Figure 5-10). Separating the non-sandbar habitat sampling stations into geomorphic classifications showed that the distance above baseline at which 50% of the total available snag material would be submerged at a range from 1.10 m for river straightaways, to 1.32 m for sampling stations entering a river bend (Figure 5-11). Additional geomorphic evaluation revealed that the distance above baseline at which 50% of the available snags would be submerged was 0.94 m for inner banks of river bends, 1.10 m for river straightaways, and 1.32 m for outer banks of river bends (Figure 5-12).

Discussion

In fluvial systems, hydrologic variability analyses and incremental methodologies are useful approaches for determining minimum instream flow requirements needed to maintain optimal ecological function and biotic integrity during low flows (Richter et al. 1997). Examination of the discharge duration curve (Figure 5-1) indicates that the Apalachicola River has experienced a remarkable variation of discharge readings during its 77-year period of record, with extreme high and low-discharges recorded on occasion. Because the curve generated in Figure 5-1 covers such a large range of values with many occurring infrequently, the data were transformed to produce a logarithmic flow duration plot to assist with interpretation of low-flow frequency (Figure 5-2). The relatively linear shape of the curve in Figure 5-2 indicates that the Apalachicola River has a strong baseflow component that can be attributed to both the immense size of the upstream Chattahoochee and Flint drainage basins, as well as to the significant groundwater influence within the basin (Elder et al. 1988). Mean daily discharge for the period of record at the Chattahoochee gage equaled or exceeded 15,900 cfs 50% of the time, while discharges of 8,480 cfs and 5,000 cfs were equaled or exceeded 90 and 99% of the time, respectively (Figure 5-2). Based on these curves, it appears that low-flow conditions in the Apalachicola River are not a frequent occurrence. However, the long period of record at the Chattahoochee gage includes 25 years of pre-impoundment, non-regulated discharge data that tends to amplify daily discharge means (Leitman et al. 1983). Additionally, increased consumption of water in the ACF basin in response to growing urbanization and agricultural demands, as well as increased rates of evapotranspiration due to river impoundment have decreased downstream river flows (Elder et al. 1988). Light et al. (2006) reported that recent discharge reductions in the Apalachicola River have been exacerbated by anthropogenic activities along with natural climatic change throughout the ACF basin.

The equivalency curve in Figure 5-3 indicates that a Blountstown gage stage of 4.08 ft (baseline in this study) is generated by a discharge of 10,200 cfs at the Chattahoochee gage. Other key Blountstown stage values are 5.72 ft and 7.36 ft (0.5 and 1.0 m above baseline, respectively), which correspond to a Chattahoochee discharge of 13,100 and 16,400 cfs, respectively (Figure 5-3). The shape of the stage-discharge rating curve for the Blountstown gage (Figure 5-4) indicates that at low flows, the river is confined primarily to its main channel; a small change in discharge results in a fairly substantial change in river stage. The curve also illustrates the distributional effects of the expansive Apalachicola River floodplain, in that once the river exceeds flood stage, large increases in discharge must be attained to notice appreciable changes in river stage (Figure 5-4).

The digital imaging analysis methods for snag quantification that are presented in this study are best suited for extremely low-flow conditions. Exposed snag material along the banks of the Apalachicola River was to be photographed at the lowest water levels possible; ideally at or near the State of Florida mandated minimum flow of 5,000 cfs at the Chattahoochee gage. Photography was scheduled to take place the first week of September 2004, as the summer had been reasonably dry, and water levels in the river were steadily declining towards the state minimum (Figure 5-5). However, the untimely arrival of Hurricane Frances to the Florida panhandle curtailed these plans. To complicate things further, Hurricanes Ivan and Jeanne followed several weeks later, which elevated water levels above flood stage, and eliminated low-flow photographic opportunities until September of the following year. With yearly tropical cyclone activity highly unpredictable, and having a limited timeframe for data collection, photography for this study transpired in mid-October 2005 at relatively low flows (Figure 5-5).

In an attempt to reduce the introduction of additional error and complexity to the digital imaging analysis methods, all photographs were filtered, processed, and analyzed in exactly the same manner. However, a major limitation of using the analysis techniques as described in Chapter 4 was revealed for sandbar habitats when comparing the computed mean snag area values with the original color images captured in the field. Nearly all photographs of sandbar habitats on the Apalachicola River resembled the image in Figure 4-3C; large aggregations of river sand with an occasional interspersed woody snag. Mean snag area values generated for sandbar habitats were erroneously elevated, particularly in the lowest intervals above baseline (Figure 5-6). Visual examination of the original color photographs revealed that images of generally shaded habitats types such as steep sloping banks, gentle sloping banks, and dike fields normally presented lightly-colored snag material against relatively darker backgrounds (e.g., Figure 4-5). Conversely, images captured of sandbar habitats typically contained darker-colored snag material against a nearly white sandy background. It was also determined that wet, and consequently darker-colored sand in the measurement intervals nearest to the water level for sandbar habitats were inaccurately interpreted by the analysis methods as snag material. Similar setbacks were not evident for snag areas computed of steep sloping banks, gentle sloping banks, or dike field habitats which were usually shaded from direct sunlight. Another drawback that was encountered exclusively at sandbar habitat stations was a depth-of-field problem created by the low-gradient bank angle that often produced a considerable distance between the stadia rod and the river bank in the background of the image. Unfortunately, the limitations of the analysis methods for sandbar habitats were not discovered during the pilot study on the Suwannee River, as only steep and gentle sloping banks were photographed in that trial. Potential modifications of the methods to reduce errors produced for sandbar habitats include: 1) an adjustment of the white-balance, or exposure settings of the camera when images of sandbar habitat are captured in the field; 2) use of an alternative threshold value other than 128 in the image processing steps; or 3) the development of an image processing algorithm that further isolates snag material from its surroundings by incorporation of ambient light levels with image contrast ratios. With visual inspections of original color photographs indicating nominal snag quantities on sandbar habitats, and due to the considerable quantification error revealed using the existing imaging analysis methods, mean snag area data from the 12 sandbar stations examined in this study were omitted from all subsequent analyses.

Mean area of snags submerged for steep sloping bank habitat appears to diminish fairly evenly from baseline to the 3.5-m distance above baseline, whereas snag area for gentle sloping banks was greatest from baseline to 1.0 m above baseline, then quickly tapered off (Figure 5-7). Diehl and Bryan (1993) indicated that steep sloping banks generally occur along the outside banks of river bends where erosion rates are fairly high. As the steep river banks are incised and undercut, snag material from the top of the bank tends to fall towards the river where it will often hang by its rootwad, or by a network of vines, until it ultimately becomes detached from the bank and deposited in the river (Golladay and Battle 2006). Gentle sloping banks stations on the Apalachicola River were for the most part located along the inner banks of a bend in the river, or along straightaways of the river (Table 4-2). Water velocities tend to be lower at these locations and consequently snags have a propensity to be deposited during decreased flows conditions. Additionally, Figure 5-7 indicates that when discharge and river stage increase, snags are picked up and transported by the increased water velocity, leaving the upper elevations of gentle sloping bank habitats without appreciable snag quantities. Dike field habitat was only represented in this study at one sampling station within the 55.3-km intensive study reach (Table 5-2), therefore the curve generated by these data was plotted for reference only, and further interpretation was reserved (Figure 5-7).

The entering bend geomorphic classification was also weakly represented in this study with a total of two sampling stations (Table 5-3). Only one of the two entering bend geomorphic stations was a non-sandbar habitat type, therefore, these data were plotted as a reference and interpretations should be made with caution (Figure 5-8). Analyses of the other curves within Figure 5-8 indicates that geomorphic classification of non-sandbar bank habitat type had relatively little influence on mean area of snags submerged, as well as vertical distribution within the 3.5-m distance above baseline. The majority of snags that would be submerged at greater river flows are found in the elevations just above baseline conditions, and steadily decreases as river stages approach flood stage levels (Figure 5-8). This trend would be expected as the main river channel is the usual collection site, and downstream transport vehicle for products washed off the floodplain (Ward et al. 1999). In addition, river velocities tend to decrease as stage decreases, so snags that would ordinarily be transported downriver during high-flow conditions end up being deposited along the river banks at lower river velocities (Gurnell 2003). Throughout the bends and straightaways of the Apalachicola River, snags quantities appear to be encountered fairly homogeneously.

Figure 5-9 illustrates that snag distribution along the outer banks of a river bend are somewhat more evenly distributed throughout the range from baseline to 3.5 m above baseline than along river straightaways, or along the inner banks of river bends. As the majority of outer bank river bend sampling stations were also classified as steep sloping habitats (Table 4-2), the increased rates of erosion and bank failure observed at these locations contributes to the slightly more even distribution of snag material. At both inner bank and river straightaway stations, the distribution of snag material is greater in the first 1.5-m elevations above baseline, and then rapidly diminishes at higher elevations (Figure 5-9).

Interpretation of the curves in Figures 5-6 through 5-9 should not be extrapolated beyond the axes on which they are plotted. The abundance and distribution of snag material below baseline conditions remains undetermined, and it should not be assumed that the existing trends would hold true at distances other than what are presented. Since this was a low-flow study, all snag material below baseline conditions was presumed to remain submerged at baseline discharges or greater.

Assuming that all quantified snags would be submerged at the 3.5-m distance above baseline, a relative percentage of total available snags was computed for each 0.5-m distance above baseline. In order to compare these curves, the distance above baseline where 50% of the available snags would be submerged was arbitrarily selected as a point of evaluation, in addition to the general shape of the curve. Analyses of the plots illustrating the relative percentage of snags submerged based on bank habitat classifications (Figure 5-10), geomorphic classifications (Figure 5-11), and by bank location within a river bend (Figure 5-12) exhibited results similar to those of the area submerged plots: 1) steep sloping banks exhibited the most uniform vertical snag distribution, while gentle sloping banks tended to have greater snag quantity at lower distances above baseline; 2) geomorphic classification had no considerable effect on snag distribution; and 3) outer banks of river bends tended to have a more balanced distribution of available snags throughout the range of measurement than did inner banks of river bends or straightaways.

Recommendations

Officials responsible for making water allocation choices should continue to rely on scientific research to assist with their decisions. Based on the results of this study, and those of scientists from other agencies, the State of Florida should continue to pursue increased river discharges from the Jim Woodruff Lock and Dam. As water levels decline in the river, so does the availability of valuable snag habitat which can have cascading negative effects on vital physical, chemical, and biological components of the ecology within the Apalachicola River and Bay ecosystems.

The water allocation issues that have developed within the ACF watershed are truly complex. Development of a water management plan for the basin which satisfies human consumptive demands while maintaining optimal ecosystem integrity and functionality is a difficult task, especially for a basin of its size. Intuitively, humans prefer to fix problems at the local level so immediate results can be achieved (Sparks 1995). However, in large watersheds like the ACF, a system-level approach should be encouraged for increased equity of success. Based on the increasing number of stakeholders with interests for water in the basin, there appears to be no feasible way to completely satisfy all of its users during unseasonably dry periods. With urbanization and agricultural growth continuing at a rapid pace, the inability to meet future water demands will most likely occur with increased frequency. Water conservation efforts among all users in the basin must continue to be endorsed, and rates of urban growth need to be checked. The effects of urbanization greatly exceed the delineated boundaries of urban areas. With this in mind, it is imperative that local, state, and federal officials remain committed to seeking common sense solutions towards an effective balance between the competing water demands and conservation needs of river-impacted flora and fauna within the ACF basin.

Table 5-1. Equivalent stage, discharge, and elevation readings at baseline conditions, and at 0.5-m distances above baseline, for the Blountstown gage, Apalachicola River.

Distance Above Baseline (m)	Blountstown Stage (ft)	Blountstown Discharge (cfs)	Blountstown Discharge (cms)	Distance Above NGVD29 (m)
3.5	15.56	39,700	1,124.6	12.96
3.0	13.92	33,400	946.2	12.46
2.5	12.28	28,300	801.7	11.96
2.0	10.64	23,700	671.4	11.46
1.5	9.00	19,600	555.2	10.96
1.0	7.36	16,000	453.3	10.46
0.5	5.72	12,900	365.6	9.96
Baseline	4.08	10,100	286.0	9.46

Table 5-2. Bank habitat classification summary for the 60 sampling stations within the 55.3-km intensive study reach of the Apalachicola River. Habitat classifications based on Ager et al. (1987).

Habitat Classification	Number of Stations	% of Total Study Area
Steep Sloping Bank	29	48.3
Gentle Sloping Bank	18	30.0
Sandbar	12	20.0
Dike Field	1	1.7

Table 5-3. Geomorphic classification summary for the 60 sampling stations within the 55.3-km intensive study reach of the Apalachicola River. Geomorphic classifications derived from USGS topographic maps.

Geomorphic Classification	Number of Stations	% of Total Study Area
Bend Apex	20	33.3
Exiting Bend	24	40.0
Straightaway	14	23.3
Entering Bend	2	3.4

Table 5-4. Bank habitat classification effects on mean area of snags submerged, and relative percentage of total snag area submerged, for non-sandbar habitat stations on the Apalachicola River. Snag areas submerged at 0.5-m distances above baseline are per 10.0-m of shoreline, and percentages are relative to all snags being submerged at 3.5-m above baseline. Bank habitat classifications based on Ager et al. (1987).

Distance Above Baseline (m)	Steep Sloping Bank (*n*=29)		Gentle Sloping Bank (*n*=18)		Dike Field (*n*=1)	
	Snag Area (m^2)	Relative %	Snag Area (m^2)	Relative %	Snag Area (m^2)	Relative %
3.5	0.99	100.0	0.71	100.0	0.62	100.0
3.0	1.12	90.7	0.67	93.1	0.47	91.6
2.5	1.40	81.3	0.63	86.9	0.47	85.9
2.0	1.78	70.0	1.11	81.3	0.73	79.9
1.5	2.28	56.4	2.36	72.1	1.35	70.0
1.0	2.56	38.5	3.14	53.7	1.61	51.8
0.5	2.29	18.0	3.19	28.1	1.71	28.3

Table 5-5. Geomorphic classification effects on mean area of snags submerged, and relative percentage of total snag area submerged, for non-sandbar habitat stations on the Apalachicola River. Snag areas submerged at 0.5-m distances above baseline are per 10.0-m of shoreline, and percentages are relative to all snags being submerged at 3.5-m above baseline. Geomorphic classifications derived from USGS topographic maps.

Distance Above Baseline (m)	Bend Apex (*n*=14)		Exiting Bend (*n*=19)		Straightaway (*n*=14)		Entering Bend (*n*=1)	
	Snag Area (m^2)	Relative %	Snag Area (m^2)	Relative %	Snag Area (m^2)	Relative %	Snag Area (m^2)	Relative %
3.5	0.88	100.0	0.76	100.0	0.99	100.0	1.47	100.0
3.0	0.94	91.5	0.86	92.2	1.00	91.0	1.70	92.5
2.5	1.15	83.1	1.00	84.5	1.06	82.5	2.43	83.7
2.0	1.50	72.9	1.56	76.0	1.32	74.0	3.03	71.4
1.5	2.05	60.0	2.39	63.8	2.29	63.8	3.60	56.3
1.0	2.37	41.7	2.83	45.4	2.97	46.4	3.97	38.4
0.5	2.29	20.6	2.61	22.4	2.86	23.1	3.72	18.6

Table 5-6. Effects of bank location within a river bend on mean area of snags submerged, and relative percentage of total snag area submerged, for non-sandbar habitat stations on the Apalachicola River. Snag areas submerged at 0.5-m distances above baseline are per 10.0-m of shoreline, and percentages are relative to all snags being submerged at 3.5-m above baseline. Bank locations derived from USGS topographic maps.

Distance Above Baseline (m)	Inner Bank (n=11)		Outer Bank (n=23)	
	Snag Area (m^2)	Relative %	Snag Area (m^2)	Relative %
3.5	0.62	100.0	0.94	100.0
3.0	0.60	93.7	1.06	91.1
2.5	0.56	87.8	1.37	82.0
2.0	1.19	82.6	1.77	70.8
1.5	2.44	72.3	2.21	57.1
1.0	3.04	53.1	2.50	39.2
0.5	3.09	28.0	2.23	18.5

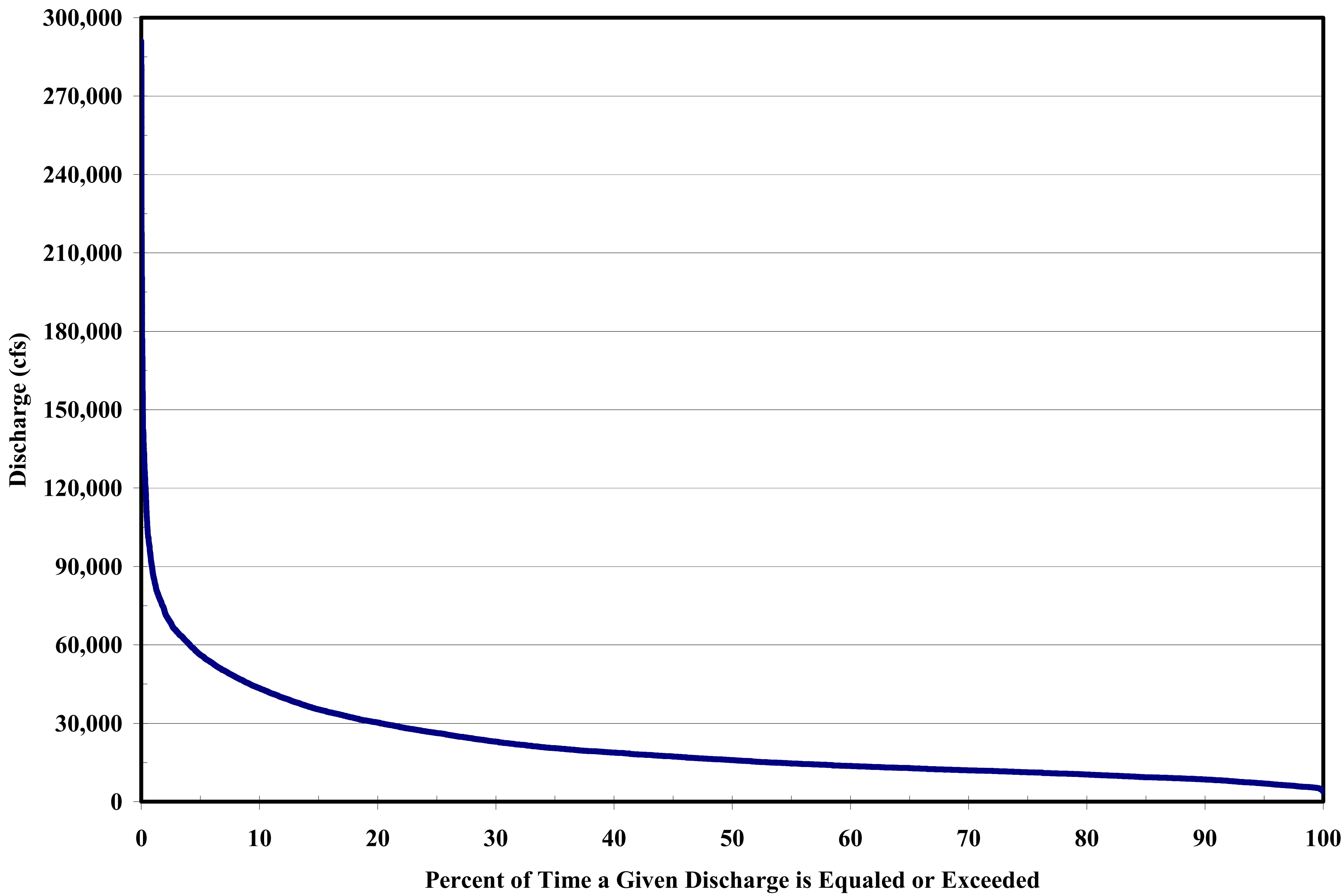

Figure 5-1. Linear mean daily discharge duration curve for the Apalachicola River at the Chattahoochee gage (1929-2006).

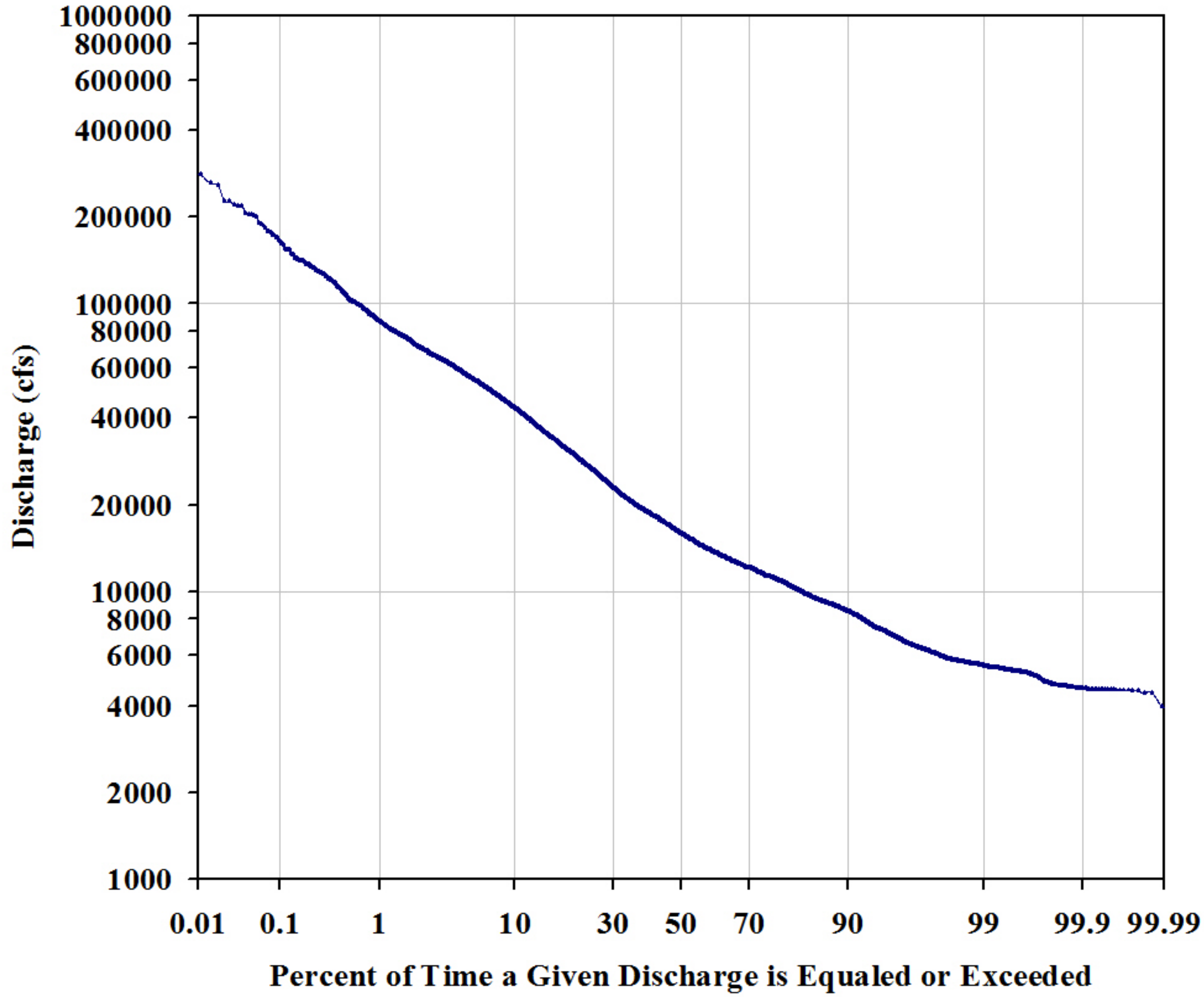

Figure 5-2. Logarithmic mean daily discharge duration curve for the Apalachicola River at the Chattahoochee gage (1929-2006).

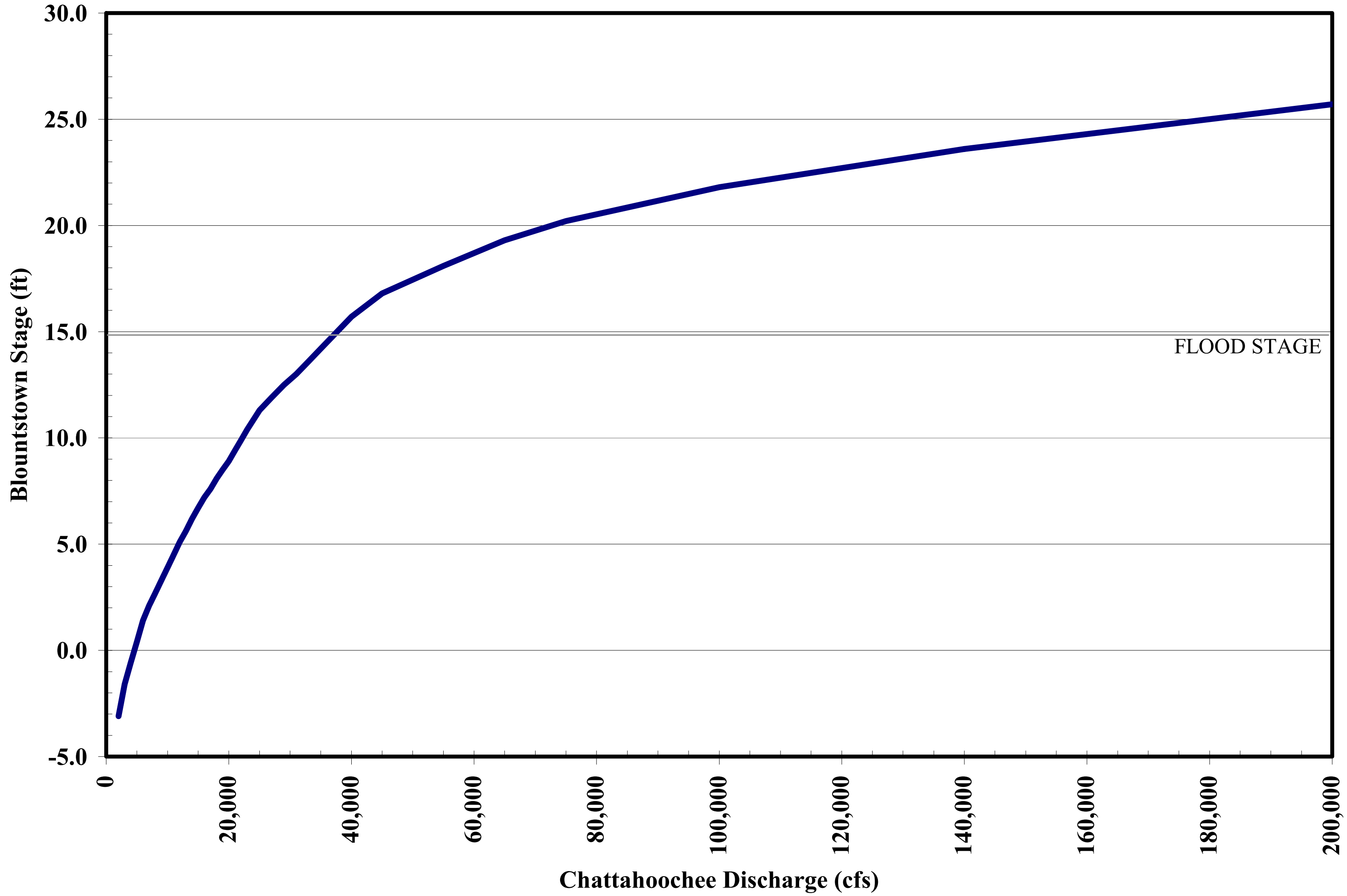

Figure 5-3. Equivalency curve approximating the relation of Chattahoochee gage discharge readings with Blountstown gage stage readings for the Apalachicola River. Curve based on unpublished data supplied by H. Light, USGS Tallahassee, 2002.

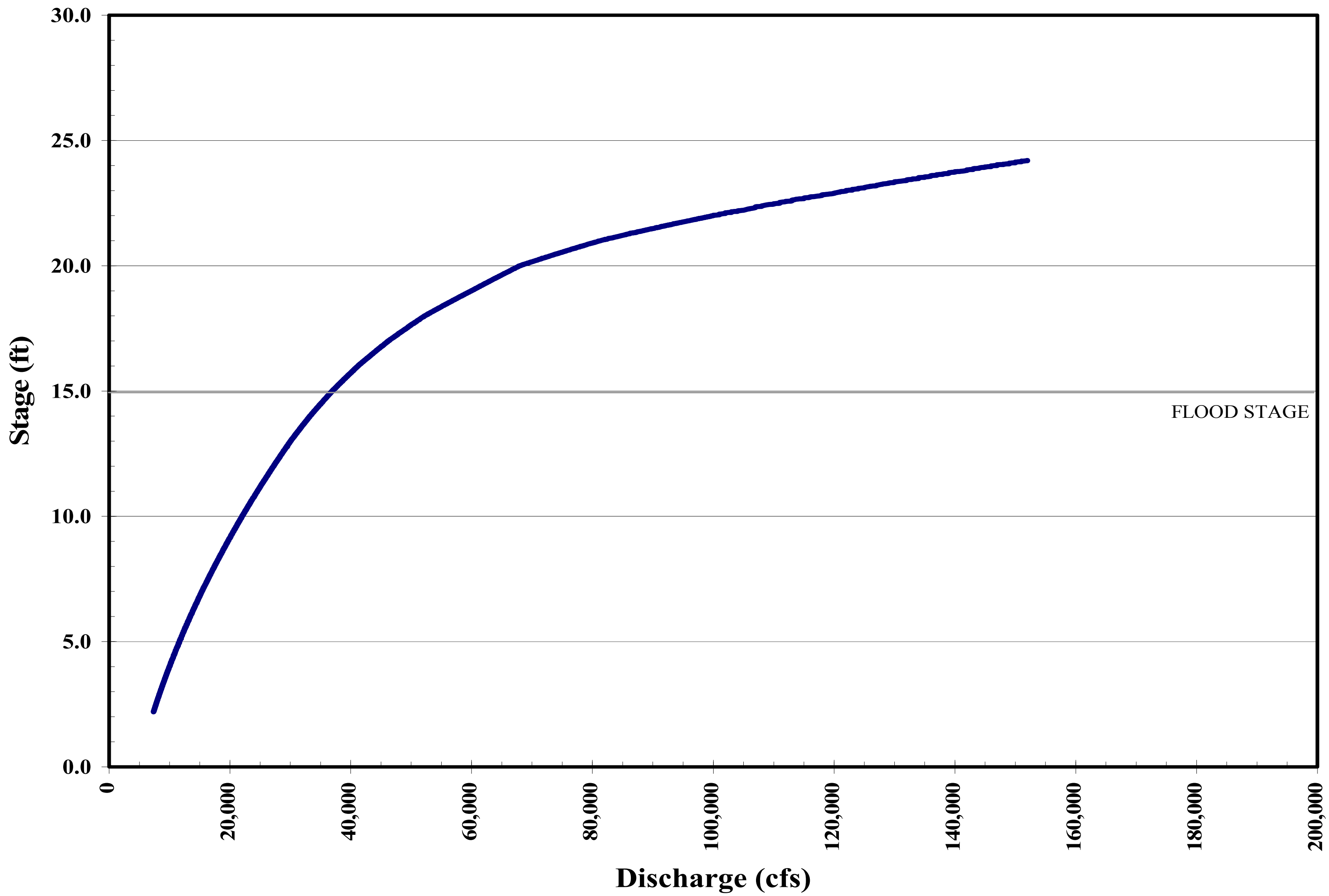

Figure 5-4. Stage-discharge rating curve for the Apalachicola River at the Blountstown gage (March 2004-March 2006).

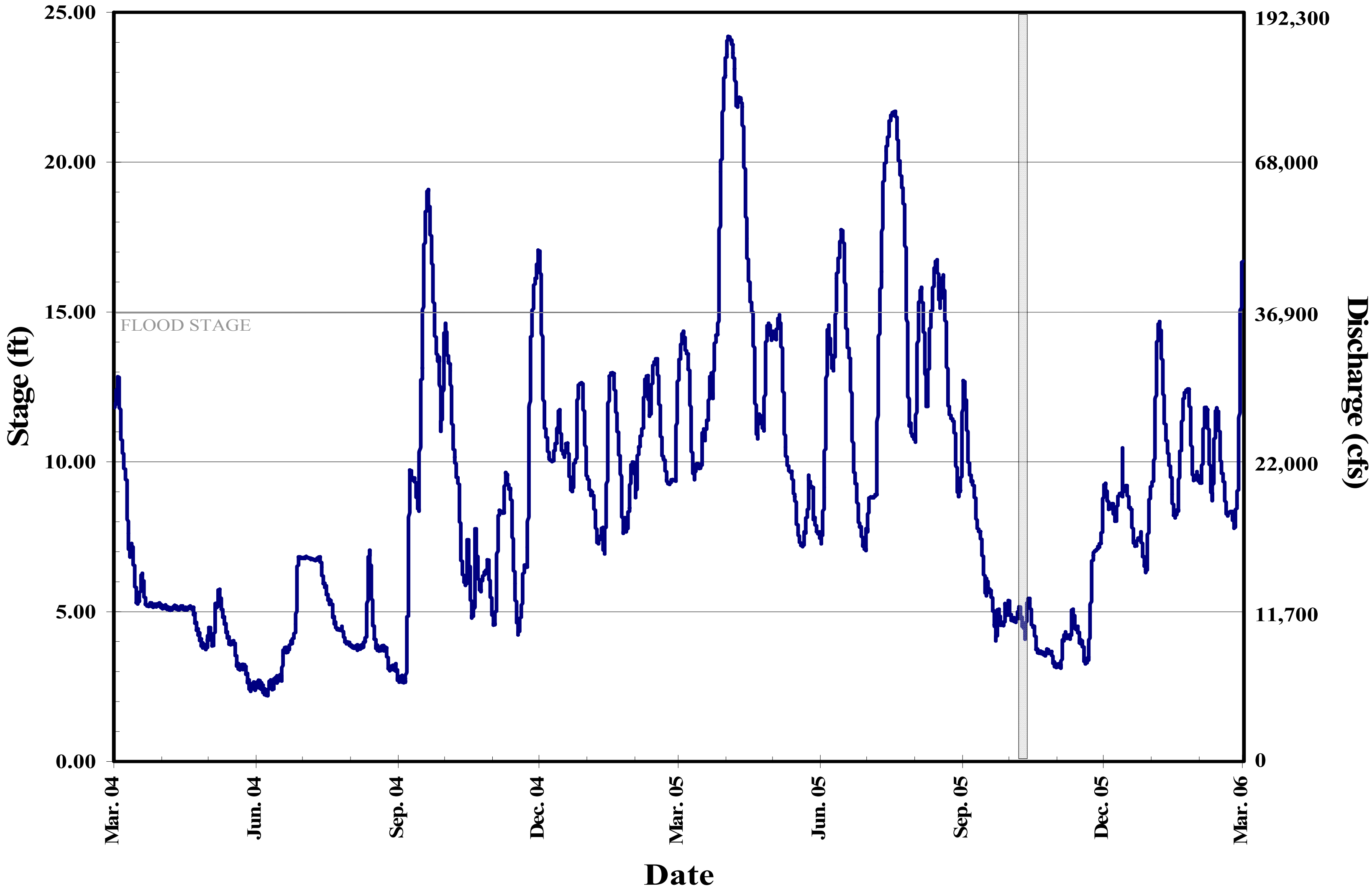

Figure 5-5. Stage-discharge hydrograph for the Apalachicola River at the Blountstown gage (March 2004-March 2006). Shaded box indicates the period of time when fieldwork was conducted.

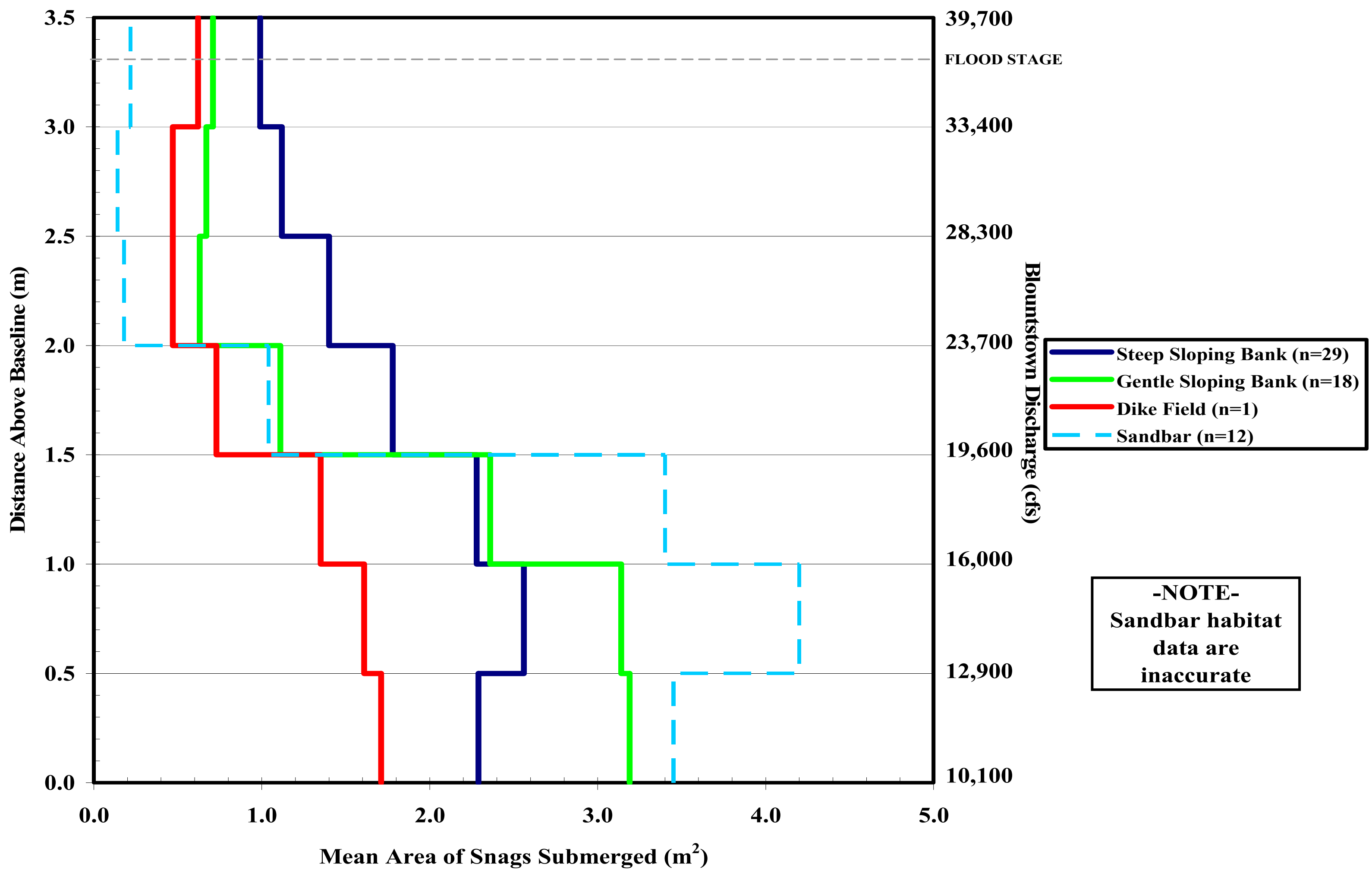

Figure 5-6. Bank habitat classification effects on mean area of snags submerged at 0.5-m distances above baseline. Curves generated from all data collected within the 55.3-km intensive study reach of the Apalachicola River. Snag areas submerged at 0.5-m distances above baseline are per 10.0-m of shoreline. Sandbar habitat data are inaccurate, and plotted with a dashed-line as a reference only. Data from these curves should not be extrapolated.

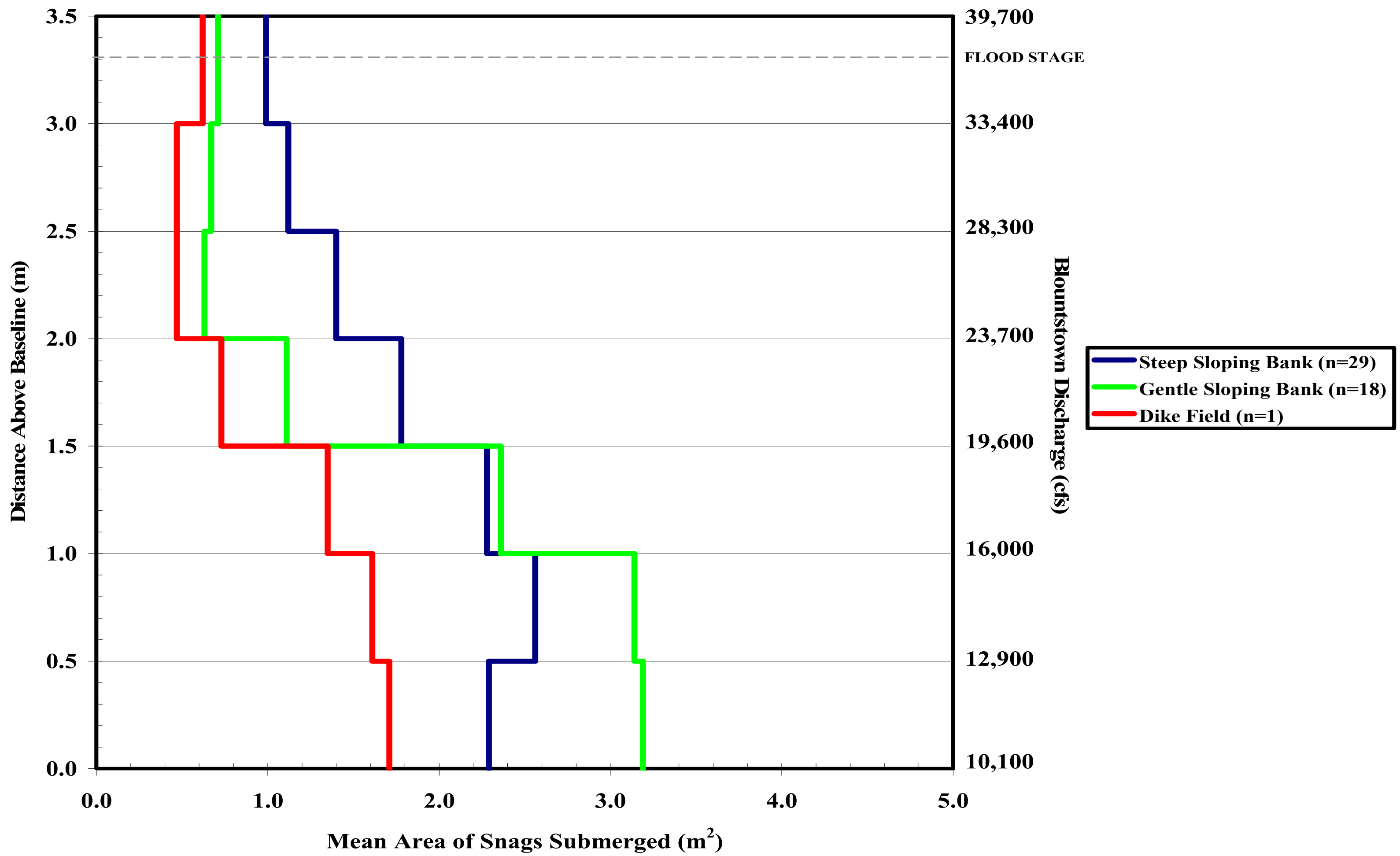

Figure 5-7. Non-sandbar bank habitat classification effects on mean area of snags submerged at 0.5-m distances above baseline. Curves generated from data collected within the 55.3-km intensive study reach of the Apalachicola River. Snag areas submerged at 0.5-m distances above baseline are per 10.0-m of shoreline. Data from these curves should not be extrapolated.

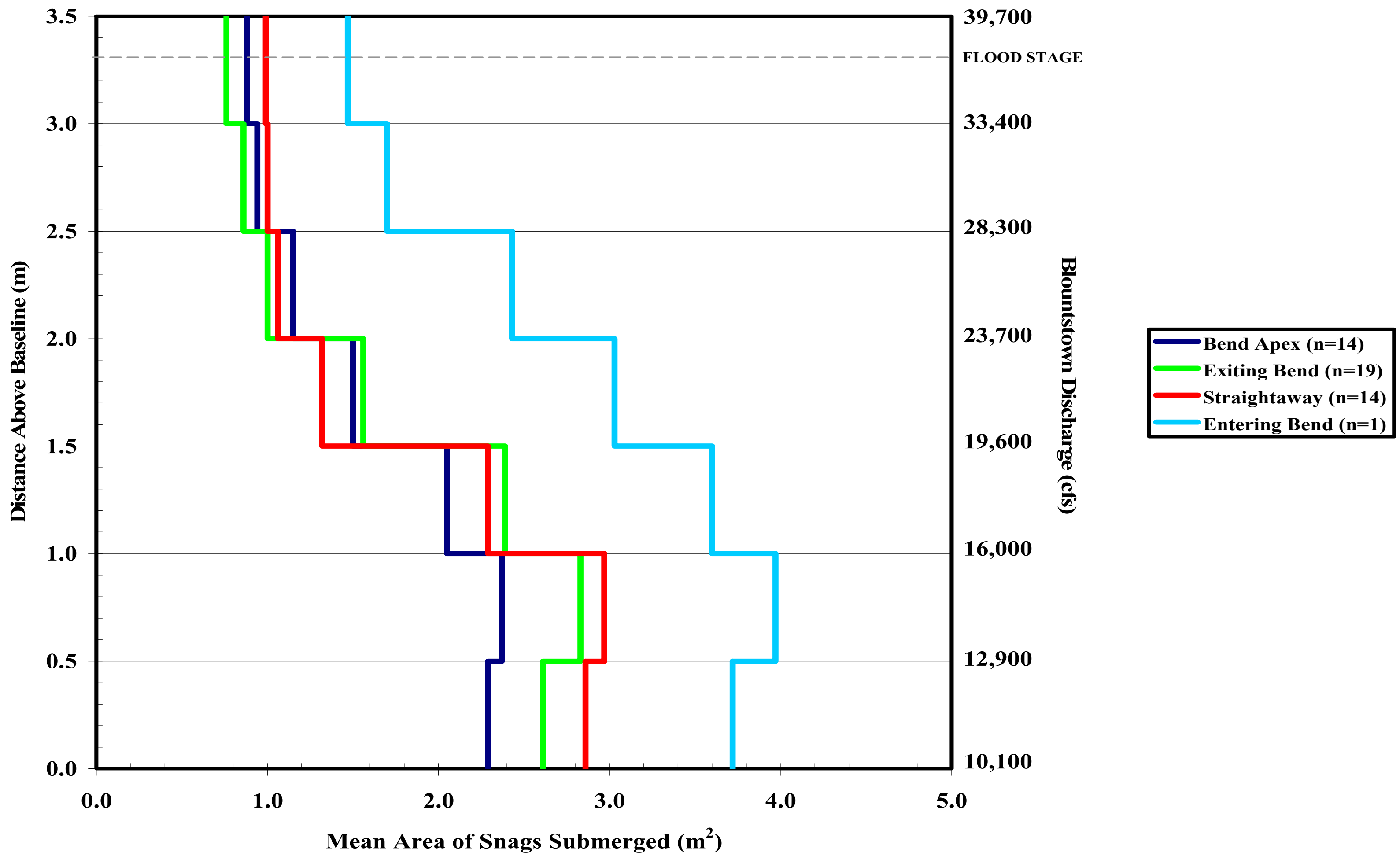

Figure 5-8. Geomorphic classification effects on mean area of snags submerged at 0.5-m distances above baseline. Curves generated from non-sandbar habitat data collected within the 55.3-km intensive study reach of the Apalachicola River. Snag areas submerged at 0.5-m distances above baseline are per 10.0-m of shoreline. Data from these curves should not be extrapolated.

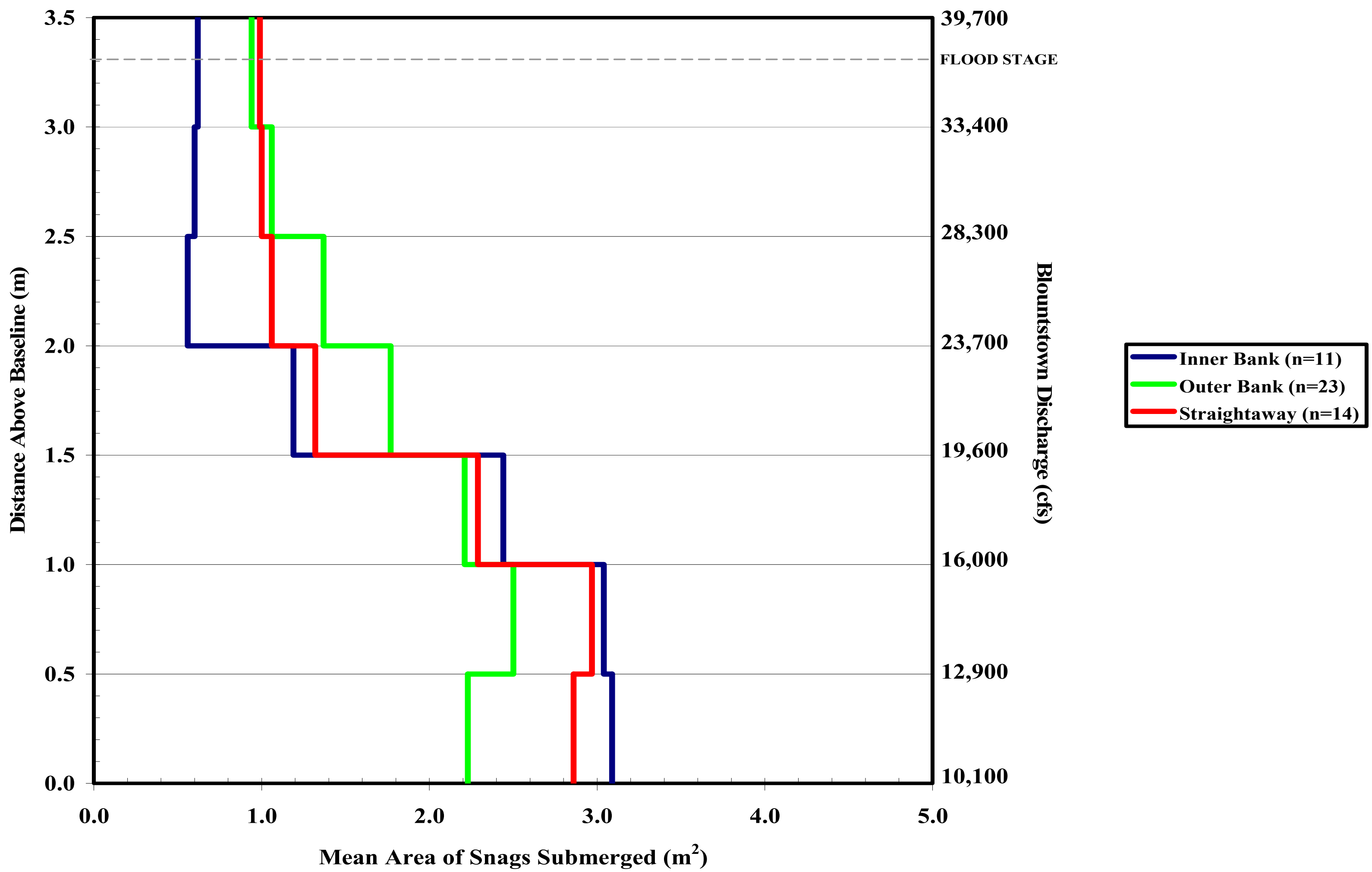

Figure 5-9. Effects of bank location within a river bend on mean area of snags submerged at 0.5-m distances above baseline. Curves generated from non-sandbar habitat data collected within the 55.3-km intensive study reach of the Apalachicola River. Snag areas submerged at 0.5-m distances above baseline are per 10.0-m of shoreline. Data from these curves should not be extrapolated.

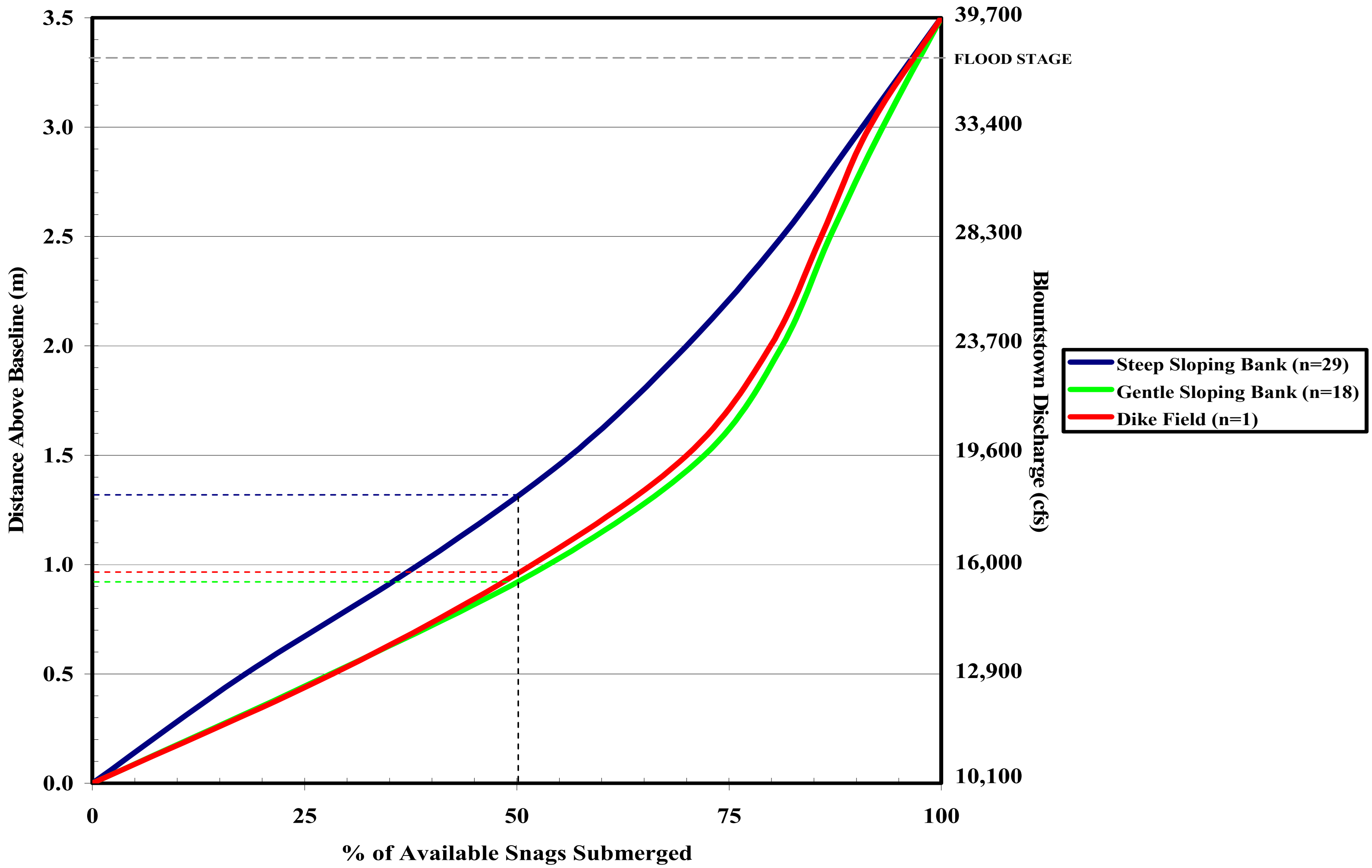

Figure 5-10. Bank habitat classification effects on percentage of total available snags submerged in relation to distance above baseline. Curves generated from non-sandbar habitat data collected within the 55.3-km intensive study reach of the Apalachicola River. Dotted lines correspond to the location where 50% of the total available snag material would be submerged.

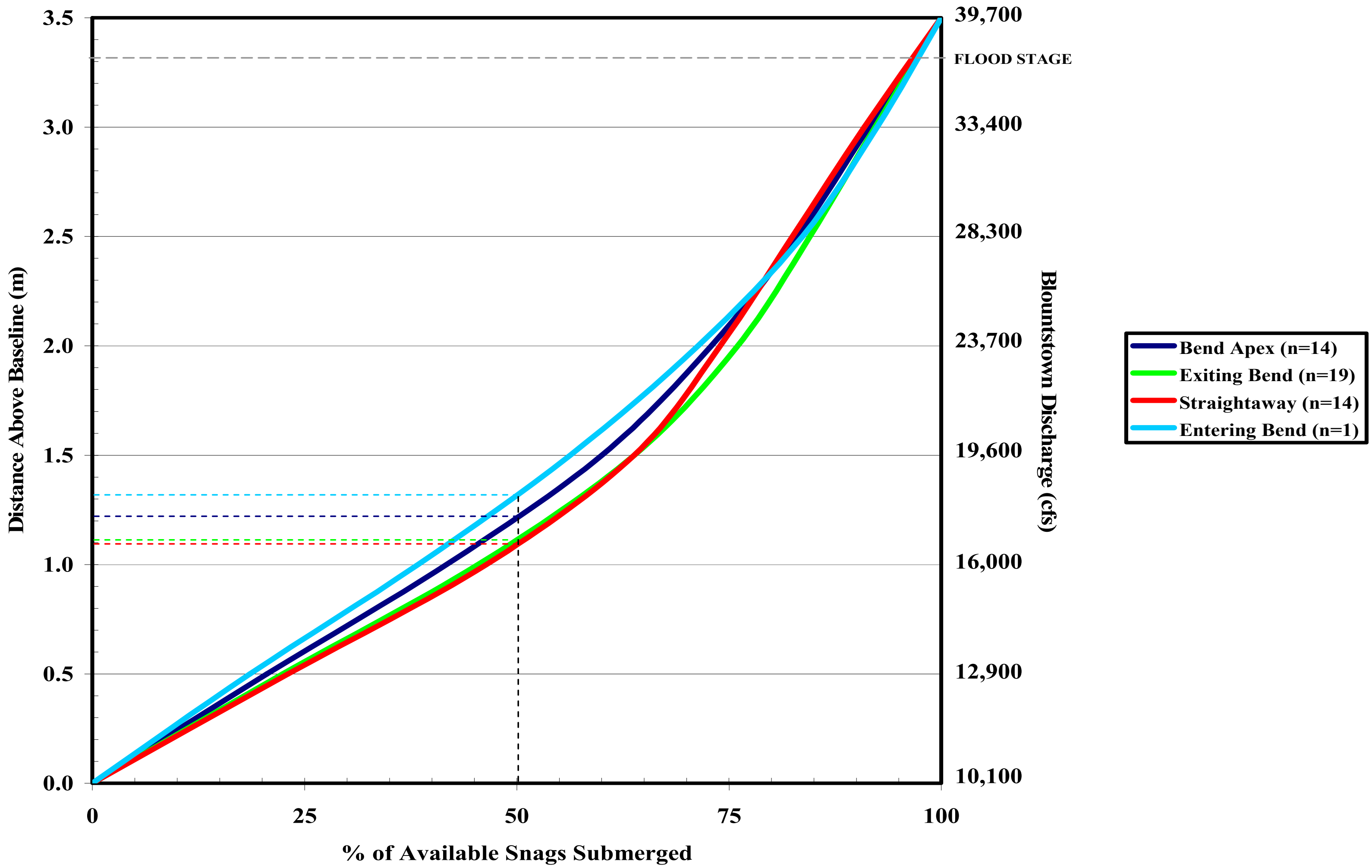

Figure 5-11. Geomorphic classification effects on percentage of total available snags submerged in relation to distance above baseline. Curves generated from non-sandbar habitat data collected within the 55.3-km intensive study reach of the Apalachicola River. Dotted lines correspond to the location where 50% of the total available snag material would be submerged.

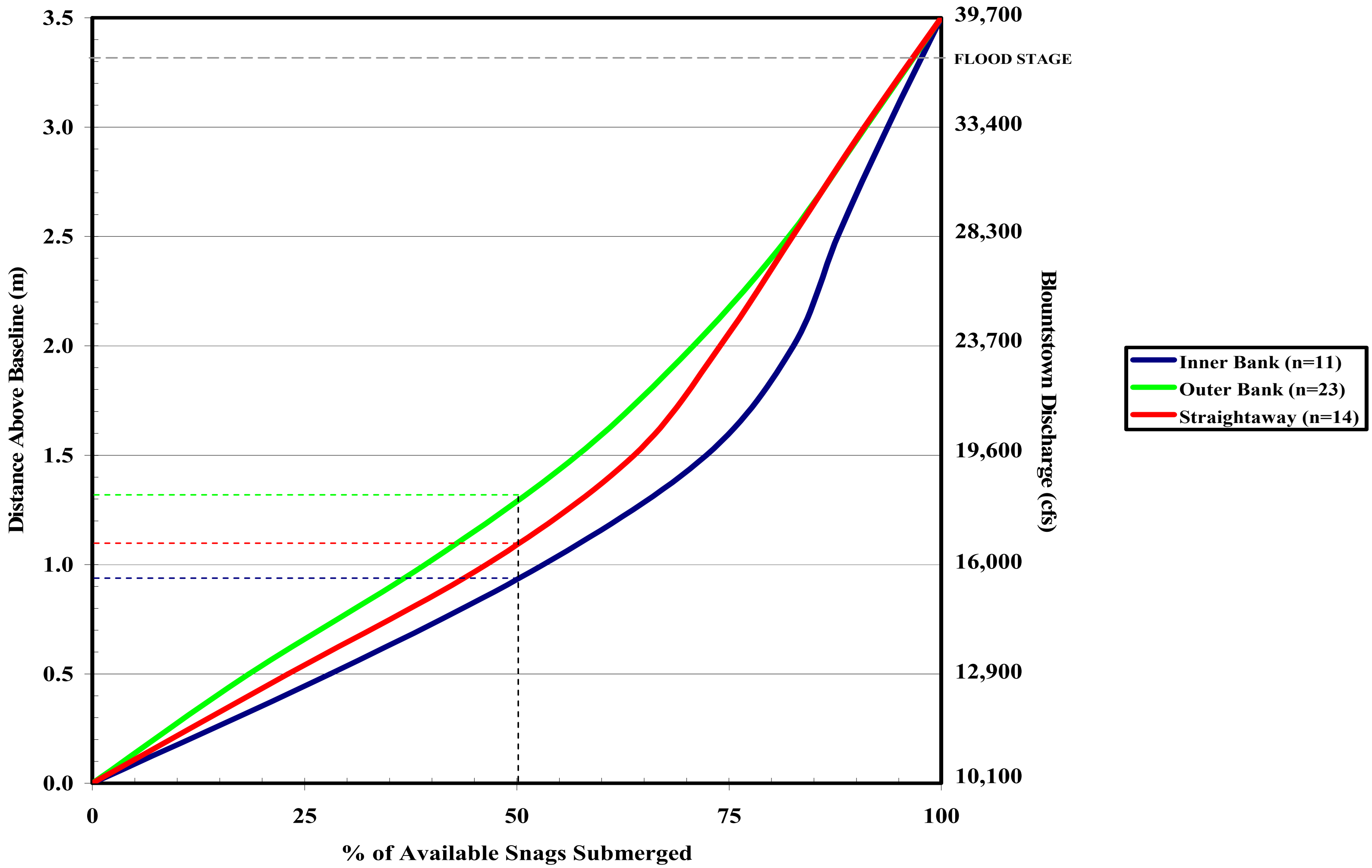

Figure 5-12. Effects of bank location within a river bend on percentage of total available snags submerged in relation to distance above baseline. Curves generated from non-sandbar habitat data collected within the 55.3-km intensive study reach of the Apalachicola River. Dotted lines correspond to the location where 50% of the total available snag material would be submerged.

APPENDIX A
SAMPLING STATION FIELD DATASHEET

USGS science for a changing world

Apalachicola River Snag Survey Field Datasheet

U.S. Geological Survey - Florida Integrated Science Center - 7920 NW 71st Street - Gainesville, Florida - 32653

Date:	Field #:	
Locality: *(Be sure to include River Mile)*		
Collectors:		Nearest Gage:
GPS Datum:	Waypoint Code:	Gage Reading:
Latitude:	Longitude:	UTM:

Habitat: steep bank (≥ 45°) gentle bank (< 45°) dike field sandbar

Channel Geomorphology: left descending bank right descending bank
straightaway entering bend bend apex exiting bend
If bend: inner bank outer bank

Diagram: *(Indicate large features, points of reference, aids to navigation, direction of flow, etc.)*

Photograph Information: *(List image time, DSCN range, camera used, etc.)*

Time	Range
1)	1)
2)	2)
3)	3)
4)	4)

Notes:

APPENDIX B
GUIDELINES FOR IMAGE OPTIMIZATION, PROCESSING, AND ANALYSIS OF DIGITAL PHOTOGRAPHS

ADOBE PHOTOSHOP:

1. *File* ► *Open* ► (filename.jpg) **[CTRL + 'O']**
   -'Discard the embedded profile (don't color manage)'
2. *Image* ► *Adjustments* ► *Invert* **[CTRL + 'I']**
3. *Image* ► *Mode* ► *Grayscale*
4. *Image* ► *Image Size* ► *Pixel Dimensions*
   -Width: '55' / 'Percent'
5. *File* ► *Save As* ► (filename.tif) **[CTRL + SHIFT + 'S']**
   -'No image compression'

IMAGE J:

1. *File* ► *Open* (filename.tif) **[CTRL + 'O']**
2. Adjust window size as needed
3. Select the magnifying glass tool and magnify the stadia rod to the height of the screen
   -**[mouse click = Zoom In**; **CTRL+ mouse click = Zoom Out]**
4. Select the line tool and stretch a vertical line the length of the stadia rod
5. *Analyze* ► *Set Scale*
   -Enter known distance (3.5) and unit of length (meter); check box 'global'
6. Zoom back out to 100% using magnifying glass tool
   -**[mouse click = Zoom In**; **CTRL+ mouse click = Zoom Out]**
7. Select the line tool and stretch a 10-m horizontal line along the waterline
   -**[CTRL + 'M']** to measure the line
   -Add or subtract length to the stretched out horizontal line using either of the boxes on the ends of the line
   -**[CTRL + 'M']** and adjust repeatedly until a 10-m length is achieved
8. On the stretched out horizontal line, grab the center box located at 5 m, and drag the whole line to the bottom of the stadia rod where the water and shoreline meet to create a right angle
   -**[CTRL + 'D']** to draw to line on the image at the waterline
9. *Analyze* ► *Clear Results*
   -'Do not save'
10. Select the magnifying glass tool and magnify the non-stadia rod end of the drawn-in line
    -**[mouse click = Zoom In**; **CTRL+ mouse click = Zoom Out]**
11. Using the line tool, stretch a vertical line upwards of at least 3.5-m length from the end of the drawn-in line
    -**[CTRL + 'M']** to measure the line
    -If the length is 3.5-m or longer, continue on; if the length is short, use the uppermost box of the vertical line to add length and measure again; when at least 3.5-m is achieved, continue on
    -**[CTRL + 'D']** to draw the vertical line on the image
12. Select the magnifying glass tool and magnify the stadia rod to the height of the screen
    -**[mouse click = Zoom In**; **CTRL+ mouse click = Zoom Out]**

13. Select the rectangle tool and using the stadia rod as a vertical guide, stretch a 0.5-m tall rectangle of any length
14. Use the hand tool, select the center box on each of the vertical edges of the rectangle and extend the rectangle to fit between the stadia rod and the drawn-in vertical line, which will create a rectangle with dimensions of 10-m x 0.5-m.
15. With the hand tool, use click and drag techniques anywhere within the rectangle to move it to the lowermost 0.5-m interval above the drawn-in horizontal line
16. *Image* ► *Adjust* ► *Threshold* **[CTRL + SHIFT + 'T']**
    -'Set' range from '0' to '128'; and 'Black and White'
17. *Analyze* ► *Analyze Particles*
    -Use the following settings: '1' / '999999' / '2' / 'Nothing'; and check the following boxes: 'Display Results' and 'Summarize'
18. In a spreadsheet program, record the total area value from the Summary window
19. Close the Summary window
20. Select 'Reset' on the Threshold window
21. Select the rectangle tool and click in the existing rectangle. Using the drawn in vertical line and stadia rod as guides, slide the rectangle up 0.5 m
22. Select 'Set' on the Threshold window and set range to '0' to '128'; and 'Black and White'
23. *Analyze* ► *Analyze Particles*
    -Use the following settings: '1' / '999999' / '2' / 'Nothing'; and check the following boxes: 'Display Results' and 'Summarize'
24. Repeat steps18 through 23 until all seven of the 10-m x 0.5-m sections are measured
25. In the Results window, select *File* ► *Save As* ► (filename.xls)

REFERENCE LIST

Ager, L. A., C. L. Mesing, and M. J. Hill. 1985. Fishery study -- Apalachicola River maintenance dredging disposal site evaluation program -- March 1985 -- Final report. State of Florida Game and Fresh Water Fish Commission, Miscellaneous Paper, Tallahassee, Florida.

Ager, L. A., C. L. Mesing, R. S. Land, M. J. Hill, M. J. Spelman, R. W. Rousseau, and K. Stone. 1987. Five year completion report -- fisheries ecology and dredging impacts on the Apalachicola River system -- 1981-1986. State of Florida Game and Fresh Water Fish Commission, Miscellaneous Paper, Tallahassee, Florida.

Anderson, N. H. and J. R. Sedell. 1979. Detritus processing by macroinvertebrates in stream ecosystems. Annual Review of Entomology 24:351-357.

Anderson, N. H., J. R. Sedell, L. M. Roberts, and F. J. Triska. 1978. The role of aquatic invertebrates in processing of wood debris in coniferous forest streams. American Midland Naturalist 100:64-82.

Andrus, C. W., B. A. Long, and H. A. Froehlich. 1988. Woody debris and its contribution to pool formation in a coastal stream 50 years after logging. Canadian Journal of Fisheries and Aquatic Sciences 45:2080-2086.

Angermeier, P. L. and J. R. Karr. 1984. Relationships between woody debris and fish habitat in a small warmwater stream. Transactions of the American Fisheries Society 113:716-726.

Aumen, N. G., P. J. Bottomley, G. M. Ward, and S. V. Gregory. 1983. Microbial decomposition of wood in streams: distribution of microflora and factors affecting [^{14}C] lignocellulose mineralization. Applied and Environmental Microbiology 46:1409-1416.

Bailey, G. R. 1970. A simplified method of sampling logging residue. Forestry Chronicle 46:288-294 .

Baillie, B. R. and T. R. H. Davies. 2002. Influence of large woody debris on channel morphology in native forest and pine plantation streams in the Nelson region, New Zealand. New Zealand Journal of Marine and Freshwater Research 36:763-774.

Baker, J. H., R. Y. Morita, and N. H. Anderson. 1983. Bacterial activity associated with the decomposition of woody substrates in a stream sediment. Applied and Environmental Microbiology 45:516-521.

Baldwin, D. S. and A. M. Mitchell. 2000. The effects of drying and re-flooding on the sediment and soil nutrient dynamics of lowland river--floodplain systems: a synthesis. Regulated Rivers: Research & Management 16:457-467.

Bash, J. S. and C. M. Ryan. 2002. Stream restoration and enhancement projects: is anyone monitoring? Environmental Management 29:877-885.

Beechie, T. J. and T. H. Sibley. 1997. Relationships between channel characteristics, woody debris, and fish habitat in northwestern Washington streams. Transactions of the American Fisheries Society 126:217-229.

Benke, A. C. 2001. Importance of flood regime to invertebrate habitat in an unregulated river--floodplain ecosystem. Journal of the North American Benthological Society 20:225-240.

Benke, A. C., R. L. Henry, D. M. Gillespie, and R. J. Hunter. 1985. Importance of snag habitat for animal production in southeastern streams. Fisheries 10:8-13.

Benke, A. C., T. C. Van Arsdall, D. M. Gillespie, and F. K. Parrish. 1984. Invertebrate productivity in a subtropical blackwater river: the importance of habitat and life history. Ecological Monographs 54:25-63.

Benke, A. C. and J. B. Wallace. 1990. Wood dynamics in Coastal Plain backwater streams. Canadian Journal of Fisheries and Aquatic Sciences 47:92-99.

Berg, N., A. Carlson, and D. Azuma. 1998. Function and dynamics of woody debris in stream reaches in the central Sierra Nevada, California. Canadian Journal of Fisheries and Aquatic Sciences 55:1807-1820.

Bilby, R. E. 1981. Role of organic debris dams in regulating the export of dissolved and particulate matter from a forested watershed. Ecology 62:1234-1243.

Bilby, R. E. 2003. Decomposition and nutrient dynamics of wood in streams and rivers. Pages 135-148 *in* S. V. Gregory, K. L. Boyer, and A. M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Bethesda, Maryland.

Bilby, R. E., J. T. Heffner, B. R. Fransen, J. W. Ward, and P. A. Bisson. 1999. Effects of immersion in water on deterioration of wood from five species of trees used for habitat enhancement projects. North American Journal of Fisheries Management 19:687-695.

Bilby, R. E. and G. E. Likens. 1980. Importance of organic debris dams in the structure and function of stream ecosystems. Ecology 61:1107-1113.

Bilby, R. E. and J. W. Ward. 1991. Characteristics and function of large woody debris in streams draining old-growth, clear-cut, and second-growth forests in southwestern Washington. Canadian Journal of Fisheries and Aquatic Sciences 48:2499-2508.

Bisson, P. A., R. E. Bilby, M. D. Bryant, C. A. Dolloff, G. B. Grette, R. A. House, M. L. Murphy, K. V. Koski, and J. R. Sedell. 1987. Large woody debris in forested streams in the Pacific Northwest: past, present, and future. Pages 143-190 *in* E. O. Salo and T. W. Cundy, editors. Streamside management: forestry and fishery interactions. Seattle, Washington.

Boling, R. H., E. D. Goodman, J. A. Van Sickle, J. O. Zimmer, K. W. Cummins, R. C. Petersen, and S. R. Reice. 1975. Toward a model of detritus processing in a woodland stream. Ecology 56:141-151.

Bovee, K. D. 1982. A guide to stream habitat analysis using the instream flow incremental methodology. US Fish and Wildlife Service, Instream Flow Informational Paper 12, Washington, D.C.

Bovee, K. D. 1986. Development and evaluation of habitat suitability criteria for use in the instream flow incremental methodology. US Fish and Wildlife Service, Biological Report 86(7), Washington, D.C.

Braccia, A. and D. P. Batzer. 2001. Invertebrates associated with woody debris in a southeastern U.S. forested floodplain wetland. Wetlands 21:18-31.

Braudrick, C. A., G. E. Grant, Y. Ishikawa, and H. Ikeda. 1997. Dynamics of wood transport in streams: a flume experiment. Earth Surface Processes and Landforms 22:669-683.

Brim Box, J. and J. D. Williams. 2000. Unionid mollusks of the Apalachicola Basin in Alabama, Florida, and Georgia. Bulletin Alabama Museum of Natural History 21:1-143.

Brown, T. K. 2002. Creating and maintaining wildlife, insect, and fish habitat structures in dead wood. US Department of Agriculture, Forest Service Technical Report PSW-GTR-181, Washington, D.C.

Carriker, R. R. 2000. Water wars: water allocation law and the Apalachicola-Chattahoochee-Flint River basin. Department of Food and Resource Economics, Florida Cooperative Extension Service, Institute of Food and Agricultural Sciences, University of Florida, Document FE 208, Gainesville, Florida.

Chastain, C. A., S. W. Golladay, and T. K. Muenz. 2006. Distribution of unionid mussels in tributaries of the lower Flint River, southwestern Georgia: an examination of current and historical trends. Four pages *in* K. J. Hatcher, editor. Proceedings of the 2005 Georgia Water Resources Conference. University of Georgia, Institute of Ecology, Athens, Georgia.

Cherry, J. and R. L. Beschta. 1989. Coarse woody debris and channel morphology: a flume study. Water Resources Bulletin 25:1031-1036.

de Vries, P. G. 1974. Multi-stage line intersect sampling. Forest Science 20:129-133.

Dellapenna, J. W. 2004. Special challenges to water markets in riparian states. Georgia State University Law Review 21:1-8.

Diehl, T. H. and B. A. Bryan. 1993. Supply of large woody debris in a stream channel. Pages 1055-1060 *in* H. W. Shen, S. T. Su, and F. Wen, editors. Proceedings of the 1992 American Society of Civil Engineers Hydraulic Engineering Conference. American Society of Civil Engineers, San Francisco.

Díez, J. R., A. Elosegi, and J. Pozo. 2001. Woody debris in north Iberian streams: influence of geomorphology, vegetation, and management. Environmental Management 28:687-698.

Dolloff, C. A. 1994. Large woody debris, fish habitat, and historical land use. Pages 130-138 *in* Proceedings of the workshop on coarse woody debris in southern forests: effects on biodiversity. US Department of Agriculture, Forest Service General Technical Report SE-94, Washington, D.C.

Dolloff, C. A. and J. R. Webster. 2000. Particulate organic contributions from forests to streams: debris isn't so bad. Pages 125-138 *in* E. S. Verry, J. W. Hornbeck, and C. A. Dolloff, editors. Riparian management in forests of the continental eastern United States. CRC Press LLC, Boca Raton, Florida.

Drury, D. M. and W. E. Kelso. 2000. Invertebrate colonization of woody debris in coastal plain streams. Hydrobiologia 434:63-72.

Dye, C. W. and R. B. Frydenborg. 1999. Investigations of potential habitat effects of deadhead logging in north Florida. Florida Department of Environmental Protection, Miscellaneous Reports and Studies, Tallahassee, Florida.

Elder, J. F. and D. J. Cairns. 1982. Production and decomposition of forest litter fall on the Apalachicola River flood plain, Florida. US Geological Survey, Water-Supply Paper 2196-B, Alexandria, Virginia.

Elder, J. F., S. D. Flagg, and H. C. Mattraw. 1988. Hydrology and ecology of the Apalachicola River, Florida: a summary of the river quality assessment. US Geological Survey, Water-Supply Paper 2196-D, Alexandria, Virginia.

Erskine, W. D. 1994. River response to accelerated soil erosion in the Glenelg River catchment, Victoria. Australian Journal of Soil and Water Conservation 7:39-47.

Estevez, E. D. and J. Sprinkel. 2000. Dispersion and abundance of wood habitat of the Santa Fe and lower Suwannee River systems in relation to flow and flow-related variables - A report of findings. Mote Marine Laboratory, Technical Report Number 646, Sarasota, Florida.

FFWCC (Florida Fish and Wildlife Conservation Commission). 2008. “Welcome to Florida - fishing capital of the world.” Informational Publication. Available: http://floridafisheries.com/pdf/Fishing%20AD.pdf. (last accessed: March 2008).

Fitzgerald, D. G., B. Zhu, S. B. Hoskins, D. E. Haddad, K. N. Green, L. G. Rudstam, and E. L. Mills. 2006. Quantifying submerged aquatic vegetation using aerial photograph interpretation: application in studies assessing fish habitat in freshwater ecosystems. Fisheries 31:61-73.

Flug, M., H. Seitz, and J. Scott. 1998. Measurement of stream channel habitat using sonar. Regulated Rivers: Research & Management 14:511-517.

Gibson, C. A., J. L. Meyer, N. L. Poff, L. E. Hay, and A. Georgakakos. 2005. Flow regime alterations under changing climate in two river basins: implications for freshwater ecosystems. River Research and Applications 21:849-864.

Gippel, C. J. 1995. Environmental hydraulics of large woody debris in streams and rivers. Journal of Environmental Engineering 121:388-395.

Gippel, C. J., B. L. Finlayson, and I. C. O'Neill. 1996. Distribution and hydraulic significance of large woody debris in a lowland Australian river. Hydrobiologia 318:179-194.

Gippell, C. J., I. C. O'Neill, and B. L. Finlayson. 1992. The hydraulic basis of snag management. Center for Environmental Applied Hydrology, University of Melbourne, Melbourne, Australia.

Golladay, S. W., P. Gagnon, M. Kearns, J. M. Battle, and D. W. Hicks. 2004. Response of freshwater mussel assemblages (Bivalvia: Unionidae) to a record drought in the Gulf Coastal Plain of southwestern Georgia. Journal of the North American Benthological Society 23:494-506.

Golladay, S. W. and J. M. Battle. 2006. Wood debris recruitment from differing riparian landforms in a Gulf Coastal Plain stream: the role of floods. Four pages *in* K. J. Hatcher, editor. Proceedings of the 2005 Georgia Water Resources Conference. University of Georgia, Institute of Ecology, Athens, Georgia.

Gore, J. A. 1985. The restoration of rivers and streams -- theories and experience. Butterworth Publishers, Boston.

Gregory, S. V., K. L. Boyer, and A. M. Gurnell, editors. 2003. The ecology and management of wood in world rivers, Symposium 37. American Fisheries Society, Bethesda, Maryland.

Gurnell, A. M. 2003. Wood storage and mobility. Pages 75-92 *in* S. V. Gregory, K. L. Boyer, and A. M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Bethesda, Maryland.

Hamilton, K. and E. P. Bergersen. 1984. Methods to estimate aquatic habitat variables. Colorado Cooperative Fishery Research Unit, Colorado State University, Fort Collins, Colorado.

Harmon, M. E., J. F. Franklin, F. J. Swanson, P. Sollins, S. V. Gregory, J. D. Lattin, N. H. Anderson, S. P. Cline, N. G. Aumen, J. R. Sedell, G. W. Lienkaemper, K. Cromack, and K. W. Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. Pages 133-302 *in* A. MacFadyen and E. D. Ford, editors. Advances in Ecological Research. Academic Press, Harcourt Brace Jovanovich, London.

Hart, C. W. and S. L. H. Fuller. 1974. Pollution ecology of freshwater invertebrates. Academic Press, New York.

Heard, W. H. 1977. Freshwater Mollusca of the Apalachicola drainage. Pages 20-21 *in* R. J. Livingston and E. A. Joyce, editors. Proceedings of the Conference on the Apalachicola Drainage System. Florida Marine Research Institute, Publication Number 26, St. Petersburg, Florida.

Helfman, G. S. 2007. Fish conservation: a guide to understanding and restoring global aquatic biodiversity and fishery resources. Island Press, Washington, D.C.

Hilderbrand, R. H., A. D. Lemly, C. A. Dolloff, and K. L. Harpster. 1998. Design considerations for large woody debris placement in stream enhancement projects. North American Journal of Fisheries Management 18:161-167.

Hoehn, T. 2001. The Apalachicola River today. Florida Wildlife 55:12.

Hrodey, P. J. and T. M. Sutton. 2008. Fish community responses to half-log additions in warmwater streams. North American Journal of Fisheries Management 28:70-80.

Hynes, H. B. N. 1975. The stream and its valley. Verhandlungen der Internationalen Vereinigung fur Theoretische und Angewandte Limnologie 19:1-15.

Junk, W. J., P. B. Bayley, and R. E. Sparks. 1989. The flood pulse concept in river-floodplain systems. Special Publication, Canadian Journal of Fisheries and Aquatic Sciences 106:110-127.

Keller, E. A. and F. J. Swanson. 1979. Effects of large organic material on channel form and fluvial processes. Earth Surface Processes 4:361-380.

Kemp, W. M. and W. R. Boynton. 2004. Productivity, trophic structure. and energy flow in the steady-state ecosystems of Silver Springs, Florida. Ecological Modeling 178:43-49.

Kiffney, P. M. and P. Roni. 2007. Relationships between productivity, physical habitat, and aquatic invertebrate and vertebrate populations of forest streams: an information-theoretic approach. Transactions of the American Fisheries Society 136:1088-1103.

Kirkland, S. H. 2001. The steamboat era on the Apalachicola, Chattahoochee, and Flint rivers. Florida Wildlife 55:9-11.

Knighton, D. 1998. Fluvial forms & processes: a new perspective. Arnold Publishers, London.

Leitman, H. M. 1984. Forest map and hydrologic conditions, Apalachicola River floodplain, Florida. US Geological Survey, Hydrologic Investigations Atlas HA-672, Alexandria, Virginia.

Leitman, H. M., M. R. Darst, and J. J. Nordhaus. 1991. Fishes in the forested flood plain of the Ochlockonee River, Florida, during flood and drought conditions. US Geological Survey, Water-Resources Investigations Report 90-4202, Tallahassee, Florida.

Leitman, H. M., J. E. Sohm, and M. A. Franklin. 1983. Wetland hydrology and tree distribution of the Apalachicola River flood plain, Florida. US Geological Survey, Water-Supply Paper 2196-A, Alexandria, Virginia.

Leitman, S. F., L. A. Ager, and C. L. Mesing. 1991. The Apalachicola experience: environmental effects of physical modifications for navigation purposes. Pages 223-246 *in* R. J. Livingston, editor. The rivers of Florida. Springer-Verlag Inc., New York.

Leitman, S F.. 2005. An evaluation of the availability of the federally authorized navigation channel on the Apalachicola River without maintenance dredging. Apalachicola Bay and Riverkeeper, Apalachicola, Florida.

Lemly, A. D. and R. H. Hilderbrand. 2000. Influence of large woody debris on stream insect communities and benthic detritus. Hydrobiologia 421:179-185.

Lienkaemper, G. W. and F. J. Swanson. 1987. Dynamics of large woody debris in streams in old-growth Douglas-fir forests. Canadian Journal of Forest Research 17:150-156.

Light, H. M., M. R. Darst, and J. W. Grubbs. 1998. Aquatic habitats in relation to river flow in the Apalachicola River floodplain, Florida. US Geological Survey, Professional Paper 1594, Reston, Virginia.

Light, H. M., K. R. Vincent, M. R. Darst, and F. D. Price. 2006. Water-level decline in the Apalachicola River, Florida, from 1954 to 2004, and effects on floodplain habitats. US Geological Survey, Scientific Investigations Report 2006-5173, Reston, Virginia.

Livingston, R. J. 1984. Trophic response of fishes to habitat variability in coastal seagrass systems. Ecology 65:1258-1275.

Livingston, R. J., F. G. Lewis, G. C. Woodsum, X. Niu, B. Galperin, W. Huang, J. D. Christensen, M. E. Monaco, T. A. Battista, C. J. Klein, R. L. Howell, and G. L. Ray. 2000. Modeling oyster population response to variation in freshwater input. Estuarine, Coastal, and Shelf Science 50:655-672.

Lobb, M. D. and D. J. Orth. 1991. Habitat use by an assemblage of fish in a large warmwater stream. Transactions of the American Fisheries Society 120:65-78.

Macdonald, A. and E. A. Keller. 1987. Stream channel response to the removal of large woody debris, Larry Damm Creek, northwestern California. Pages 405-406 *in* R. L. Beschta, T. Blinn, G. E. Grant, F. J. Swanson, and G. G. Ice, editors. Erosion and sedimentation in the Pacific Rim -- Proceedings of the Corvalis symposium. International Association of Hydrological Sciences, Publication No. 165, Washington, D.C.

Mattraw, H. C. and J. F. Elder. 1984. Nutrient and detritus transport in the Apalachicola River, Florida. US Geological Survey, Water-Supply Paper 2196-C, Alexandria, Virginia.

McCuen, R. H. 2005. Hydrologic analysis and design. 3rd Ed. Prentice Hall, Inc., Upper Saddle River, New Jersey.

McTaggart, S. T. and T. A. Nelson. 2003. Composition and demographics of beaver (*Castor canadensis*) colonies in central Illinois. American Midland Naturalist 150:139-150.

Means, D. B. 1977. Aspects of the significance to terrestrial vertebrates of the Apalachicola River drainage basin, Florida. Pages 37-67 *in* R. J. Livingston and E. A. Joyce, editors. Proceedings of the conference on the Apalachicola drainage system. Florida Marine Research Institute, Publication Number 26, St. Petersburg, Florida.

Meeter, D. A., R. J. Livingston, and G. Woodsum. 1979. Short and long-term hydrologic cycles of the Apalachicola drainage system with application to Gulf coastal populations. Pages 315-338 *in* R. J. Livingston, editor. Ecological processes in coastal and marine systems. Plenum Press, New York.

Milhous, R. T., D. L. Wegner, and T. Waddle. 1984. Users guide to the physical habitat simulation system (PHABSIM). US Fish and Wildlife Service, Instream Flow Information Paper No. 11, FWS/OBS-81/43, Fort Collins, Colorado.

Milhous, R. T. 1999. History, theory, use, and limitations of the physical habitat simulation system. Pages 1-25 *in* Proceedings of the 3rd international symposium on ecohydraulics. Utah State University Extension, Utah State University, Logan, Utah.

Milhous, R. T., M. A. Updike, and D. M. Schneider. 1989. Physical habitat simulation system reference manual -- version II. US Fish and Wildlife Service, Biological Report 89(16), Washington, D.C.

Mitsch, W. J. and J. G. Gosselink. 2000. Wetlands. John Wiley & Sons, Inc., New York.

Molles, M. C. 1982. Trichopteran communities of streams associated with aspen and conifer forests: long-term structural change. Ecology 63:1-6.

Mueller, E. A. 1990. Perilous journeys: a history of steamboating on the Chattahoochee, Apalachicola, and Flint rivers, 1828-1928. Historic Chattahoochee Commission, Eufala, Alabama.

Murphy, M. L. and K. V. Koski. 1989. Input and depletion of woody debris in Alaska streams and implications for streamside management. North American Journal of Fisheries Management 9:427-436 .

Myers, R. L. and J. J. Ewel, editors. 1990. Ecosystems of Florida. University of Central Florida Press, Orlando, Florida.

Myers, V. B. 1977. Aspects of nutrient limitation of phytoplankton productivity in the Apalachicola Bay system. Pages 68-74 *in* R. J. Livingston and E. A. Joyce, editors. Proceedings of the conference on the Apalachicola drainage system. Florida Marine Research Institute, Publication Number 26, St. Petersburg, Florida.

Naiman, R. J. and J. R. Sedell. 1979. Relationships between metabolic parameters and stream order in Oregon. Canadian Journal of Fisheries and Aquatic Sciences 37:834-847.

Nilsen, H. C. and R. W. Larimore. 1973. Establishment of invertebrate communities on log substrates in the Kaskaskia River, Illinois. Ecology 54:366-374.

Oesterling, M. L. and G. L. Evink. 1977. Relationship between Florida's blue crab population and Apalachicola Bay. Pages 101-121 *in* R. J. Livingston and E. A. Joyce, editors. Proceedings of the conference on the Apalachicola drainage system. Florida Marine Research Institute, Publication Number 26, St. Petersburg, Florida.

Osborne, L. L. and D. A. Kovaic. 1993. Riparian vegetated buffer strips in water quality restoration and stream management. Freshwater Biology 29:243-258.

Peterson, J. T. and P. B. Bayley. 1993. Colonization rates of fishes in experimentally defaunated warmwater streams. Transactions of the American Fisheries Society 122:199-207.

Piégay, H. 2003. Dynamics of wood in large rivers. Pages 109-134 *in* S. V. Gregory, K. L. Boyer, and A. M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Bethesda, Maryland.

Pollock, M. M., M. Heim, and D. Werner. 2003. Hydrologic and geomorphic effects of beaver dams and their influence on fishes. Pages 213-234 *in* S. V. Gregory, K. L. Boyer, and A. M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Bethesda, Maryland.

Postel, S. L. and B. D. Richter. 2003. Rivers for life: managing water for people and nature. Island Press, Washington, D.C.

Rasband, W.S. 2002. Image J – image processing and analysis in Java. US National Institutes of Health, Bethesda, Maryland. Available: http://rsb.info.nih.gov/ij/. (last accessed: March 2008).

Ray, D. 1999a. Ecological and morphological significance of old growth deadhead logs in the Apalachicola River. Florida Department of Environmental Protection, Unpublished Bioassessment Report, Tallahassee, Florida.

Ray, D. 1999b. Ecological and morphological significance of old growth deadhead logs in the Blackwater River. Florida Department of Environmental Protection, Unpublished Bioassessment Report, Tallahassee, Florida.

Ray, D. 1999c. Ecological and morphological significance of old growth deadhead logs in the Chipola River. Florida Department of Environmental Protection, Unpublished Bioassessment Report, Tallahassee, Florida.

Ray, D. 1999d. Ecological and morphological significance of old growth deadhead logs in the Yellow River. Florida Department of Environmental Protection, Unpublished Bioassessment Report, Tallahassee, Florida.

Rayner, A. D. M. and N. K. Todd. 1979. Population and community structure and dynamics of fungi in decaying wood. Advances in Botanical Research 7:333-420.

Reich, M., J. L. Kershner, and R. C. Wildman. 2003. Restoring streams with large wood: a synthesis. Pages 355-366 *in* S. V. Gregory, K. L. Boyer, and A. M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Bethesda, Maryland.

Richter, B. D., J. V. Baumgartner, R. Wigington, and D. P. Braun. 1997. How much water does a river need? Freshwater Biology 37:231-249.

Richter, B. D., R. Mathews, D. L. Harrison, and R. Wigington. 2003. Ecologically sustainable water management: managing river flows for ecological integrity. Ecological Applications 13:203-224.

Ridler, T. W. and S. Calvard. 1978. Picture thresholding using an iterative selection method. IEEE Transactions on Systems, Man, and Cybernetics SMC-8:630-632.

Robertson, A. I., P. E. Bacon, and G. Heagney. 2001. The responses of floodplain primary production to flood frequency and timing. Journal of Applied Ecology 38:126-136.

Ross, S. T. and J. A. Baker. 1983. The response of fishes to periodic spring floods in a southeastern stream. American Midland Naturalist 109:1-14.

Ruhl, J. B. 2005. Water wars, eastern style: divvying up the Apalachicola-Chattahoochee-Flint river basin. Journal of Contemporary Water Research & Education 131:47-54.

Seabrook, C. 2002. "Atlanta guzzling water metro thirst exceeds predictions". The Atlanta Journal-Constitution. Section A, 15 May.

Sherk, G. W. 2005. The management of interstate water conflicts in the 21st century: is it time to call uncle? New York University Environmental Law Journal 12:764-827.

Smock, L. A., E. Gilinsky, and D. L. Stoneburner. 1985. Macroinvertebrate production in a southeastern United States blackwater stream. Ecology 66:1491-1503.

Snowden, B. L. 2005. Bargaining in the shadow of uncertainty: understanding the failure of the ACF and ACT Compacts. New York University Environmental Law Journal 13:134-196.

Southwick, R. and T. Allen. 2003. The 2001 economic benefits of watchable wildlife recreation in Florida. Report by Southwick Associates, Inc. to the Florida Fish and Wildlife Conservation Commission, Tallahassee, Florida.

Spänhoff, B. and E. I. Meyer. 2004. Breakdown rates of wood in streams. Journal of the North American Benthological Society 23: 189-197.

Sparks, R. E. 1995. Need for ecosystem management of large rivers and their floodplains. BioScience 45:168-182.

Stiling, P. 1999. Ecology: theories and applications. 3rd Ed. Prentice Hall, Inc., Upper Saddle River, New Jersey.

Struhs, D. B. and F. P. Mainella. 2000. Outdoor recreation in Florida: Florida's statewide comprehensive outdoor recreation plan. Florida Department of Environmental Protection, Tallahassee, Florida.

Swanson, F. J. 2003. Wood in rivers: a landscape perspective. Pages 299-314 *in* S. V. Gregory, K. L. Boyer, and A. M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Bethesda, Maryland.

Tank, J. L. and J. R. Webster. 1998. Interaction of substrate and nutrient availability on wood biofilm processes in streams. Ecology 79:2168-2179.

Thorp, J. H., E. M. McEwan, M. F. Flynn, and F. R. Hauer. 1985. Invertebrate colonization of submerged wood in a cypress-tupelo swamp and blackwater stream. American Midland Naturalist 113:56-68.

Triska, F. J. and K. Cromack. 1980. The role of wood debris in forests and streams. Pages 171-190 *in* R. H. Waring, editor. Forests: fresh perspectives from ecosystem analysis. Oregon State University Press, Corvallis, Oregon.

USFWS and USCB (US Fish and Wildlife Service and US Census Bureau). 2002. The 2001 national survey of fishing, hunting, and wildlife-associated recreation. US Department of the Interior and US Department of Commerce, Report FHW/01-NAT, Washington, D.C.

Van Wagner, C. E. 1968. The line intersect method in forest fuel sampling. Forest Science 14:20-26.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130-137.

Wallace, J. B. and A. C. Benke. 1984. Quantification of wood habitat in subtropical coastal plain streams. Canadian Journal of Fisheries and Aquatic Sciences 41:1643-1652.

Wallerstein, N. P. 2003. Dynamic model for constriction scour caused by large woody debris. Earth Surface Processes and Landforms 28:49-68.

Wang, H., W. Huang, M. A. Harwell, L. Edmiston, E. Johnson, P. Hsieh, K. Milla, J. Christensen, J. Stewart, and X. Liu. 2008. Modeling oyster growth rate by coupling oyster population and hydrodynamic models for Apalachicola Bay, Florida, USA. Ecological Modeling 211:77-89.

Ward, G. M. and N. G. Aumen. 1986. Woody debris as a source of fine particulate organic matter in coniferous forest stream ecosystems. Canadian Journal of Fisheries and Aquatic Sciences 43:1635-1642.

Ward, J. V., K. Tockner, and F. Schiemer. 1999. Biodiversity of floodplain river ecosystems: ecotones and connectivity. Regulated Rivers: Research & Management 15:125-139.

Warren, W. G. and P. F. Olsen. 1964. A line intersect technique for assessing logging waste. Forest Science 10:267-276.

Webb, A. A. and W. D. Erskine. 2003. Distribution, recruitment, and geomorphic significance of large woody debris in an alluvial forest stream: Tonghi Creek, southeastern Australia. Geomorphology 51:109-126.

Whiles, M. R. and J. B. Wallace. 1997. Leaf litter decomposition and macroinvertebrate communities in headwater streams draining pine and hardwood catchments. Hydrobiologia 353:107-119.

Williams, J. D., M. L. Warren, K. S. Cummings, J. L. Harris, and R. J. Neves. 1993. Conservation status of freshwater mussels of the United States and Canada. Fisheries 18:6-22.

Wilson, L. 1973. Variations in mean annual sediment yield as a function of mean annual precipitation. American Journal of Science 273:335-349.

Woldendorp, G., R. J. Keenan, S. Barry, and R. D. Spencer. 2004. Analysis of sampling methods for coarse woody debris. Forest Ecology and Management 198:133-148.

Zimmerman, R. C., J. C. Goodlett, and G. H. Comer. 1967. The influence of vegetation on channel form of small streams. Pages 255-275 *in* Symposium on River Morphology. International Association of Scientific Hydrology, Publication Number 75, Gentbrugge, Belgium.

BIOGRAPHICAL SKETCH

Destined to be a University of Florida Gator, Matthew Alexander Burgess was born in 1977, in Gainesville, Florida. Growing up, his routine participation in fieldwork with his father, annual family vacations to Crescent Beach, and summers sailing on the Hudson River and Long Island Sound helped foster his love of science and the environment. In June 1995, Matthew graduated near the top of his class from Gainesville High School. After beginning his collegiate career as a civil engineering major, Matthew soon realized that his desire for working in the outdoors was stronger than computing integrals, so he changed his major to zoology. In the summer of 1999, he earned his Bachelor of Science degree from the University of Florida, and began working full time in the UF Zoology Department where he gained tremendous experience as a laboratory assistant and international field technician for Drs. Colin and Lauren Chapman. Matthew then accepted a job in late-2001 as a fisheries biologist for the USGS in Gainesville where he worked extensively in the panhandle rivers of Florida over the next five years. During this time, he began his graduate degree in interdisciplinary ecology at UF, and conducted the planning and field-portions of his thesis research. When his term appointment at USGS ended in 2006, Matthew then worked part-time as an employee of the Florida Program for Shark Research, at the Florida Museum of Natural History, which kept him in contact with science, and allowed him time to complete his thesis.